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Effects of prescribed fire on fuels, vegetation, and Golden-cheeked Warbler (*Setophaga chrysoparia*) demographics in Texas juniper-oak woodlands

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ABSTRACT

The Golden-cheeked Warbler (*Setophaga chrysoparia*) is an endangered songbird that breeds in mature juniper-oak woodlands restricted to Central Texas. This habitat is increasingly susceptible to crown fire due to climate change, land use change, and fire suppression. Prescribed fire is a potential tool to reduce the risk of crown fire and may be a management tool to enhance juniper-oak woodlands for breeding warblers. However, no experimental study has been undertaken to investigate how well prescribed fire can meet these goals. We conducted a before-after control-impact study on three plot-pairs within Balcones Canyonlands National Wildlife Refuge, Texas, from 2012 to 2014 to evaluate the response of fuel loads, vegetation structure, and warblers to prescribed fire. We measured fuel loads and vegetation structure in summer 2012 (pre-treatment) and 2014 (post-treatment). We burned one randomly-chosen plot within each plot-pair during February 2013 and measured fire severity in May 2013. We monitored populations of warblers to determine plot abundance and breeding success each season. Impact of the prescribed fires was highly variable across treatment plots with ~49% of points showing no effects of fire on junipers and ~9% showing high mortality in the juniper canopy. All 12 fuel and vegetation measures responded to fire in the direction expected; however, only juniper seedling density, juniper sapling density, hardwood sapling density, canopy cover, litter cover, and litter depth were significant (i.e., year × treatment effect $P < 0.05$). Warbler density decreased 23% and 40% in the two post-treatment years in response to fire but other demographics did not have significant year × treatment effects. Across all plots and years, 67–93% of males were aged after-second year ("ASY"), pairing success was high (94–100%), average breeding success was 50–63%, and mean daily nest survival was 0.957 (SE = 0.010). Return rates averaged 45% and 35% for control and treatment plots. Discrete choice analysis based on locations of males in treated plots revealed the highest probability of use was in closed-canopy woodlands that experienced low to moderate fire severity effects from the fire (canopy intact but some intermediate level of subcanopy mortality) and the lowest probability of use was where fire severity was high. A single application of prescribed fire achieved many but not all fuel and vegetation objectives. The resulting reductions in warbler densities appeared due to avoidance of areas with high burn severity, whereas areas of low to moderate burn severity had high warbler use.

1. Introduction

Fire is a natural disturbance that can change the vegetation structure at local to landscape scales depending on its severity, intensity, frequency, and extent (Graham et al., 2004). Some forest types, especially woodlands, may require fire for forest health and vigor and to be maintained in their current state (Brose et al., 2013; Ryan et al., 2013). While fire is an important disturbance for maintaining or creating woodland habitat, it can be costly for humans when not effectively managed (Stavros et al., 2014; USDA, 2015). Crown fires, fires that consume tree canopies and kill either the trees or their above ground tissues, are difficult and costly to contain and may have long-lasting impacts to the vegetation structure as well (Reemts and Hansen, 2008; USDA, 2015). The frequency of wildfires is increasing across the United States due to

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decades of fire suppression and a build-up of fuels on the ground and because climate change is causing hotter and drier conditions in many portions of the United States (USDI et al., 2009; USDA, 2015), including central Texas (USFWS, 2014). Increased duration and frequency of drought conditions will lead to higher probabilities of wildfire (Guyette et al., 2014), as already experienced in the central Texas region during the drought of 2011–2012 (Texas A and M Forest Service, 2011).

Crown fires are considered a primary threat to the Golden-cheeked Warbler (*Setophaga chrysoparia*; hereafter, "warbler") (USWFS, 2014), an endangered songbird species that breeds only in the wooded areas co-dominated by Ashe juniper (*Juniperus ashei*; hereafter "juniper") and hardwoods found in central Texas (USFWS, 1992; Ladd and Gass, 1999). Female warblers build nests primarily made of strips of bark collected from the trunks of mature junipers and adults and juveniles forage in junipers and a variety of hardwood species, mostly oaks (*Quercus* spp.) (Ladd and Gass, 1999; Marshall et al., 2013). Therefore, the presence of mature junipers and a variety of hardwood species is required for warbler breeding habitat. A crown fire started by military activities on Fort Hood, Texas, destroyed ~2100 ha of juniper-oak woodlands during February 1996. This fire resulted in many small patches of intact habitat surrounded by large expanses of dead trees. Juniper mortality was severe, with virtually no recovery evident 14 years post-fire and the area is unlikely to be suitable as breeding habitat for several more decades (Reemts and Hansen, 2008, 2013); abundance of warblers within these burned areas was concomitantly low (Baccus et al., 2007; Reemts and Hansen, 2008, 2013). These findings illustrate that reducing the risk of crown fires is a justifiable management goal for warbler conservation.

One of the most widely used management tools to reduce risk of a crown fire is to reduce surface fuel loads and decrease the vertical continuity between surface fuels and canopy fuels with prescribed fire (Graham et al., 2004; Ryan et al., 2013). Little information exists on how prescribed fire alters the fuel loads and vegetation composition in juniper-oak woodlands; yet understanding fire behavior in this area is important in maintaining or improving warbler habitat (White et al., 2010). Yao et al. (2012) reported higher oak recruitment in plots that had experienced high intensity burns and concluded that moderate intensity fire in woodlands dominated by young junipers may promote tree species diversity. Andruk et al. (2014) found controlled surface fires in combination with selective understory thinning in juniper-oak woodlands reduced density of young junipers and increased hardwood re-sprouting while leaving the canopy intact. Thomas et al. (2016) measured the vegetative characteristics of the four dominant tree species in juniper-oak woodlands in relation to crown fire and found that junipers could sustain crown fire under lower wind speeds than the three dominant oak species due to higher leaf mass per unit area and canopy bulk density. They also reported higher canopy mass values closer to the ground for junipers than the oaks, indicating junipers provide more ladder fuels to reach the canopy. Overall, current research indicates juniper-dominated woodlands result in fuel loads that are less amenable to low-intensity fire and at greater risk of crown fire (White et al., 2010; Andruk et al., 2014), and are then slow to recover from such high-intensity fires (Reemts and Hansen, 2008; White et al., 2010).

