Forest Ecology and Management 492 (2021) 119191

Contents lists available at ScienceDirect

Forest Ecology and Management

journal homepage: www.elsevier.com/locate/foreco

Effects of prescribed fire on fuels, vegetation, and Golden-cheeked warbler (*Setophaga chrysoparia*) demographics in Texas juniper-oak woodlands: An update six years post-fire

Jennifer L. Reidy [a,*], Frank R. Thompson III [b], Scott Rowin [c], Carl Schwope [c], James M. Mueller [c]

[a] Department of Fisheries and Wildlife Sciences, 302 Anheuser-Busch Natural Resources Building, University of Missouri, Columbia, MO 65211, United States

[b] U.S.D.A. Forest Service Northern Research Station and Department of Fisheries and Wildlife Sciences, 202 Anheuser-Busch Natural Resources Building, University of Missouri, Columbia, MO 65211, United States

[c] U.S. Fish and Wildlife Service, Balcones Canyonlands National Wildlife Refuge, 24518 FM 1431, Marble Falls, TX 78654, United States

A R T I C L E I N F O

Keywords:
Ashe juniper
BACI design
Demography
Density
Fire severity
Oaks
Productivity

A B S T R A C T

The juniper (*Juniperus ashei*) - oak (*Quercus* sp.) woodlands of central Texas are susceptible to crown fire due to climate change, land use change, and fire suppression. Low-intensity prescribed fire is one method used to reduce fuel loads and lower the risk of crown fire. Better knowledge of the impacts of prescribed fire on fuel loads, vegetation structure, and the endangered golden-cheeked warbler (*Setophaga chrysoparia*), a songbird that breeds exclusively in these woodlands, is needed to inform management decisions. We conducted a before-after control-impact study on three plot-pairs within Balcones Canyonlands National Wildlife Refuge, Texas, to determine the response of fuel loads, vegetation structure, and golden-cheeked warblers to prescribed fire. We measured fuel loads and vegetation structure before fire (2012), and again one year (2014) and six years (2019) after fire. We then evaluated the year × treatment interaction in each time interval to determine if there was a fire effect by fire severity score (a measure of fire impact). We found no significant year × treatment interaction on fuel loads in 2014 or in 2019, but there was a significant year × treatment interaction for juniper seedling density, juniper sapling density, non-juniper seedling density, non-juniper sapling density, canopy cover, and litter depth in one or both years. Juniper seedling and sapling density had significant declines in areas of severe fire effects in 2014, and juniper sapling density stayed lower even through 2019. Non-juniper seedling and sapling density increased dramatically in 2019 in areas that experienced severe fire effects; however, we note that the majority of sapling stems were from resprouting which is the predominate method of oak persistence in this ecosystem. Juniper seedling density, canopy cover, and litter depth all showed an interaction effect in 2014 but not in 2019, possibly indicative of recovery. We monitored golden-cheeked warblers during 2012–2014 and 2019 (although we did not determine breeding success in 2013) and found prescribed fire had varying effects on golden-cheeked warblers. There was a significant interaction for territory density in 2013 and 2014 but not in 2019; density remained similar on control plots in 2013 and 2014 but declined in 2019, whereas density declined in 2013 and 2014 but stabilized in 2019 on treatment plots. Breeding parameters were similar across time intervals, except number of fledglings per successful territory was lower on treatment plots in 2019 than 2012. Six years post-fire, golden-cheeked warblers mostly avoided areas where canopy mortality had occurred, except in woodlands categorized with a red oak-juniper canopy and a juniper understory pre-fire, where they had a relatively high probability of use in severely impacted areas. Prescribed fire achieved some management goals (e.g., lower densities of young junipers and higher densities of young oaks) but did not meaningfully reduce fuel loads in the short or intermediate time periods. Additional time is necessary to determine if oak regeneration on the treatment plots results in mature trees.

* Corresponding author.
E-mail address: jennifer.reidy@gmail.com (J.L. Reidy).

https://doi.org/10.1016/j.foreco.2021.119191
Received 11 December 2020; Received in revised form 15 March 2021; Accepted 18 March 2021
Available online 24 April 2021
0378-1127/© 2021 Elsevier B.V. All rights reserved.

1. Introduction

Juniper-oak woodlands are an important and unique vegetative community in central Texas (Diamond et al., 1995; Diamond et al., 1997). These woodlands are also the only breeding habitat of the golden-cheeked warbler (*Setophaga chrysoparia*), an endangered songbird that builds nests using Ashe juniper (*Juniperus ashei*) bark (Ladd and Gass, 1999; USFWS, 1990). Crown fire is rare in mature juniper-oak woodlands, but when it does occur, fire intensity can be high enough to induce stand-replacing fire (Diamond et al., 1995; Reemts and Hansen, 2008; White et al., 2010). Ashe juniper is slow to recover after crown fires and those areas affected by crown fire are unsuitable for breeding golden-cheeked warblers, potentially for decades (Baccus et al., 2007; Reemts and Hansen, 2008). Therefore, a primary management objective is to minimize the risk of crown fire within protected golden-cheeked warbler habitat (White et al., 2010; Andruk and Fowler, 2015).

Prescribed fire is a common method used to reduce crown fire risk (Fernandes and Botelho, 2003; Ryan et al., 2013). Low intensity fires can temporarily reduce horizontal and vertical fuel sources, thereby reducing the risk of crown fires that could kill canopy trees and devastate natural and man-made communities (Fernandes and Botelho, 2003). Additionally, prescribed fire may increase oak regeneration in these woodlands, which is currently sparse in some areas (Russell and Fowler, 2002; Brose et al., 2013; Andruk et al., 2014; Andruk and Fowler, 2015). This lack of oak regeneration is an area of management concern, particularly because golden-cheeked warbler density is highest in areas of mixed oak and juniper woodland (Peak and Thompson, 2013; Reidy et al., 2016b). However, the potential benefits from prescribed fire may be limited by the underlying composition of the vegetation community. For example, canopy tree composition and leaf litter composition influence fire behavior, and therefore the risk of crown fire, but also may limit the effectiveness of a prescribed fire (Bunting, 1986; White et al., 2010; Thomas et al., 2016). In particular, woodlands with a high proportion of juniper may be capable of sustaining and spreading crown fire at lower wind thresholds than woodlands with a higher proportion of oak due to juniper's much higher leaf mass per unit area (Thomas et al., 2016).