Besides increasing risk of crown fire, lack of management across the warbler's breeding range could pose significant challenges to maintaining or increasing available breeding habitat. While habitat loss and fragmentation were principal reasons listed by USFWS (1990) for listing this species under the Endangered Species Act, habitat succession through lack of oak recruitment was also cited and may be more important than previously thought (Russell and Fowler, 2002; Groce et al., 2010). Several sites across central Texas were shown to have no or little recruitment of Texas red oak (*Quercus buckleyi*) in the preceding 35–60 years (Russell and Fowler, 2002). Texas red oak is a dominant hardwood in warbler habitat and a favored foraging substrate (Marshall et al., 2013). Lack of oak recruitment may lead to lower diversity woodland stands due to increased juniper composition (Russell and Fowler, 2004; Andruk et al., 2014). These low diversity juniper woodlands support lower densities of warblers than mixed juniper-oak woodlands (Peak and Thompson, 2013; Reidy et al., 2016; Sesnie et al., 2016). Prescribed fire in combination with selective thinning of the juniper understory was effective at killing juniper seedlings and increasing hardwood sprouts but not hardwood seedlings and saplings in juniper-oak woodlands (Andruk et al., 2014). However, fire may be insufficient as a stand-alone management technique to increase hardwood recruitment in areas with high densities of white-tailed deer (*Odocoileus virginianus*) because herbivory by deer is a significant limitation in oak regeneration (Russell and Fowler, 2004; Andruk et al., 2014).

While prescribed fire may be a necessary management tool to promote woodland health and reduce the risk of crown fire, it is unknown how warbler populations will respond to changes to the vegetation structure caused by prescribed fire. Historically, management guidelines in warbler habitat recommended no or little active management such as prescribed fire or understory thinning, instead favoring less aggressive management actions such as invasive plant control, white-tailed deer and feral pig (*Sus scrofa*) culling, and selective tree removal to maintain or enhance juniper-oak woodlands (BCNWR, 2001; BCP, 2007). However, thus far there has been little research investigating warbler response to habitat management or manipulation, including prescribed fire.

Balcones Canyonlands National Wildlife Refuge (BCNWR) was created in 1992 to preserve habitat for endangered species. It is located northwest of the Austin metropolitan area, one of the fastest growing metropolitan areas in the United States since 1990 (Groce et al., 2010; U.S. Census Bureau, 2012). Rapid growth in the area around BCNWR has expanded the exurban-wildland interface along the borders of the refuge increasing risks to human life and property from natural disasters such as crown wildfires (Radeloff et al., 2005; USDI et al., 2009). Beyond the concern of protecting human infrastructure, one of the goals established by BCNWR is to "protect warbler habitat from wildfire and reduce hazardous fuel loads" (BCNWR, 2001). Thus, management actions should reduce the risk of crown fire in juniper-oak woodlands and minimize the negative impacts to breeding warblers. However, land managers need to understand the potential impacts of prescribed fire on warbler populations before its use is recommended on a larger scale. We conducted an experimental study using a before-after control-impact design to investigate the response of fuel loads, vegetation structure, and warblers to prescribed fire on BCNWR. We monitored warbler populations on three control and three treatment plots to compare territory density, age structure and return rates, and pairing and breeding success. Our objectives were to determine (1) changes in fuel loads that may reduce risk of crown fire, (2) changes in vegetation structure in response to prescribed fire, and (3) the short-term response of warblers to prescribed fire in terms of abundance, return rates, productivity, and habitat selection.

2. Methods

2.1. Study area and experimental design

We conducted our study at BCNWR, in Travis and Burnet counties, Texas. BCNWR is ~9900 ha, of which 7700 ha have been identified to be managed for the warbler (BCNWR, 2015). BCNWR is situated in the Edwards Plateau and is a network of properties owned by the U.S. Fish and Wildlife Service in recovery region 5 (Fig. 1); this region contains some of the largest remaining patches of warbler habitat (Groce et al., 2010; Duarte et al., 2013). We identified three 40-ha plot-pairs that were similar in slope, aspect, and dominant woodland structure (Fig. 1). Each plot-pair was chosen to represent different local habitat conditions present within potential warbler habitat in BCNWR. One plot-pair was separated by 100 m of open grassland (not suitable for warblers) and two plot-pairs were separated by 200 m of contiguous woodland. Different plot-pairs were separated by >4 km. One plot-pair, "Flying X" (30°38′N, 98°5′W), was located in the northwestern part of the refuge and comprised of a mixture of mature and immature juniper-dominated and mixed juniper-oak woodland; the plots also included portions of cleared land on the fringes that were not considered breeding habitat (~8 ha). One plot-pair, "Rodgers" (30°35′N, 98°2′W), was in the central part of the refuge, comprised of a mixture of juniper-dominated and mixed juniper-oak woodland, and included closed and open canopy woodland. The third plot-pair, "Post Oak" (30°32′N, 98°1′W), was located in the southeastern portion of the refuge and contained primarily mature juniper-dominated woodland, with several narrow linear bands (10–30 m wide) of open-canopy woodland. All plots were co-dominated by juniper, with a variety of hardwoods including live oak (*Quercus fusiformis*), Texas red oak, shin oak (*Quercus sinuata*), cedar elm (*Ulmus crassifolia*), and escarpment black cherry (*Prunus serotina*). No habitat management (e.g., thinning, prescribed fire, cowbird control) had been conducted on the six plots since establishment of BCNWR in 1992; additionally no wildfires had occurred. However, evidence of past juniper clearing (i.e. cut stumps) for hunting, agricultural, and logging purposes was present within all plots.

We randomly chose one plot per plot-pair for treatment by prescribed fire resulting in 3 treatment and 3 control plots. The containment lines for each of the prescribed fires were located outside of the plot boundary creating a prescribed fire area larger than the actual study plot. We burned treatment plots in mid and late February 2013, during the dormant season and prior to the arrival

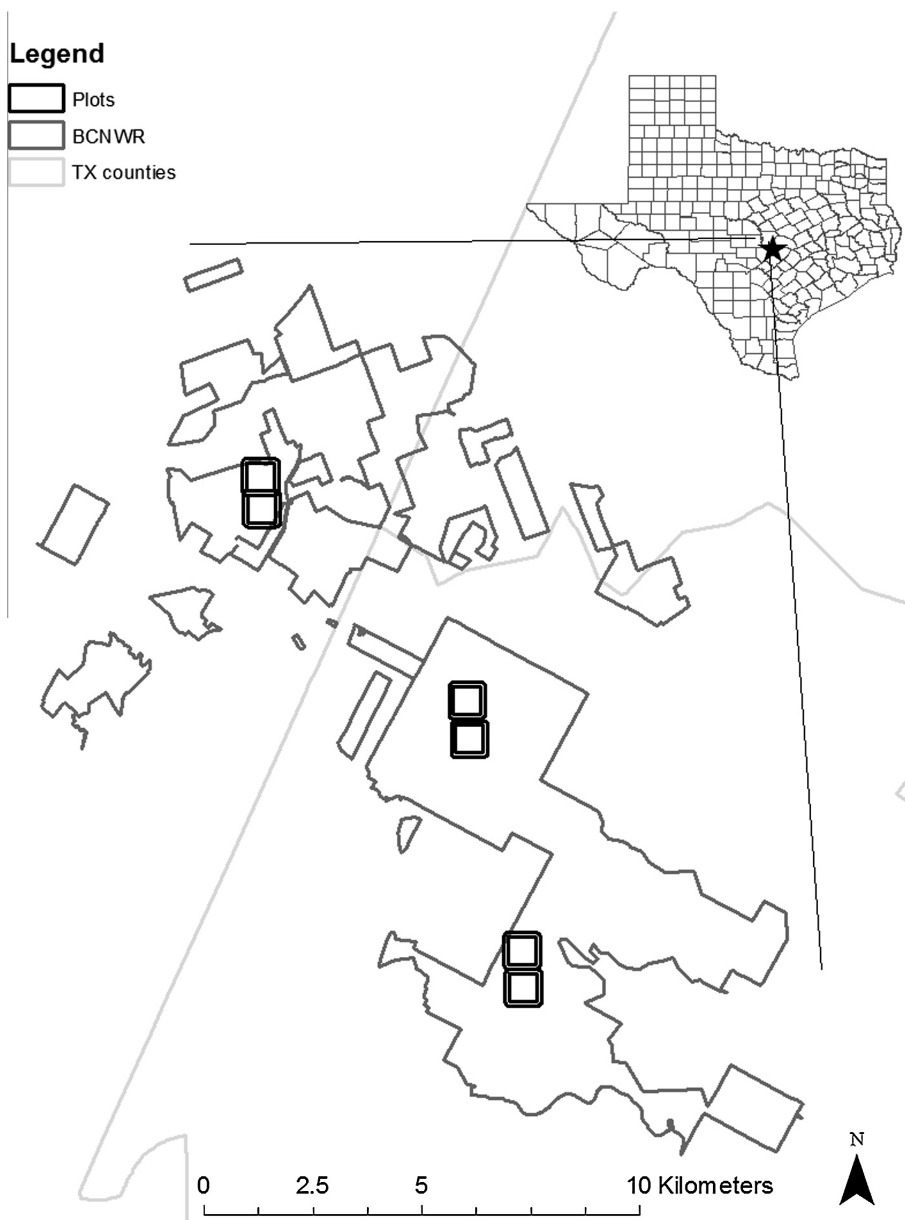

Fig. 1. Location of three sets of paired 40-ha study plots used to monitor Golden-cheeked Warblers on Balcones Canyonlands National Wildlife Refuge, Texas, 2012–2014.

of breeding warblers. The objectives for the prescribed fires were to (1) decrease 1- and 10-h surface fuel loads by 70%, (2) decrease density of seedling junipers by 70% and sapling junipers by 20%, (3) decrease density of small juniper trees by 20%, (4) increase hardwood recruitment, and (5) increase average canopy base height.

2.2. Fire severity, fuel loads, and vegetation structure

To assess direct impacts of fire on hardwoods and junipers, we established a 30-m grid of points (hereafter, fire severity points) across the treatment plots in May 2013. We rated each of these points for fire severity using a modification of the composite burn index (Key and Benson, 2006) to focus on fire severity for mature hardwoods and junipers, the key components of warbler habitat (Table 1). We recorded a separate index for hardwoods and junipers to differentiate between the re-sprouting potential of each group (juniper does not re-sprout). The woodland composition and associated fuel load varied within each treatment plot; hence, we categorized the woodland types to relate fire behavior to this variability (Table 2). We converted the fire severity point layer into a 30-m raster for each attribute in ArcMap 10 (ESRI, Redlands, CA). We also recorded juniper scorch height at each fire severity point and the mean scorch height in an 11.3 m radius around each fire severity point to evaluate the potential change to canopy base height from the prescribed fire treatment. Scorch height was recorded for juniper only because for this non-resprouting species, scorch height could be correlated to a change in canopy base height.

We measured fuel loads and vegetation structure in June 2012 and 2014 at 32 vegetation structure points on a 150-m grid on each of the six plots. We flagged and staked each point at the beginning of the study to allow for repeat measurements in subsequent years. We followed a modified FIREMON protocol to measure fuel loads (Lutes et al., 2006). We sampled fine and coarse woody debris along a 25-m transect that was centered on the point and oriented east-west. We tallied fine woody debris crossing the transect using a go-no-go gauge between the 5- and 7-m portion of the transect for the 1-h and 10-h categories. We measured slope in the field and calculated fuel loads for each category per calculations provided in Brown (1974). We did not determine species composition of downed fuel and thus used an average diameter for the composite of species and approximate specific gravities of 0.48 for diameters <2.5 cm and 0.40 for diameters >2.5 cm (Brown, 1974). We joined each point within treatment plots to the attributes of the fire severity raster cell it fell within in ArcMap to identify points impacted by fire. We tallied all live seedlings (woody stems <1.37 m tall) in a 1.8-m radius from the point and converted these counts to density (ha^{-1}) of juniper and non-juniper seedlings. Similarly, we tallied all live saplings (woody stems >1.37 m tall and diameter-at-breast height ["DBH"] < 7.5 cm) in an 11.3-m radius around the point and converted these to density of juniper and non-junipers saplings. Finally, we measured DBH of all individual live stems of all mature trees (DBH > 7.5 cm) and converted to stand basal area of junipers and non-junipers, which provided a measure of total number of and size of trees in a stand. We measured live canopy cover with a densiometer and canopy base height (estimated to the nearest half-meter) at the point and at 5-m from the point in each cardinal direction and averaged each for a single estimate of canopy cover and canopy base height around the point.

2.3. Warbler monitoring

2.3.1. Territory delineation

We monitored warbler populations on all plots from early March to early June 2012–2014. We color-banded adult warblers

Table 1
Frequency (and percent of total) of fire severity points categorized by woodland type and burn severity for junipers and hardwoods measured at 3 plots treated with fire on Balcones Canyonlands National Wildlife Refuge, Texas, in February 2013.

Burn severity	Description	Juniper					Description	Hardwood				
		W1[a]	W2	W3	W4	W5		W1	W2	W3	W4	W5
0	No fire	43 (20.5)	87 (23)	169 (68)	156 (70.5)	162 (73)	No fire	48 (23)	106 (28)	210 (85)	169 (76.5)	193 (87)
0.5	Subcanopy 100–75% green	22 (10.5)	47 (13)	40 (16)	11 (5)	25 (11)	No crown damage, no basal char	33 (16)	62 (17)	6 (2.5)	5 (2)	9 (4)
1	Subcanopy 25–75% green	41 (19.5)	65 (17.5)	19 (8)	21 (9.5)	13 (6)	No crown damage, evidence of basal char	35 (17)	114 (30.5)	18 (7)	18 (8)	10 (4.5)
1.5	Subcanopy <25% green	26 (12)	59 (16)	7 (3)	11 (5)	3 (1)	Less than 10% top kill, stem or canopy equivalent	34 (16)	41 (11)	7 (3)	11 (5)	5 (2)
2	Subcanopy scorched, upper canopy green	36 (17)	70 (19)	4 (2)	10 (4.5)	9 (4)	10–50% top kill, stem or canopy equivalent	32 (15)	30 (8)	4 (2)	12 (5.5)	2 (1)
2.5	Upper canopy scorched	26 (12)	37 (10)	6 (2.5)	9 (4)	8 (4)	Greater than 50% top kill, stem or canopy equivalent	17 (8)	12 (3)	0 (0)	5 (2)	0 (0)
3	Torched	16 (8)	8 (2)	2 (1)	3 (1)	1 (0.5)	Torched	11 (5)	8 (2)	2 (1)	1 (0.5)	2 (1)
Total		210	373	247	221	221		210	373	247	221	221

[a] Woodland types are described in detail in Table 2.