We performed a before-after control-impact study to evaluate the response of fuels, vegetation structure, and golden-cheeked warblers to prescribed fire within juniper-oak woodlands with varying canopy composition. We previously reported the response of these variables to prescribed fire one-year post-fire. Treatments resulted in lower densities of juniper seedlings, juniper saplings, non-juniper saplings, canopy cover, litter depth, and litter cover on treatment plots but did not affect fuel loads (Reidy et al., 2016a). There was a significant decrease in golden-cheeked warbler territory density and concomitant increase in territory size and lower number of total fledglings produced on treatment plots after the fire (Reidy et al., 2016a). In the present study, we collected new vegetation and bird data at previously measured points and plots, respectively, six years after prescribed fire to better understand longer-term effects of prescribed fire. We evaluated vegetation and warbler recovery six years after prescribed fire (e.g., examining if previously detected effects were still evident or if we discovered time-lagged effects). Evaluating the effects of fire six-years post-fire is relevant to woodland recovery because six to seven years may be the historical fire frequency in this area (White et al., 2010; Andruk and Fowler, 2015). Our results will inform land managers about the possible effects of using prescribed fire to reduce fuel load and young juniper density and promote increased density of young oaks, which are expected to increase habitat quality for the warbler in the long-term, without having negative impacts on the golden-cheeked warbler in the immediate years following fire.

2. Methods

2.1. Study area and experimental design

Balcones Canyonlands National Wildlife Refuge (hereafter BCNWR) is a discontinuous network of properties totaling ~ 10,000 ha, in Travis, Williamson, and Burnet counties, Texas, of which ~ 4,500 ha have been identified for management as closed-canopy juniper-oak woodlands. We chose three plot-pairs spread across the refuge (plot-pairs separated by > 4 km). Woodland composition (percent of juniper versus oak and dominant oak species present) varied among plot pairs, but plots within pairs were similar in slope, aspect, and woodland composition and structure. Each plot was a 40-ha square but one plot-pair had ~ 8 ha (~20%) of grassland included within the plot boundary that was not suitable as breeding habitat for golden-cheeked warblers. Ashe juniper co-occurred with different dominant oaks such as Plateau live oak (*Q. fusiformis*), shin oak (*Q. sinuata*), and Texas red oak (*Q. buckleyi)*, and a variety of other hardwood species, including cedar elm (*Ulmus crassifolia*) and escarpment black cherry (*Prunus serotina* var. *eximia*). Mean canopy cover, juniper basal area, and hardwood basal area ranged 69–78%, 11.4–15.5/ha, and 3.7–4.9/ha, respectively, across the three plot pairs. No prescribed fire or wildfire had occurred on these plots since the creation of BCNWR in 1992. Additional plot details are provided in Reidy et al. (2016b).

We collected pre-fire data in spring 2012, and post-fire data in spring 2013, 2014, and 2019. We randomly assigned one plot per plot-pair as treatment (fire) and the other as control prior to burning. Treatment plots were burned in mid and late February 2013, during the dormant season and prior to the breeding season of golden-cheeked warblers, which begins in March. The fire containment lines were bounded outside of the defined treatment plots, thus the area treated with fire was actually larger than the plots.

2.2. Fuel loads, vegetation structure, and fire severity

We measured fuel loads and vegetation structure from late May through early July 2012, 2014, and 2019 at 32 points spaced at 150-m intervals across the six plots; we opted to exclude five points during analysis because they were outside woodland habitat. We followed a modified FIREMON protocol (Lutes et al., 2006) to measure fine and coarse woody debris (described in detail in Reidy et al., 2016a). We measured attributes of vegetation we considered important for golden-cheeked warblers (i.e., canopy height, canopy cover, juniper basal area, and non-juniper basal area; Reidy et al., 2017) or that might be affected by fire and reflect changes in woodland composition and structure (i.e., density of young junipers and hardwoods, canopy base height; measurements described in detail in Reidy et al., 2016a). We rated fire severity for junipers and hardwoods (Appendix A) and recorded five woodland types (Appendix B) at separate points established from a 30-m grid across treatment plots in May 2013 using a modification of the composite burn index (Key and Benson, 2006). We converted the fire severity point layer into a 30-m raster for each attribute in ArcMap 10 (ESRI, Redlands, CA). We then assigned each point within treatment plots to the attributes of the fire severity raster cell it fell within in ArcMap to determine which points had been affected by fire.

2.3. Golden-cheeked warbler monitoring

2.3.1. Territory monitoring

We monitored golden-cheeked warbler populations on all plots weekly from early March to early June 2012–2014 and 2019. We captured and banded adult golden-cheeked warblers to enable unique identification of as many individuals as possible, increasing our confidence in determining the number of territories and productivity (Reidy et al., 2015). We performed banding under U.S. Geological Survey permits # 23615 and #22593. During each survey, we resighted birds and recorded band combinations if present. We attempted to collect ≥ 15 locations for each territorial male (i.e., male present on plot for > 4 weeks) between March and May (the primary territorial season). We delineated individual territories at the end of the field season using cumulative observations of banded and unbanded males, distinguishing characteristics such as unique song or plumage, counter singing, and nest monitoring and fledgling date data. We then bound those observations assigned to a unique male in ArcMap using minimum convex polygons and calculated the territory centroid and the territory size. We determined the plot-level territory density by summing all the territory centroids that fell within the study plots (including centroids up to 10-m outside the plot boundary) and dividing by the total woodland within each plot.

2.3.2. *Nest survival and productivity*

We surveyed each plot 2–4 times per week from early March through June 2012, 2014, and 2019, to document breeding activity and locate golden-cheeked warbler nests (we were unable to accurately assess breeding activity in 2013 due to logistical constraints). We monitored active nests every 2–4 days, and more frequently near predicted hatch and fledge dates, until the chicks fledged or the nest failed. We considered a nest as successful if we confirmed at least one golden-cheeked warbler fledgling and as failed if we did not confirm activity during at least two monitoring visits prior to the expected fledge date. 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 measured vegetation structure around each active nest during June of each year. We were specifically interested in juniper basal area, small woody stem density, and slope, because these factors affected nest survival at the nearby Balcones Canyonlands Preserve (Reidy et al., 2017) and can affect or be affected by fire behavior. We recorded DBH of all live juniper stems for trees > 2.5 cm and converted to stand juniper basal area ha^{-1}. We tallied each live 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 slope in the steepest direction using a clinometer.

We obtained a total fledgling count for every territory we monitored. We attempted to get complete fledgling counts on the day of fledge for nests we monitored. Golden-cheeked warblers typically lay 4 eggs in the first nest attempt, and 3–4 in subsequent nest attempts; they very rarely lay 5 eggs (J.L.R., pers. obs.). Therefore, we considered 4 fledglings a complete count. We recorded a description of fledglings to estimate age and aid in future identification. We searched for fledglings in all territories until we confirmed 4 (or more) fledglings or until the end of season. We used the highest number of fledglings observed in the territory being attended by one or both of the territorial adults.

We considered a territory successful if we confirmed an adult feeding ≥ 1 fledgling. If we were unable to confirm evidence of fledglings, but the adults were still present in the territory for the duration of the season, we considered the territory to be unsuccessful. In a few instances, we were unable to observe adults enough during the fledgling period (after ~ April 15) to confidently determine breeding success and we recorded success as unknown. We calculated productivity using the greatest number of fledglings confirmed for a territory, regardless of the number of nesting attempts. We also determined the average number of fledglings per successful territory across years and treatment types.

2.4. *Data analysis*

We performed a before-after control-impact (BACI) analysis in which we investigated the effects of prescribed fire on fuel loads, vegetation structure, and golden-cheeked warbler density and parameters of breeding success six years after the fire and compared those to the immediate effects of the fire in 2014 (Reidy et al., 2016a). We analyzed each response variable in a generalized linear mixed model (PROC GLIMMIX, SAS 9.3, SAS Institute, Cary, NC). We used a Gaussian or gamma distribution for continuous parameters such as fuel and vegetation responses; a Poisson distribution and log link for count responses such as number of territories or fledglings; and a binomial distribution and a logit link for binomial responses such as breeding success (successful or not successful), which resulted in Pearson residuals with mean and variance close to 0 and 1, respectively. We included treatment (control versus treatment plot for golden-cheeked warbler parameters and fire severity for fuels and vegetation), year, and the year × treatment interaction as fixed effects in all models. We were primarily interested in the interpretation of the year × treatment interaction, which indicated if the response of interest varied differently between treatment and control plots across pre-treatment and post-treatment years (we considered P-values < 0.05 as significant).

We determined the effect of fire severity (0–3; referred to as “no”, “low”, “moderate”, and “severe” fire effects in text) on fuels and vegetation because we thought this was a more informative parameter than the simpler burned versus unburned classification used by Reidy et al. (2016b). We conducted separate tests comparing pre-treatment (i.e., 2012) to post-treatment years to determine which variables were affected immediately (i.e., 2014), which remained affected several years later or became affected later (time lag; i.e., 2019), and which recovered (i.e., there was a significant effect compared to 2012 in 2014, but not in 2019); we refer to these year and interaction tests by the terminal year considered (i.e., 2014, 2019). We included year as a random effect with subject = point to account for repeated measures on points and plot as a random effect to account for other site factors not directly measured. We report the lsmeans generated from the full model with all three years, which accounted for repeated measures and created a balanced sample across years.

We similarly analyzed the effects of fire on several parameters of golden-cheeked warblers. We included woodland area as an additional fixed effect because one pair of plots had less woodland than the other plots and amount of 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 the analyses of territory density and breeding success parameters. Due to reduced survey effort in 2013, we only included golden-cheeked warbler observations in our analyses of territory density and habitat selection, not in analyses of breeding success. Additional details are provided in Reidy et al. (2016a).

We calculated daily nest survival using the logistic exposure method (Shaffer, 2004) using PROC GENMOD (SAS 9.3) 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 interaction) and three additional models with a single variable shown to affect nest survival - juniper basal area, small stem density, and slope (Reidy et al., 2017). We included year and cubic relationship of day of year in all models because of the strong influence of these temporal factors on nest survival (Reidy et al., 2009; Peak and Thompson, 2014). We included a repeated effect specifying plot as the subject to address monitoring multiple nests per plot across years. The GENMOD procedure used General Estimating Equations to adjust parameter estimates to account for correlations among responses (Liang and Zeger, 1986).

We conducted a discrete choice analysis to evaluate the relationships between observations of golden-cheeked warblers and woodland type (W1-W5) and fire severity. We restricted our analysis to observations on treatment plots after the fire (2013, 2014, and 2019) because we were interested in how golden-cheeked warblers used burned and unburned areas. Discrete choice analysis estimates the relative probability of a golden-cheeked warbler male choosing a location over other available locations as a function of the attributes of those locations (Cooper and

Millspaugh, 1999). We generated four random points 30–200 m from each male observation within the plot boundary in 2013–2014, but increased this to 10 random points in 2019 to increase precision of estimates. We joined woodland type and fire severity to each male and random location based on proximity. Discrete choice models are fit with a multinomial logit model (Cooper and Millspaugh, 1999), which we fit with PROC PHREG (SAS 9.3) and the robust sandwich estimate for the covariance matrix to account for repeated observations on individuals.

3. Results

We evaluated year × fire severity effects on fuel load and vegetation structure parameters around 187 previously measured points, of which 54 (29%) were burned to some degree (fire severity > 0). There was a significant year × fire severity effect on juniper seedling density, juniper sapling density, non-juniper sapling density, canopy cover, and litter depth in 2014, and on juniper sapling density, non-juniper seedling density, and non-juniper sapling density in 2019 (Table 1). Juniper and non-juniper sapling densities were the only variables with significant interaction effects in both time intervals. Juniper seedling density decreased across all fire severity classes from 2012 to 2014, but the effect was more pronounced in areas of moderate (2) or severe (3) fire severity (Fig. 1). By 2019, juniper seedling density was similar to pre-fire levels for all fire severity classes but the most severe, which was still lower than 2012 levels (Fig. 1). Juniper sapling density was similar across all years on unburned (0) points, but was increasingly affected by fire severity in 2014; the response was largest in areas that experienced moderate or severe fire effects (Fig. 1). Non-juniper seedling density showed a strong, time-lagged response, with density much higher in 2019 in areas that experienced severe fire effects; areas of no to moderate effects showed lesser effects (Fig. 1). Non-juniper sapling density showed a very strong, positive time-lagged response in areas that experienced severe fire severity; additionally, areas of low (1) to moderate fire severity experienced small increases six years post-fire, whereas unburned points stayed similar to pre-treatment levels (Fig. 1). Canopy cover was significantly lower in areas of severe fire severity in 2014, but had returned to similar levels in areas of lesser fire effects in 2019 (Fig. 1). Litter depth was substantially higher pre-fire in areas that experienced moderate fire severity levels, and those points had the lowest litter depth in 2014, but by 2019, litter depth was again highest in these areas (Fig. 1). In addition to these interaction effects, we also detected significant effects of year on almost all variables we evaluated and treatment effects on several variables (Table 1).

Table 1
P-values for year, fire severity score, and year × fire severity score effects on fuel load and vegetation structure parameters based on repeated measures generalized linear models comparing 2012 to 2014 (F tests, df = 1, 361; 3, 361; and 3, 361, respectively) and 2012 to 2019 (F tests, df = 1, 361; 3, 361; and 3, 361, respectively). Tests comparing 2012 (pre-fire) to 2014 (1-year post-fire) represented immediate effects while 2012 to 2019 (6-years post-fire) represented intermediate and potentially time-lagged effects and potential recovery of measured parameters.