Table 2
Description of woodland types used for discrete choice analysis of habitat use of male Golden-cheeked Warblers on 3 plots treated with prescribed fire on Balcones Canyonlands National Wildlife Refuge, Texas, 2013–2014.

Woodland type	Canopy cover (%)	Dominant Canopy	Dominant understory (>1.2 m tall)	Surface fuel composition	Canopy base height (m)
W1	>60	Texas red oak/Ashe juniper	Open	Deciduous	1.8
W2	>60	Texas red oak/Ashe juniper	Juniper	Deciduous	0.6
W3	>60	Ashe juniper	Juniper	Any	1.8
W4	>60	Live oak/shin oak/Ashe juniper	Any	Any	0.6
W5	30–60	Ashe juniper or Mixed	Any	Any	0.6

from March through early May 2012–2014 to enable accurate delineation of territories and monitor productivity. We captured unbanded adult warblers in mist nets and attached 2–3 color bands and 1 U.S. Geological Survey numbered aluminum band to legs of each unbanded individual we captured. We determined the age of each banded adult male or female as second-year ("SY") or after-second-year ("ASY") using criteria established by Peak and Lusk (2010) and Pyle (1997). If an adult could not be reliably aged, we recorded the age as after-hatch-year ("AHY"). Banding was performed under permits # 23615 and 22593 and color combinations were issued by Fort Hood. We searched for banded adults within 40-ha plots and also all woodlands within a 100-m buffer around each plot.

We mapped territories of male warblers and recorded locations of males, females, and juveniles. Three to five observers surveyed each plot 2–4 times per week from early March through mid-June 2012 and 2014 and one observer surveyed each plot once per week from early March through mid-June 2013. Observers were assigned primary plots where they spent the majority of their time, but also rotated through plots to help as needed and to reduce any potential observer bias; this included a second observer surveying each plot 2–3 times across the season in 2013. Surveys lasted 6–9 h depending on plot density and warbler activity levels. Observers traversed the entire plot at least once per week to locate warblers. Additional weekly surveys focused on gathering productivity data for previously identified territories (2012 and 2014). During each survey, observers searched for adult and juvenile warblers, re-sighted color-banded and unbanded adults, and recorded locations in Universal Transverse Mercator coordinates (Garmin GPS unit). Observers collected as many locations for each male as possible during every survey and attempted to accumulate $\geqslant$15 locations per territory across the season. Observers used distinguishing features of males (songs or plumage) and females (e.g., throat patterns) to help differentiate territories prior to banding, as well as cumulative observations of banded and unbanded males, contemporaneous singing, nest locations, and fledgling age to identify territories. We considered a male territorial if we documented him on the plots for a minimum of 4 weeks. This definition eliminated non-territorial males that were recorded on the plots only in March and likely left the plots after failing to acquire a mate or those seen only in May or June that were likely prospecting for future territories or moving with dependent juveniles.

To determine the number of territories in each plot, we calculated the territory centroid for all territorial males monitored (in the plot and buffer) and summed the total number of territory centroids that fell within the plot boundaries. We created minimum convex polygons by bounding observations of individual males assigned a unique identification and calculated the area of polygons in ha to approximate territory size. We deleted observations considered extra-territorial by observers; these observations were typically of silent males following pre-nesting females or family groups, or feeding mobile fledglings. We categorized males of unknown age ("AHY") as SY to simplify the age structure analysis. We calculated return rate in 2013 and 2014 as the proportion of males banded in 2012, or 2012 and 2013, respectively, that returned to a plot. We considered a male as a "return" if his territory centroid was within the plot the year he was banded and he returned in a subsequent year to the same plot he originally settled on (territory centroid within plot or buffer).

2.3.2. Nest survival and productivity

We monitored warbler nests from late March through early June 2012 and 2014. We located nests using a combination of parental behavior and systematic searching. We monitored nests every 2–4 days, and more frequently near predicted hatch and fledge dates, until the chicks fledged or the nest failed. If we did not confirm fledglings on the expected fledge date, we continued monitoring that territory for evidence of fledging or new nesting activity. We collected vegetation measurements at each nest during June 2012 and 2014. We averaged canopy cover measured with a densiometer below the nest facing each cardinal direction. We tallied each live, healthy stem at 10 cm above ground for all woody stems >50 cm tall and <2.5 cm thick within a 5-m radius of the nest and converted this to a small stem density (stems ha^{-1}). We recorded DBH of all live stems for trees >2.5 cm and converted to stand basal area ha^{-1}. We recorded slope in the steepest direction.

We searched for fledglings for every territory from mid-April through early June 2012 and 2014, regardless of whether a nest was located, and recorded a description and behavior of fledglings to estimate age and aid in future identification. We attempted to obtain a complete count of fledglings produced for each territory. If an initial search produced four fledglings being fed by the male, female, or pair, we considered the fledgling count complete and focused on other territories in future surveys. We continued searching for fledglings in all territories until we confirmed four fledglings or until the end of season, at which time we used the highest number of fledglings observed in the territory being attended by one or both of the territorial adults. We additionally attempted to locate nests and verify total number of fledglings for additional nest attempts made by successful pairs (i.e., double-brooding).

We categorized a male as paired if he was seen with a female, attended a nest, or fed juveniles. We considered a territory successful if we observed an adult feeding fledglings; unsuccessful if we never observed one of the adults feeding fledglings; or unknown if we failed to observe the adults during the fledgling period (late April through June). We estimated productivity as the greatest number of fledglings observed for a territory (we summed fledglings by territory regardless of the number of nesting attempts). We also evaluated the average number of fledglings per successful territory across years and treatment types.