Variable	2012–2014			2012–2019		
	Year	Severity	Year × Severity	Year	Severity	Year × Severity
1-hr fuels (Mg/ha)	0.009	0.963	0.680	<0.001	0.046	0.056
10-hr fuels (Mg/ha)	<0.001	0.744	0.879	0.722	0.903	0.548
100-hr fuels (Mg/ha)	0.378	0.120	0.625	0.440	0.465	0.745
Juniper seedling density ha^{-1}	<0.001	<0.001	<0.001	0.381	0.147	0.276
Juniper sapling density ha^{-1}	<0.001	0.452	<0.001	<0.001	0.225	<0.001
Juniper tree basal area ha^{-1}	<0.001	0.013	0.252	0.075	0.156	0.186
Non-juniper seedling density ha^{-1}	0.011	<0.001	0.396	0.004	0.002	0.042
Non-juniper sapling density ha^{-1}	<0.001	<0.001	0.005	<0.001	0.034	<0.001
Non-juniper basal area ha^{-1}	<0.001	0.049	0.371	0.006	0.107	0.710
Canopy cover (%)	<0.001	0.093	<0.001	<0.001	0.875	0.589
Canopy base height (m)	<0.001	0.082	0.183	0.002	0.177	0.068
Litter depth (mm)	<0.001	0.600	<0.001	<0.001	0.152	0.637

We monitored 187 golden-cheeked warbler territories, for which 63% were banded. We detected a significant year × treatment effect on territory density in 2014 (Table 2). Territory densities were relatively stable on control plots from 2012 through 2014, but decreased from 2012 to 2013 and 2014 on treatment plots (Fig. 2). We did not detect a significant year × treatment effect in 2019 (Table 2) because densities increased slightly from 2014 levels on treatment plots while decreasing on control plots (Fig. 2). We also detected a significant year × treatment effect on territory size in 2014, but not in 2019 (Table 2). Territory size increased more on treatment plots than control plots in 2014, but then decreased to a similar size as control plots in 2019 (Fig. 2).

Pairing success of golden-cheeked warblers was high across all years and plots (96%), so we did not analyze it further. There was a significant year × treatment effect on productivity of successful territories in 2019 (Table 2); successful territories on control plots produced similar number of young in 2012 and 2019, but the number of young produced by successful territories was lower on treatment plots in 2019 (Table 3). Our other parameters of breeding success did not have significant year × treatment effects (Table 2).

We monitored 88 active golden-cheeked warbler nests, of which 54 successfully fledged at least one host young (61%). We confirmed 179 fledglings from 54 successful nests (3.3 ± 0.1 SE fledglings per successful nest). Daily nest survival did not show a year × treatment effect ($\chi^2{}_2$ = 0.19, P = 0.66) or treatment effect ($\chi^2{}_2$ = 0.05, P = 0.82), but did show a marginal year effect ($\chi^2{}_2$ = 3.35, P = 0.07) between 2012 and 2014. Daily nest survival also did not show a year × treatment effect ($\chi^2{}_2$ = 0.01, P = 0.93) or treatment effect ($\chi^2{}_2$ = 2.11, P = 0.15), but did show a significant year effect ($\chi^2{}_2$ = 4.64, P = 0.03) between 2012 and 2019. Daily nest survival averaged 0.95 (SE = 0.01), 0.97 (SE = 0.021) and 0.99 (SE = 0.01) in 2012, 2014, and 2019, respectively. We measured vegetation around 86 nests and because there was no interaction effect in either interval, we analyzed vegetation effects on the full dataset (Table 4). There was no effect of juniper basal area ($\chi^2{}_1$ = 0.39, P = 0.53), small stem density ($\chi^2{}_1$ = 1.58, P = 0.21), or slope ($\chi^2{}_1$ = 2.09, P = 0.15) on nest survival.

We used observations for territorial males with > 10 observations for the discrete choice analysis; this resulted in 32 territories and 1,359 observations in 2013–2014 and 20 territories and 1,026 observations in 2019. We found significant effects for woodland type ($\chi^2{}_4$ = 111.6, P < 0.001), fire severity ($\chi^2{}_3$ = 999.7, P < 0.001), and the interaction of woodland type × fire severity ($\chi^2{}_{12}$ = 2705.5, P < 0.001). We found

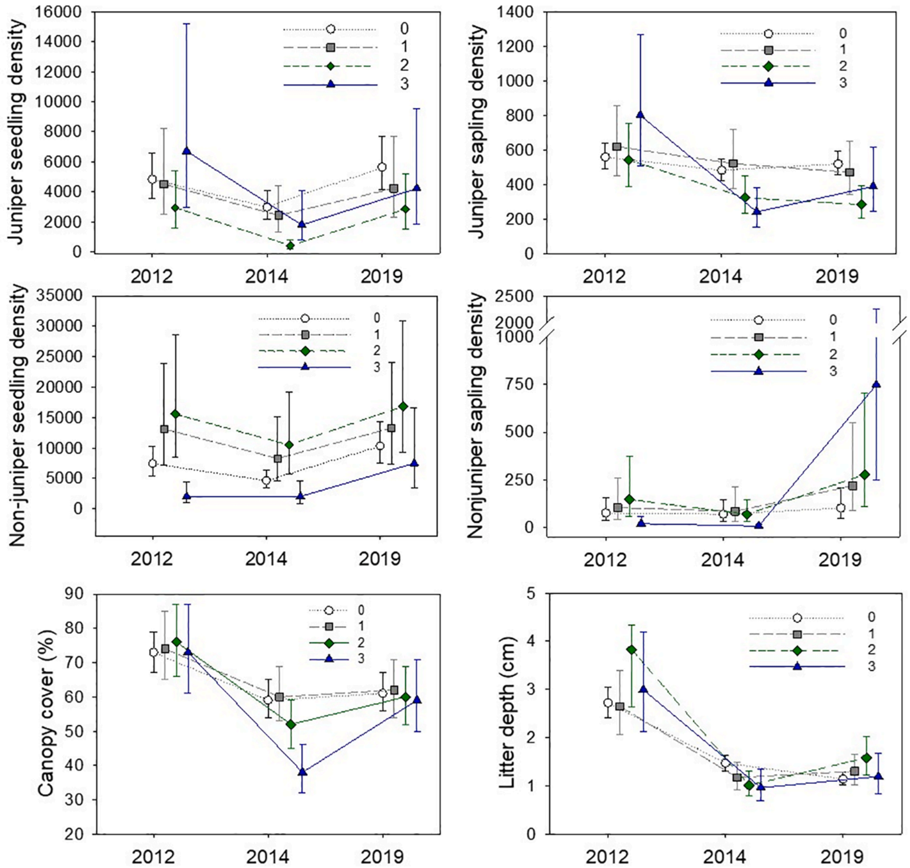

Fig. 1. Means and 95% confidence intervals by year and fire severity score for vegetation variables with significant year × treatment effects on Balcones Canyonlands National Wildlife Refuge, Texas. Year 2012 was before fire and 2014 and 2019 were 1 and 6 years after fire, respectively, and fire severity was ranked as 0 = unburned, 1 = low fire effects, 2 = moderate fire effects, 3 = severe fire effects (see Appendix A for additional details).

overall probability of use was lowest in W5 (lowest overall canopy cover) in all years, regardless of fire effects (Fig. 3). During 2013–2014, points with low or moderate fire effects generally had a higher probability of use than did unburned points or those with severe fire effects across all woodland types (Fig. 3). Habitat use differed in 2019, with more variations of use between woodland types and fire severities. The highest probability of use occurred in areas of severe fire effects in W2 (red oak-juniper canopy, juniper-dominated understory) and in areas of moderate fire effects in W3 (juniper-dominated canopy and understory) and W4 (live or shin oak-juniper canopy). However, the relative probability of use was lowest at the greatest fire severity within W1 (red oak-juniper canopy, open understory), W3, W4, and W5 (Fig. 3).

4. Discussion

Prescribed fire met some but not all management goals. One-year post-fire, we found significant fire severity effects on five vegetation parameters, but six years post-fire only juniper sapling and non-juniper seedling, and non-juniper sapling density had effects. Juniper sapling density decreased in areas with moderate and severe fire effects relative to areas with no or low fire severity in 2014 and remained lower in 2019. Non-juniper sapling density increased significantly in severely burned areas compared to lesser effected areas by 2019. Non-juniper seedling density initially decreased post fire but then increased by 2019; changes were consistent across all fire severities except areas of severe fire effects, where seedling density substantially increased by 2019, suggesting a time-lagged fire effect. Fire had no significant effect on oak seedling density in the first-year post-fire in this study or even three years post-fire in another study in these woodlands (Andruk et al., 2014), but responded positively six-years post-fire to severely affected areas, suggesting evidence of oak regeneration may occur several years after a fire and may only occur in areas of high fire intensity. Our overall results six years post-fire support lower densities of small juniper and higher densities of small non-junipers, both considered desired management outcomes and consistent with previous studies of fire in this area (Andruk et al., 2014). These changes are believed to be beneficial to golden-cheeked warblers by creating a more heterogeneous mixed woodland. However, it should be noted that golden-cheeked warbler nest survival was greater in areas of more understory cover (Reidy et al., 2017) and fledglings used areas of greater ground and understory cover in the weeks following fledging the nest (Trumbo, 2019). Prescribed fire effects were highly patchy and mostly restricted to small areas resulting in a mosaic and this patchy nature may mitigate possible negative effects to golden-cheeked warblers in the near-term while allowing for oak regeneration in the long-term.

Canopy cover decreased in all fire severities but decreased more in areas with moderate and severe fire effects in 2014. Canopy cover increased by 2019, and while still lower than pre-treatment levels, levels were similar across all fire severity classes. This suggests to us a recovery from fire effects but that some other environmental effect, such as drought, decreased levels across all fire severities. Canopy cover is an important predictor of golden-cheeked warbler density (Reidy et al., 2017) and recovery of this parameter is therefore beneficial to the golden-cheeked warbler so long as the canopy cover is similar in composition to closed juniper-oak woodlands. We did not capture canopy composition within this parameter, but we noted qualitatively that in areas of high canopy mortality, new species not associated with juniper-oak woodlands were growing and it is unknown how the canopy composition will change in these areas. Litter depth showed a similar pattern to canopy cover where it remained lower than pre-treatment

Table 2
P-values for year, treatment type (control or treatment, "Trt"), and year × treatment type effects on golden-cheeked warbler density, territory size, proportion of successful territories, overall productivity, and productivity of successful territories. We conducted tests comparing 2012 (pre-fire) and 2013 and 2014 (1-year post-fire) to assess immediate effects and 2012 and 2019 (6-years post-fire) to assess intermediate and potentially time-lagged effects and recovery of measured parameters. F-test results are presented in footnotes as year, plot, and year × plot for each time interval. We did not collect enough data to estimate territory size or breeding success in 2013.

	2012–2014			2012–2019		
Variable	Year	Trt	Year × Trt	Year	Trt	Year × Trt
Density[1]	0.005	0.748	0.023	<0.001	0.27	0.498
Territory size[2]	<0.001	0.005	<0.001	<0.001	0.341	0.741
Proportion of successful territories[3]	0.811	0.393	0.656	0.112	0.182	0.326
Overall productivity[4]	0.547	0.647	0.993	0.976	0.077	0.192
Productivity for successful territories[5]	0.029	0.394	0.429	0.057	0.737	0.053

[1] 2012–2014 F tests, df = 2,9; 1,9; 2,9 (includes 2013); 2012–2019F tests, df = 1,5; 1,5; 1,5.
[2] 2012–2014 F tests, df = 1,96; 1,96; 1,96; 2012–2019F tests, df = 1,88; 1,88; 1,88.
[3] 2012–2014 F tests, df = 1,99; 1,99; 1,99; 2012–2019F tests, df = 1,90; 1,90; 1,90.
[4] 2012–2014 F tests, df = 1,85; 1,85; 1,85; 2012–2019F tests, df = 1,83; 1,83; 1,83.
[5] 2012–2014 F tests, df = 1,56; 1,56; 1,56; 2012–2019F tests, df = 1,57; 1,57; 1,57.

levels in 2019 but the relationship to fire severity suggested other environmental factors were driving this pattern.

Prescribed fire was also intended to reduce risk of crown fire in these woodlands. We found fuel load and canopy base height, two parameters used to predict fire risk, were not significantly affected by treatment in the year after fire, nor six years post-fire. The dominant surface fuel in these woodlands is juniper needles, which under most conditions are not highly flammable and do not carry surface fires well (Twidwell et al., 2009; Andruk et al., 2014). In addition, our simple random sampling design may not adequately detect relationships between fuel loads and fire effects (Twidwell et al., 2009). Despite its common use in wildfire models, canopy base height is known to be difficult to measure and therefore prone to inconsistencies (Scott and Reinhardt, 2001; Cruz et al., 2003; Kramer et al., 2016), complicating interpretation of our results. We therefore consider our results inconclusive as to the effectiveness of prescribed fire at reducing risk of crown fire in these woodlands.

We observed significant annual variation for 11 of 12 fuel and vegetation parameters and treatment effects for 5 of 12 parameters. Annual differences could be due to changes in ecological processes or in response to weather-related patterns, neither of which we explored. For example, the prolonged drought experienced in the area during 2011–2012 seemingly affected oak seedling density on BCNWR (Andruk et al., 2014) and caused substantial canopy loss across the region (Schwantes et al., 2016). Annual effects could also be caused by changes in observers between years. We attempted to minimize any observer measurement bias by having the same person train observers every year and by using mostly quantitative measurements that followed detailed protocols; however, we acknowledge certain measurements such as canopy base height are more prone to subjective errors. Treatment differences that lacked a significant year × treatment interaction reflect innate differences in plot pairs. Those effects were controlled for with the paired-plot BACI design that allowed us to separate the fire effects from the background noise of these annual and plot differences.