2.4. Data analysis

We used a before-after control-impact (BACI) design to investigate the response of fuel loads, vegetation structure, and warblers to prescribed fire. We analyzed each response variable in a generalized linear mixed model (PROC GLIMMIX, SAS 9.3, SAS Institute, Cary, NC) using the appropriate distribution for the response. For fuel and vegetation responses, we used a Gaussian or gamma distribution. For count responses such as number of territories or fledglings, we used a Poisson distribution and log link. For binomial responses such as pairing and breeding success (successful or not successful), age structure (SY or ASY), and return rate (did return or did not return), we used a binomial distribution and a logit link. These distributions resulted in Pearson residuals with mean and variance close to 0 and 1, respectively. All models included treatment (control or treatment plot), year, and the interaction of year and treatment as fixed effects; we assessed the year × treatment interaction as a test of the treatment effect indicating whether a response variable varied more on treatment plots than control plots between pre-treatment (2012) and post-treatment (2013 and 2014) years. We also calculated the percent change in fuels, vegetation structure, and warbler density that was related to treatment by subtracting the percent change from pre-treatment to post-treatment on control plots from the percent change on treatment plots to remove temporal effects not related to treatment. An advantage of our paired BACI design is the ability to isolate treatment effects from other temporal effects (Stewart-Oaten et al., 1986).

We modified our BACI design for analysis of fuels and vegetation because only a portion of the treatment plots burned and we wanted to compare response between vegetation structure at points that actually burned or did not burn. We categorized these points as burned (burn severity >0) and nonburned (burn severity = 0) such that points on treatment plots could be burned or nonburned. We included year as a random effect with subject = point to account for repeated measures on points and plot as a random effect because plots could have burned or nonburned points on them.

For our analysis of warbler density, we included woodland area as an additional fixed effect because one pair of plots had less woodland than the other plots and woodland represented available habitat. We included year as a random effect with subject = plot to account for repeated measures on plots and block as a random effect to account for the paired plots in our analysis of territory density. For responses with multiple measures per plot within a year (e.g., territory success, productivity, age), we included plot as an additional random effect. We used a Toeplitz covariance structure for the random year effect in the model for number of territories because we hypothesized an autoregressive covariance structure across the three years. We used a simpler variance components structure for all other random effects because they were based on 2 years or within plot effects. We compared the difference in return rates between paired control and treated plots in the two post-treatment years because we could not calculate return rates for the initial pre-treatment year and use our BACI design.

We calculated daily nest survival using the logistic exposure method (Shaffer, 2004) using PROC GENMOD (SAS 9.3, SAS Institute Inc., Cary, NC, USA) with a binomial distribution and the logit link. We defined a nest interval as successful if the nest was still active or had successfully fledged young (success = 1) and as unsuccessful if the nest was not active since the last successful interval (success = 0). We evaluated a treatment model (treatment type, year, and year × treatment) and 4 additional models that included either a single habitat feature affected by fire (canopy cover, total basal area, or small stem density) or slope, which can affect fire behavior and warbler habitat. All models included year and cubic relationship of day of year because of the strong influence of these temporal factors on nest survival (Reidy et al., 2009; Peak and Thompson, 2013). We included a repeated effect specifying plot as the subject to address monitoring multiple nests per plot across two years; the GENMOD procedure used a General Estimating Equation to adjust parameter estimates to account for correlations among responses (Liang and Zeger, 1986).

We used discrete choice analysis to estimate the probability a male warbler used different combinations of woodland types and burn severity. We joined each male warbler observation within treatment plots in 2013 and 2014 to the attributes of the fire severity raster cell it fell within using ArcMap. We only used data from the three treated plots in 2013 and 2014 because we wanted to assess habitat use when individuals had the choice of burned or nonburned habitats. Discrete choice models calculate the probability of an individual selecting a resource as a function of the attributes of that resource and all other available resources (Cooper and Millspaugh, 1999). We defined available resources relative to each male observation by randomly selecting points within a 200-m radius of an observation; this distance represented a distance that was within the range of the minimum and maximum dimensions of observed territories during 2014 and also defined an area at the upper end of observed territory sizes and we thought likely defined resources available to a warbler. We described resources in terms of woodland type and burn severity. A preliminary analysis comparing burn severity scores for juniper, hardwood, and an average of junipers and hardwoods indicated the average was more influential so we used the average burn severity score and rounded each score to the nearest whole number to assign a value of 0–3. Woodland type and burn severity were treated as categorical variables in a model with each factor and their interaction because we hypothesized the effect of burn severity would vary non-linearly and by woodland type.

3. Results

We mapped fire severity across 1272 fire severity points within the treated plots. Fire severity varied greatly across plots with a burn severity rating of 0 (no fire effects) for 49% and 57% of points for junipers and hardwoods, respectively (Table 1). Approximately 9% of points showed complete scorch in the canopy for junipers (Table 1). Burn severity was highest in treatment types categorized as W1 (Texas red oak-juniper woodland with an open understory) or W2 (Texas red oak-juniper woodland with a juniper understory), which represented ~46% of points (Table 1). Mean scorch height at the 11.3 m radius was 2.1 m for W1 and W2 woodland types and 0.5 m for W3 (juniper-dominated) and W4 (live oak/shin oak/juniper mixture) woodland types.

We assessed year × treatment effects on fuels and vegetation based on 54 burned and 138 non-burned vegetation structure points from the 6 plots. One- and 10-h fuel loads were 15% and 1% lower in response to treatment (after removing temporal effects), but the year × treatment effect was not significant ($P > 0.05$; Table 3). Juniper seedling and sapling density were both 27% lower in response to treatment and the year × treatment effect was significant for both (Table 3). Non-juniper seedling density was 6% greater and sapling density 43% lower in response to treatment but the year × treatment effect was only significant for sapling density (Table 3). Juniper basal area decreased 4% and non-juniper basal area increased 5% in response to treatment but the year × treatment effect was not significant for either (Table 3). Canopy cover decreased 14% and canopy base height increased 7% in response to treatment but the year × treatment effect was only significant for canopy cover (Table 3). Litter cover and litter depth decreased 9% and 14%, respectively, in response to treatment and the year × treatment effect was significant for both (Table 3).

We banded 68 territorial males whose centroids were within the plots (29, 17, and 22 in 2012, 2013, and 2014, respectively), of which 38 were banded in control and 30 in treatment plots (Table 4). We aged 21 males as SY (31%), 44 as ASY (65%), and 3 as AHY (4%). We additionally banded 4 females. Overall, delineations of 61% (92 of 150) of territories with centroids within the

Table 3
Fuel loads and vegetation structure before (2012) and after (2014) prescribed fire treatments across 192 points on 3 plots in juniper-oak woodland on Balcones Canyonlands National Wildlife Refuge, Texas, and the significance of year, treatment (Trt), and year × treatment effects based on repeated measures generalized linear models (F tests, df = 1, 10; 1, 370; and 1, 370; respectively).