We saw varying effects of prescribed fire on golden-cheeked warblers. The significant year × treatment interaction for territory density and size in 2014 but not in 2019 could be viewed as evidence of recovery in 2019 from prescribed fire effects evident in 2014, but this was largely the result of declines in density on control plots in 2019. Golden-cheeked warbler density declined in 2013 and 2014 on treatment plots, while remaining steady on control plots, whereas density slightly increased on treatment plots but declined on control plots in 2019. We do not know the cause for this decline, or if it represents a trend in lower abundance since 2014, but it is worth noting that intensively monitored sites

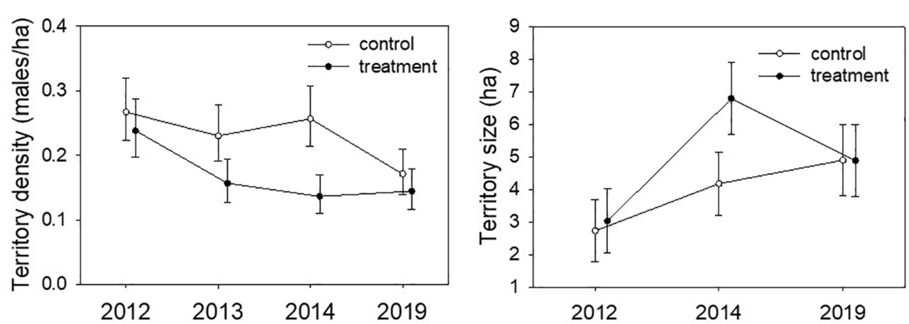

Fig. 2. Mean golden-cheeked warbler territory density and territory size by year and treatment type on Balcones Canyonlands National Wildlife Refuge, Texas. We did not collect enough observations of males in 2013 to estimate territory size.

Table 3
Breeding success parameters of golden-cheeked warblers on six plots within Balcones Canyonlands National Wildlife Refuge, Texas, spring 2012, 2014, and 2019.

		Control						Treatment					
		2012 (n = 31)		2014 (n = 31)		2019 (n = 20)		2012 (n = 28)		2014 (n = 16)		2019 (n = 17)	
Variable	n	Mean	SE	Mean	SE	Mean	SE	Mean	SE	Mean	SE	Mean	SE
Proportion of successful territories	142	0.61	0.09	0.63	0.09	0.85	0.08	0.57	0.10	0.50	0.13	0.64	0.39
Average number of fledglings per territory[1]	127	2.6	0.4	2.3	0.4	3.2	0.5	2.4	0.4	2.2	0.5	1.9	0.4
Mean fledglings per successful territory	90	3.6	0.3	3.2	0.2	3.6	0.3	4.1	0.3	3.2	0.4	3.0	0.3
Total number of fledglings	90	22.7	8.4	20.3	1.9	20.3	4.8	21.7	7.8	8.7	2.3	11.0	4.0

[1] Excludes 15 territories with unknown reproductive output.

Table 4
Descriptive statistics for vegetation and habitat parameters measured around golden-cheeked warbler nests monitored on plots within Balcones Canyonlands National Wildlife Refuge, Texas.

	Control						Treatment					
	2012 (n = 11)		2014 (n = 17)		2019 (n = 19)		2012 (n = 14)		2014 (n = 11)		2019 (n = 14)	
Variable	Mean	SD	Mean	SD	Mean	SD	Mean	SD	Mean	SD	Mean	SD
Juniper basal area (ha^{-1})	20	6	20	10	17	7	20	8	15	7	12	3
Small stem density (ha^{-1})	5042	4628	3390	3054	5151	3376	6182	7559	9812	10,676	2955	2257
Slope (%)	15	8	12	8	8	7	17	12	17	13	10	7

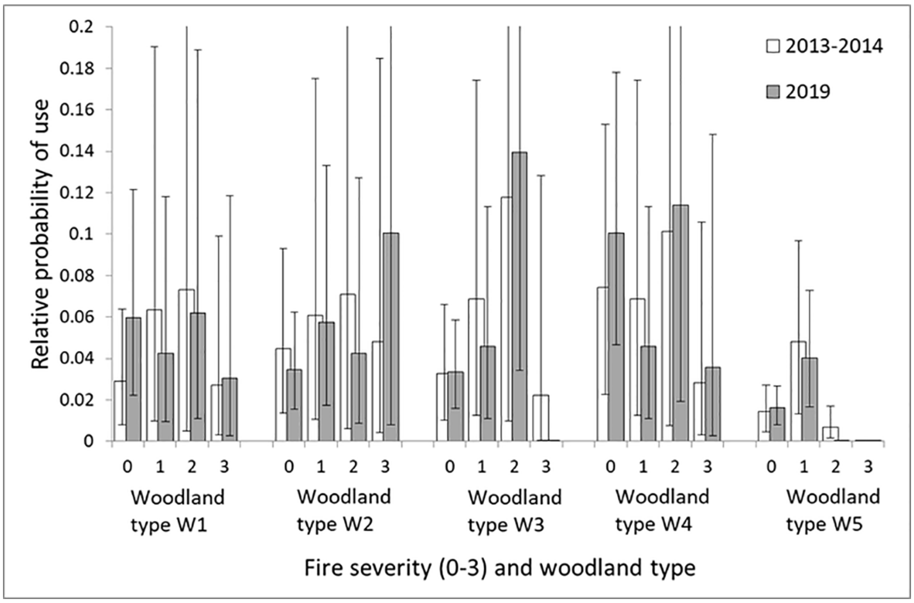

Fig. 3. Relative probability of habitat use as a function of fire severity within woodland types based on observations of male golden-cheeked warblers on three study plots treated with prescribed fire on Balcones Canyonlands National Wildlife Refuge, Texas. The years 2013–2014 represent the short-term, immediate effect, and 2019 represents the intermediate effect. Large confidence intervals in probability of use are the result of variation among individuals. W1-W4 all had > 60% canopy cover but varying canopy and understory composition; W5 had < 60% canopy cover. See Appendix B for full description of woodland types.

elsewhere documented fewer golden-cheeked warbler territories or lower densities since the conclusion of our original study in 2014 (City of Austin and Travis County, 2019; Finn et al., 2019). It is possible that conditions experienced during migration or on the wintering grounds contributed to the lower territory densities we and others observed. We estimated larger territory sizes on control plots across years, whereas territory sizes increased significantly on treatment plots from 2012 to 2014, but then decreased to a similar size as control plots in 2019. Territory size is a proxy for habitat quality, with larger sizes typically indicating lower quality. We surmise the greater change in territory size in 2014 on treatment plots was due to changes in or elimination of appropriate breeding habitat in small patches that somewhat recovered by 2019, or because males roamed more with lower plot-level densities. Given our BACI design, the differences observed in territory density and size between 2012 and 2014 and then between 2012 and 2019 could be interpreted as a recovery in each parameter on treatment plots.

Pairing success of golden-cheeked warblers was high across all years and plots, and productivity was within the range reported on Fort Hood (Peak and Thompson, 2014) and the Balcones Canyonlands Preserve (Reidy et al., 2018). Most parameters of breeding success did not show a significant interaction effect but there was a significant interaction effect for number of young produced by successful territories in 2019. The number of young produced by successful territories was similar in 2012 and 2019 on control plots but there were fewer young produced by successful territories on treatment plots in 2019. Although there was not a significant interaction effect for density or for proportion of successful territories, control plots did support slightly higher densities of golden-cheeked warblers and had a higher proportion of successful territories, which may explain this result.

We documented changes in golden-cheeked warbler habitat use between 2013 and 2014 and 2019 that we consider important for guiding management. Our results suggest that the impact of fire on golden-cheeked warbler habitat preferences was strongly linked to the pre-fire canopy and understory structure. For example, golden-cheeked warblers had a much higher probability of using juniper-dominated woodland (W3) that experienced moderate fire effects, resulting in much of the understory being scorched or killed, than any other fire severity class. In mixed oak-juniper woodlands, the response differed by oak species and understory composition. For example, in red oak-juniper woodlands with a sparse understory (W1), the probability of use was least in areas of severe fire effects in both 2013–2014 and in 2019; whereas in red oak-juniper woodlands with a juniper understory (W2), probability of use was highest in areas of severe fire effects (canopy was scorched) in 2019. In woodlands co-dominated by a live oak or shin oak canopy (W4), unburned and moderately burned areas shared almost equal and relatively high probabilities of use. Overall, golden-cheeked warblers were least likely to use areas of low canopy (W5) under any fire severity, but the probability of use was much higher in areas of low fire effects (little understory scorched).

Based on the cumulative results, low-intensity prescribed fire in mixed juniper-oak woodlands may have slightly negative effects on golden-cheeked warbler density and breeding success in the near-term but also may provide beneficial effects to the habitat structure, and therefore the golden-cheeked warbler, in the long-term if increased oak regeneration results. Golden-cheeked warblers seemed to prefer juniper-dominated woodlands where a substantial amount of understory had been killed compared to those woodlands that experienced less fire effects. We do not know if the same effect would be achieved by manipulating the understory density using mechanical methods, or if fire-induced changes to the understory are more beneficial. We also consider it important to note that areas where substantial canopy was killed did not support golden-cheeked warblers six years post-fire.

5. Conclusions

Prescribed fire achieved some, but not all, fuel and vegetation management goals and had mixed effects on golden-cheeked warblers. Prescribed fire promoted density of non-juniper species, mostly from sprouting, and initially decreased both juniper seedling and sapling densities, thus demonstrating its potential for altering woody species composition in juniper-dominated woodlands. Whereas plots treated with prescribed fire supported lower densities of golden-cheeked warblers in the immediate years post-fire, by six years after the fire, densities were similar across plots; however, this was due to a decline on control plots rather than an increase on treatment plots. Overall treatment plots supported lower numbers of fledglings post-fire. Golden-cheeked warblers made greater use of closed-canopy woodlands that experienced low or moderate fire severity from prescribed fire but generally avoided areas that experienced severe fire effects and canopy mortality, even six years post-fire. This is important because moderate fire severity also resulted in the most desirable changes to the vegetation for reducing the risk of a canopy fire and potentially increasing oaks. We suggest future management decisions strongly weigh the potential benefits from prescribed fire such as increased oak regeneration or decreased juniper density with the chance a fire may exceed prescription and cause canopy mortality. Multiple prescribed fire treatments or a mixture of treatments may be necessary to meet fuel loads and habitat objectives. We also suggest future fire studies specifically examine if oak regeneration is occurring in juniper-dominated woodlands.

CRediT authorship contribution statement

Jennifer L. Reidy: Conceptualization, Methodology, Formal analysis, Investigation, Data curation, Writing - original draft, Writing - review & editing, Visualization, Supervision, Project administration. **Frank R. Thompson:** Methodology, Validation, Formal analysis, Resources, Writing - original draft, Writing - review & editing, Visualization, Project administration, Funding acquisition. **Scott Rowin:** Resources, Writing - review & editing, Project administration. **Carl Schwope:** Conceptualization, Methodology, Resources, Writing - review & editing, Project administration, Funding acquisition. **James M. Mueller:** Validation, Resources, Data curation, Writing - review & editing.

Declaration of Competing Interest

None.

Acknowledgments

We are grateful to H. Becker, J. Best, S. Caird, A. Cronin, H. Hill, C. Hunts, D. Lumpkin, L. Moulton, C. Parra, J. Scalise, M. Wickens, and M. Wilcox for assistance with data collection and J. Stanovick for assistance with data analysis. Funding and additional support was provided by U.S. Fish and Wildlife Service Balcones Canyonlands National Wildlife Refuge (grant number 18-IA-11242311-080), the U.S.D.A. Forest Service Northern Research Station, and the University of Missouri-Columbia. The findings and conclusions in this article are those of the authors and do not necessarily represent the views of the U.S. Fish and Wildlife Service.

Appendix A. . Frequency (and percent of total) of fire severity scores categorized by woodland type and fire severity for junipers and hardwoods measured at plots treated with fire on Balcones Canyonlands National Wildlife Refuge, Texas, in February 2013 (Reidy et al., 2016a).

		Juniper						Hardwood				
Fire Severity	Description	W1[a]	W2	W3	W4	W5	Description	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 > 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)	<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)	>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 Appendix 2.

Appendix B. . Description of woodland types on plots treated with prescribed fire on Balcones Canyonlands National Wildlife Refuge, Texas.

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

References

Andruk, C.M., Schwope, C., Fowler, N.L., 2014. The joint effects of fire and herbivory on hardwood regeneration in central Texas woodlands. For. Ecol. Manage., vol. 334, pp. 193–200.

Andruk, C.M., Fowler, N.L., 2015. Conflicting short and long-term management goals: fire effects in endangered golden-cheeked warbler (*Setophaga chrysoparia*) habitat. USDA Forest Service RMRS-P-73.

Baccus, J.T., Tollé, M.E., Cornelius, J.D., 2007. Response of golden-cheeked warblers (*Dendroica chrysoparia*) to wildfires at Fort Hood. Texas Ornithological Society No, Texas, p. 7.

Brose, P.H., Dey, D.C., Phillips, R.J., Waldrop, T.A., 2013. A meta-analysis of the fire-oak hypothesis: does prescribed burning promote oak reproduction in eastern North America? Forest Sci. 59, 322–334.