	Non-burned (n = 138)				Burned (n = 54)				P-value		
	Before		After		Before		After				Year × trt
Variable	Mean	SE	Mean	SE	Mean	SE	Mean	SE	Year	Trt	
1-h fuel loads (Mg/ha)	0.56	0.06	0.50	0.05	0.70	0.10	0.52	0.07	0.147	0.310	0.182
10-h fuel loads (Mg/ha)	12.7	1.5	2.1	0.3	12.9	2.3	2.0	0.35	<0.001	0.855	0.682
Juniper seedling density ha^{-1}	4759	6013	2921	4618	4420	4734	1510	2628	<0.001	0.070	0.006
Non-juniper seedling density ha^{-1}	8193	1024	4702	588	11,990	1024	7629	1504	<0.001	0.039	0.589
Juniper sapling density ha^{-1}	534	43	460	38	629	78	372	46	0.002	0.844	0.005
Non-juniper sapling density ha^{-1}	73	24	66	22	103	40	49	19	0.363	0.949	0.011
Juniper tree basal area ha^{-1}	17.0	1.7	10.4	1.1	14.6	1.8	8.4	1.1	0.003	0.062	0.555
Non-juniper basal area ha^{-1}	3.5	0.3	2.4	0.3	5.3	0.9	3.9	0.7	0.002	0.016	0.520
Canopy cover (%)	70.3	4.9	57.1	3.9	77.4	6.1	52.0	4.1	0.008	0.986	<0.001
Canopy base height (m)	1.9	0.3	1.3	0.2	2.0	0.4	1.5	0.3	0.135	0.503	0.554
Litter cover (%)	56.7	1.9	52.7	1.8	57.8	3.1	48.8	2.7	<0.001	0.628	0.051
Litter depth (mm)	2.64	0.24	1.36	0.12	3.02	0.35	1.14	0.13	<0.001	0.797	0.014

Table 4
Number of banded territorial Golden-cheeked Warbler males (and percentage of total by group) aged second year (SY) and after second year (ASY) (banded within the year plus returning males in 2013 and 2014) on control and treatment plots on Balcones Canyonlands National Wildlife Refuge, Texas, spring 2012–2014.

	Control		Treatment	
Year	SY	ASY	SY	ASY
2012	2 (14)	12 (86)	4[a] (27)	11 (73)
2013	1 (7)	13 (93)	4 (33)	8 (67)
2014	9[b] (32)	16 (64)	4 (33)	8 (67)

[a] Includes 2 males aged AHY.
[b] Includes 1 male aged AHY.

Table 5
Number of Golden-cheeked Warbler males (and percentage of males banded in age group) that returned in the first year and second year post-fire that were banded the previous year on Balcones Canyonlands National Wildlife Refuge, Texas, spring 2012–2014.

	Control		Treatment	
Year	Return	%	Return	%
2013	5	45	3	24
2014	8	46	10	48

plots were based on banded males. The proportion of SY versus ASY males (within plots) did not vary by year × treatment ($F_{2,82} = 1.12$, $P = 0.33$). Because of the uncertainty in ages of unbanded males, we were unable to determine recruitment.

Twenty (10 in control and 10 in treatment plots) of 46 male warblers were returns (this included one male that did not count as territorial during the year after banding and two males whose territory centroid shifted off plot but within the buffer in the year after banding). We additionally documented three dispersal movements ranging from 225 to 430 m from median location between years; these males were not considered returns because their territory centroid moved beyond the plot buffer. Eight of 29 males (28%) banded in year 1 were resighted in year 2 and seven (24%) were resighted in year 3 (including one that was not resighted in year 2; Table 5). Eleven of 17 (65%) males banded in year 2 were resighted in year 3. Return rate did not differ by year banded × treatment ($F_{1,62} = 1.19$, $P = 0.28$); however, the return rate for treatment plots in 2013 was ~ half as high than in 2014 or on control plots (Table 5).

We monitored 150 warbler territories (59, 45, and 46 in 2012, 2013 and 2014, respectively) whose centroids fell within the monitoring plots. The number of territories was 23% and 40% lower on treated versus control plots in 2013 and 2014, respectively (Table 6), and there was a significant year × treatment effect ($F_{2,10} = 17.852$, $P = 0.0005$). Concordant with that, average territory size was 59% larger in response to treatment in 2014 than 2012 (year × treatment effect, $F_{1,96} = 15.84$, $P = 0.0001$); we opted to exclude 2013 because of lower survey effort and concomitant lower number of observations. We also detected a significant effect of year ($F_{1,96} = 26.27$, $P < 0.0001$) and treatment type ($F_{1,96} = 13.41$, $P = 0.0004$) on territory size. Observers recorded ~ twice as many locations in 2014 as 2012 so we included the number of observations in the model; the annual variation may have resulted from other observed changes to vegetation structure unrelated to the fire. However, the greater increase in territory size on treatment plots (77%) than control plots (19%) is indicative of a treatment effect (Table 7).

Table 6
Number (n) and density (males ha^{-1}) and 95% confidence intervals of Golden-cheeked Warbler males whose territory centroid fell within the plot boundaries on Balcones Canyonlands National Wildlife Refuge, Texas, spring 2012–2014.

	Control		Treatment	
Year	n	Density (CL)	n	Density (CL)
2012	31	0.27 (0.14–0.40)	28	0.24 (0.05–0.43)
2013	27	0.24 (0.05–0.42)	18	0.16 (0.13–0.19)
2014	30	0.26 (0.09–0.44)	16	0.14 (0.04–0.24)

We detected females for 101 of 105 territories (96%) in 2012 and 2014 (Table 7) and did not analyze it further because pairing success was high across all years and plots. We confirmed 216 fledglings for 61 (58%) territories (Table 7). Territory success did not differ by year × treatment ($F_{1,97} = 0.07$, $P = 0.80$). There was also no effect of year × treatment on the average number of young produced for territories of known reproductive status ($F_{1,83} = 0.06$, $P = 0.81$), the average number of young produced across all territories with unknown territories (14 territories; 13%) assigned zero young ($F_{1,97} = 0.27$, $P = 0.61$), or the average number of young produced per successful territories ($F_{1,53} = 0.60$, $P = 0.44$). Treatment plots had similar productivity but produced fewer total fledglings post-fire due to lower number of territories (Table 7).

We monitored 26 nests in 2012 and 33 in 2014 (we excluded 8 additional nests monitored for territories whose centroids fell outside our plots). We considered 13 (50%) and 19 (57%) nests successful in 2012 and 2014, respectively. We excluded 2 nests monitored in 2014 for which we lacked vegetation data from the daily nest survival analysis but included them in the productivity analysis. Daily nest survival averaged 0.957 (SE = 0.010; Table 5).

Table 7
Territory size, nest success and productivity of Golden-cheeked Warbler territories on Balcones Canyonlands National Wildlife Refuge, Texas, spring 2012 and 2014.

		Control				Treatment			
		2012 (n = 31)		2014 (n = 28)		2012 (n = 30)		2014 (n = 16)	
Variable	n	Mean	SE	Mean	SE	Mean	SE	Mean	SE
Territory size	105	3.03	0.38	3.60	0.39	3.49	0.39	6.19	0.45
Proportion of paired territories	105	0.94	0.04	0.97	0.03	0.96	0.03	1.00	0.00
Proportion of successful territories	105	0.61	0.10	0.63	0.10	0.54	0.10	0.51	0.13
Average number of fledglings per territory[a]	105	2.18	0.42	2.02	0.41	2.15	0.44	1.63	0.46
Average number of fledglings per territory[b]	91	2.51	0.52	2.39	0.50	2.22	0.48	2.19	0.60
Average number of fledglings per successful territory	61	3.58	0.25	3.21	0.23	4.03	0.29	3.22	0.34
Average number of fledglings per successful nest	32	3.61	0.54	3.26	0.41	3.79	0.54	3.23	0.45
Total number of fledglings	105	21.7	6.8	19.4	6.2	19.4	6.2	8.3	3.3

[a] Includes 14 territories with unknown reproductive output (no fledglings detected) and assigned a zero for number of fledglings.
[b] Excludes 14 territories with unknown reproductive output.

Table 8
Descriptive statistics for vegetation and habitat parameters measured around Golden-cheeked Warbler nests monitored on plots in Balcones Canyonlands National Wildlife Refuge, Texas, spring 2012 and 2014.

	Control						Treatment					
	2012 (n = 12)			2014 (n = 17)			2012 (n = 14)			2014 (n = 14)		
Variable	Mean	SD	Range	Mean	SD	Range	Mean	SD	Range	Mean	SD	Range
Canopy cover (%)	85	6	73–91	84	8	69–92	84	7	70–91	75	11	50–88
Total basal area (ha^{-1})	25	6	15–33	23	12	16–57	24	7	11–33	15	7	7–29
Small stem density (ha^{-1})	5042	4628	125–14,125	3390	3054	250–9750	6182	7559	750–28,000	9812	10,676	375–33,500
Slope (%)	15	8	4–26	12	8	3–31	17	12	2–36	17	13	1–37

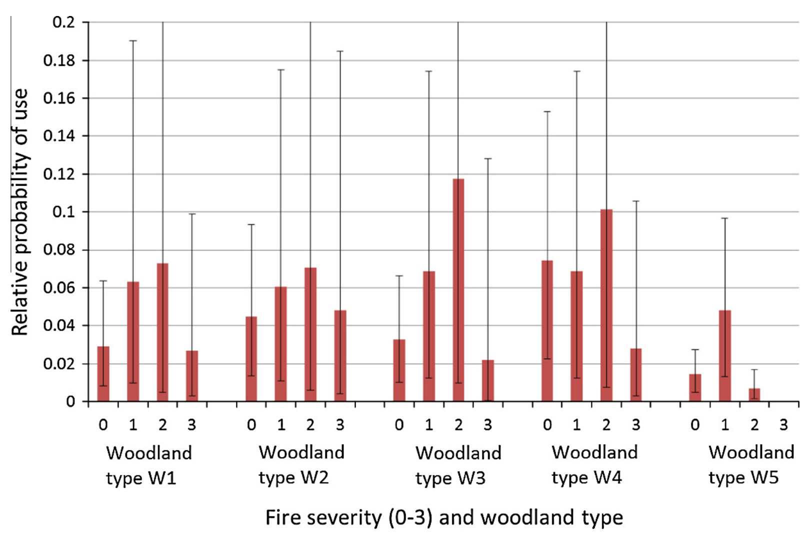

Fig. 2. Relative probability of habitat use as a function of burn severity within woodland types based on 1359 observations of male Golden-cheeked Warblers from 32 territories on three 40-ha study plots treated with prescribed fire on Balcones Canyonlands National Wildlife Refuge, Texas, 2013–2014. Relative probability of use was estimated by discrete choice analysis comparing each male observation to 4 randomly located points within a 200-m radius and indicated a significant ($P < 0.0001$) woodland type × burn severity interaction. Large confidence intervals in probability of use are the result of variation among individuals.

There was no effect of year × treatment ($\chi^2_1 = 0.03$, $P = 0.872$), canopy cover ($\chi^2_1 = 0.10$, $P = 0.754$), total basal area ($\chi^2_1 = 0.10$, $P = 0.757$), or small stem density ($\chi^2_1 = 1.51$, $P = 0.219$) on daily nest survival but there was a marginally negative effect of slope on nest survival ($\chi^2_1 = 2.92$, $P = 0.087$). We confirmed 110 fledglings from 32 successful nests (3.4 ± 0.9 fledglings per successful nest). We collected vegetation data around 57 nests. We found a significant year × treatment interaction for total basal area ($F_{1,45} = 3.18$, $P = 0.08$), but not for canopy cover ($F_{1,45} = 1.13$, $P = 0.29$), small stem density ($F_{1,45} = 0.75$, $P = 0.39$), or slope ($F_{1,45} = 0.50$, $P = 0.48$). Total basal area was lower around nests in treatment plots post-fire than in control plots or treatment plots pre-fire (Table 8).

We used observations for territorial males with >15 observations and whose territory centroid was within the plot for the discrete choice analysis; this resulted in 32 territories and 1359 observations from 2013 and 2014. Woodland type ($\chi^2_{4\ df} = 41.9$, P < 0.0001), burn severity ($\chi^2_{3\ df} = 593.1$, P < 0.0001), and the interaction of woodland type × burn severity ($\chi^2_{12\ df} = 468.1$, P < 0.0001) all had a significant effect on the relative probability of use. The effect of burn severity on the relative probability of use varied greatly among woodland types, but high burn severity always had the lowest probability of use within any woodland type and across all burn severities open canopy woodland had the lowest probability of use (Fig. 2). Although effects of woodland type and burn severity were highly significant, estimates of the relative probability of use of specific combinations of woodland type and burn severity often had large confidence intervals, likely reflecting variability among individual males in habitat use.

4. Discussion

Wildfires are expected to increase in frequency and intensity in central Texas. Concomitantly, the wild-urban interface is growing throughout much of the warbler's range (Radeloff et al., 2005; Groce et al., 2010). Land managers are increasingly tasked with protecting both wildlife habitat and human property. We evaluated the short-term, immediate effects of prescribed fire on warblers and the woodland habitats they inhabit using a paired BACI design. The paired BACI design was effective in isolating treatment effects from background temporal change that occurred during the study; we found significant year × treatment effects on many fuel and vegetation variables of interest. Evidence of non-treatment related temporal change was seen in many variables and could have resulted from ecological processes such as growth, mortality, and decomposition, weather-related phenomena such as drought, or herbivory. Likewise, Andruk et al. (2014) reported annual variation in similar measurements on control and treatment plots from a 5-year study on prescribed fire effects at BCNWR. Another source of temporal variation could be due to observer effects in data collection between years. While we attempted to control for different observers each year through training and use of quantitative measurements, measurements such as canopy base height, which was estimated, were more subjective and prone to observer error, and may be the reason we found an unexpected decrease from pre- to post-treatment. However, because of our BACI design, we were still able to isolate the treatment effect from the temporal effect and we found a positive, but not significant, response in canopy base height due to fire. Scorch height for juniper could result in an increase in canopy base height, and if so, the average scorch height for W1 and W2 woodland types (2.1 m) would be significant enough to reduce the crown fire transition for those woodland/fuels types. This would suggest that prescribed fire may be more effective at reducing the risk of crown fire in W1 and W2 woodland types than other local woodland types.

The effect of prescribed fire on all 12 fuel and vegetation variables was in the direction expected; however, for half of these variables the magnitude of change was not as great as expected and the year × treatment effect was not statistically significant. The greatest effect of the prescribed fire on vegetation structure was the 27–43% reduction in juniper seedlings and saplings and non-juniper saplings. Other large and significant changes were in canopy cover and litter depth and cover. While 1- and 10-h fuel loads were reduced by 15% and 1% respectively, this was not to the extent expected or significant; however, 1-h fuel loads decreased ~2.5 times more on burned points than non-burned points. Some of the smaller than expected responses could have been because fires were highly variable across plots, resulting in a mosaic of fire effects, with only 56% of vegetation structure points within treatment plots showing signs of burning. Fire had more impact in woodland types W1 and W2, with 78% and 30%, respectively, of the fire severity points having a burn severity >0. This was because these woodland types had a deciduous hardwood component to create a fuel layer to carry surface fire whereas the other woodland types were predominantly juniper, which had a compact needle layer that was not conducive for carrying surface fire. We measured fuel loads and vegetation structure on a random grid across each plot to allow for repeated sampling. However, this sampling design may not have been optimal for detecting changes across our treatment plots and sampling based on woodland types may have provided more insight into fire effects in these woodlands. Additional variation in fire effects may have resulted from woodland type, slope, or aspect, which we did not examine in this study because we focused on variables that we predicted would affect risk of crown fire or affect warblers.

One goal of the treatments was to increase oak regeneration. We detected a small positive but non-significant response in hardwood seedling density. However, the short duration of this project likely limited our ability to detect such changes; re-measuring these points in the future may reveal a positive effect from the prescribed fire on hardwood recruitment. Also, this was the first application of prescribed fire to these areas; hardwood recruitment may require multiple treatments (Brose et al., 2013), a mixture of treatments such as thinning and fire (Andruk et al., 2014), or a different approach such as deer management (Russell and Fowler, 2002; Andruk et al., 2014). The fires caused some tree mortality and reduced canopy cover, but the overall stand basal area was not compromised. Some canopy gaps in the treatment plots may allow for future hardwood regeneration. We focused on simple vegetative responses to fire in this study; future investigations of finer-scale vegetation measurements may reveal more significant changes to hardwood recruitment.

Warbler density was significantly lower on post-treatment plots. All of the control plots were relatively stable for the three years of this study, fluctuating 1–2 territories per plot across years. The treatment plots supported similar number of territories as the control plots in the first years but 31–47% fewer in the two years post-fire. Density was fairly stable for the two years after the fire, differing by only 1–2 territories across treatment plots. Warbler numbers also declined but quickly stabilized following crown fire in woodlands on Fort Hood (Baccus et al., 2007); however, numbers remained lower on burned areas even 10-years post-fire. We also found territories were significantly larger on post-treatment plots. Areas that supported many territories before the fire supported fewer territories after the fire. We speculate that this decrease in density and concomitant increase in territory size is related to avoidance of areas that functioned as low quality habitat or that experienced high tree mortality, possibly due to reduced insect availability, fewer nesting sites, or fewer singing perches.

Despite documented changes to warbler density and territory size, we found no evidence that the prescribed fire affected warbler productivity. Almost all males successfully paired before and after the fire, and pairing and breeding success were similar to other areas of the warbler's range (City of Austin et al., 2014). We were unable to assess warbler productivity in the year immediately after the fire due to lower survey effort, but found similar nest survival and number of fledglings per successful nest between control and treatment plots before the fire and the second year after the fire. Nest survival and productivity per successful nest were also similar to estimates from Balcones Canyonlands Preserve (Reidy et al., 2009) and Fort Hood (Peak and Thompson, 2014). While we found no significant effect of the treatment on age structure or overall on the return rate, the return rate was approximately half as high on treatment plots during the first year post-fire as the second year or either year on control plots. We documented (and banded) additional males in all plots that did not meet our definition of territorial (present for at least 4 weeks) or within the plot (territory centroid within the plot); however, we documented a higher number of "roving" males on treatment plots after the fire.

Our discrete choice analysis indicated warblers used all closed-canopy woodland types in areas of low-intermediate burn severity (canopy had green leaves, some amount of subcanopy killed), but use was very low in areas where the upper canopy had been scorched. The areas of highest burn severity were correlated with the woodlands typified by Texas red oak and juniper overstory and deciduous leaf litter. This combination contributed to highly flammable conditions where the surface fire ultimately climbed into the canopy and unintentionally harmed or killed mature junipers. These mixed woodlands were also correlated with bird observations, signifying warblers were associated with these closed-canopy mixed woodlands. We documented little use of open-canopy woodland within the treatment plots, despite this habitat being patchy and in close proximity to closed-canopy woodland. These low-quality woodland types also showed few effects of fire, further demonstrating the difficulty of achieving desired habitat conditions using prescribed fire in these woodlands (Andruk et al., 2014). While we did not document woodland types across control plots, male warblers tended to avoid areas of open-canopy woodlands and shrublands on all plots (J. Reidy, pers. obs.). Based on this, we postulate that mature woodland may benefit from some thinning of immature trees through either prescribed fire or mechanical removal, but that any habitat alteration that results in a significant reduction in canopy cover may also reduce the density of warblers, at least in the years immediately following fire. Our results indicate that prescribed fire affected vegetation structure the most in areas favored by warblers (mixed juniper-oak woodlands) and was ineffective at achieving management objectives in areas of low diversity juniper-dominated woodlands. Additional experimental studies may reveal whether prescribed fire under conditions other than those experienced herein or mechanical thinning is beneficial or neutral for warblers and warbler habitat, and whether thinning is perhaps a requisite tool to enhance prescribed fire impacts in juniper-dominated woodlands (Andruk et al., 2014).

5. Conclusions

We found strong evidence that plots treated with prescribed fire supported lower densities of Golden-cheeked Warblers in the immediate years post-fire. Warblers made greater use of closed-canopy woodlands that experienced low or intermediate effects from prescribed fire but avoided areas that experienced canopy mortality from more extreme fire behavior. This is important because intermediate fire effects were also correlated to the most desirable changes to the vegetation for reducing the risk of a canopy fire. The prescribed fire treatment met many but not all fuel and vegetation objectives. Multiple prescribed fire treatments or a mixture of treatments may be necessary to meet fuel loads and habitat objectives. Therefore, we suggest future studies evaluate effects of longer term impacts, effects of multiple treatments, and alternative treatments such as selective mechanical thinning and soil restoration.

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