Bunting, S.C., 1986. Use of prescribed burning in juniper and pinyon-juniper woodlands. In: Proceedings – Pinyon-juniper conference, Reno, NV 1986.

City of Austin and Travis County. 2019. 2019 Annual Report: Golden-cheeked Warbler (*Setophaga chrysoparia*) monitoring program. Balcones Canyonlands Preserve, Austin, TX. <https://www.traviscountytx.gov/images/tnr/Docs/bccp/2019/appendix-f.pdf> Accessed 14 July 2020.

Cooper, A.B., Millspaugh, J.J., 1999. The application of discrete choice models to wildlife resource selection studies. Ecology 80, 566–575.

Cruz, M.G., Alexander, M.E., Wakimoto, R.H., 2003. Assessing canopy fuel stratum characteristics in crown fire prone fuel types of western North America. Int. J. Wildland Fire 12, 39–50.

Diamond, D.D., Rowell, G.A., Keddy-Hector, D.P., 1995. Conservation of Ashe juniper (*Juniperus ashei Buchholz*) woodlands of the Central Texas Hill Country. Nat. Areas Assoc. 15, 189–197.

Diamond, D.D., True, C.D., He, K., 1997. Regional priorities for conservation of rare species in Texas. Southwest. Natural. 42, 400–408.

Fernandes, P.M., Botelho, H.S., 2003. A review of prescribed burning effectiveness in fire hazard reduction. Int. J. Wildland Fire 12, 117–128.

Finn, D., Raginski, N., Long, A.M., Carrasco, S.M., 2019. Golden-cheeked warbler and black-capped vireo monitoring on Joint Base San Antonio-Camp Bullis, Texas: 2019 field season report.

Key, C.H., Benson, N.C., 2006. Landscape Assessment (LA). In: Lutes, D.C., Keane, R.E., Caratti, J.F., Key, C.H., Benson, N.C., Sutherland, S., Gangi, L.J. (Eds.) FIREMON: Fire effects monitoring and inventory system. USDA Gen. Tech. Rep. RMRS-GTR-164-CD. Fort Collins, CO: U.S. Department of Agriculture, Forest Service, Rocky Mountain Research Station. p. LA-1–55.

Kramer, H.A., Collins, B.M., Lake, F.K., Jakubowski, M.K., Stephens, S.L., Kelly, M., 2016. Estimating ladder fuels: a new approach combining field photography with LiDAR. Remote Sens. 8, 766.

Ladd, C., Gass, L., 1999. Golden-cheeked Warbler (*Dendroica chrysoparia*). The birds of North America, Number, p. 420.

Liang, K.-Y., Zeger, S.L., 1986. Longitudinal data analysis using generalized linear models. Biometrika 73, 13–22.

Lutes, D.C., Keane, R.E., Caratti, J.F., Key, C.H., Benson, N.C., Sutherland, S., Gangi, L.J., 2006. FIREMON: Fire effects monitoring and inventory system. U.S. Department of Agriculture, Forest Service, Rocky Mountain Research Station. Gen. Tech. Rep. RMRS-GTR-164-CD. Fort Collins, CO.

Peak, R.G., Thompson III, F.R., 2013. Amount and type of forest cover and edge are important predictors of golden-cheeked warbler density. Condor 115, 659–668.

Peak, R.G., Thompson III, F.R., 2014. Seasonal productivity and nest survival of Golden-cheeked Warblers vary with forest type and edge density. Condor 116, 546–559.

Reemts, C.M., Hansen, L.L., 2008. Slow recolonization of burned oak-juniper woodlands by Ashe juniper (*Juniperus ashei*): Ten years of succession after crown fire. Forest Ecol. Manage., 255:1057–1066.

Reidy, J.L., O'Donnell, L., Thompson III, F.R., 2015. Evaluation of a reproductive index for estimating songbird productivity: case study of the Golden-cheeked Warbler. Wildlife Soc. Bull. 39, 721–731.

Reidy, J.L., Thompson III, F.R., Peak, R.G., 2009. Factors affecting Golden-cheeked Warbler nest survival in urban and rural landscapes. J. Wildlife Manage. 73, 407–413.

Reidy, J.L., Thompson III, F.R., Schwope, C., Rowin, S., Mueller, J.M., 2016a. Effects of prescribed fire on fuels, vegetation, and golden-cheeked warbler (*Setophaga chrysoparia*) demographics in Texas juniper-oak woodlands. Forest Ecol. Manage. 376, 96–106.

Reidy, J.L., Thompson III, F.R., Connette, G.M., O'Donnell, L., 2018. Demographic rates of golden-cheeked warblers in an urbanizing woodland preserve. Condor 120, 249–264.

Reidy, J.L., Thompson III, F.R., O'Donnell, L., 2017. Density and nest survival of golden–cheeked warblers: spatial scale matters. J. Wildlife Manage. 81, 678–689.

Reidy, J.L., Thompson III, F.R., Amundson, C., O'Donnell, L., 2016b. Landscape and local effects on occupancy and densities of an endangered wood-warbler in an urbanizing landscape. Landsc. Ecol., vol. 31:365–382.

Russell, F.L., Fowler, N.L., 2002. Failure of adult recruitment in *Quercus buckleyi* populations on the eastern Edwards Plateau, Texas. Am. Midland Natural. 148, 201–217.

Ryan, K.C., Knapp, E.E., Varner, J.M., 2013. Prescribed fire in North American forests and woodlands: history, current practice, and challenges. Front. Ecol. Environ. 11, E15–E24.

Schwantes, A.M., Swenson, J.J., Jackson, R.B., 2016. Quantifying drought-induced tree mortality in the open canopy woodlands of central Texas. Remote Sens. Environ. 181, 54–64.

Scott, J.H., Reinhardt, E.D., 2001. Assessing crown fire potential by linking models of surface and crown fire behavior. USDA Forest Service Rocky Mountain Research Station RMRS-RP-29.

Shaffer, T.L., 2004. A unified approach to analyzing nest success. Auk 121, 526–540.

Thomas, J.A., White, J.D., Murray, D.B., 2016. Tree species influence woodland canopy characteristics and crown fire potential. Forest Ecol. Manage. 362, 169–176.

Trumbo, E.M., 2019. Post-fledging ecology of an endangered species: the golden-cheeked warbler. University of Illinois, Urbana, Thesis.

Twidwell, D., Fuhlendorf, S.D., Engle, D.M., Taylor Jr., C.A., 2009. Surface fuel sampling strategies: linking fuel measurements and fire effects. Rangeland Ecol. Manage. 62, 223–229.

U. S. Fish and Wildlife Service (USFWS), 1990. Endangered and threatened wildlife and plants; emergency rule to list the golden-cheeked warbler as endangered. Federal Register 55, 53153–53160.

White, J.D., Thomas, J., Yao, J., Murray, D., 2010. The Balcones Canyonlands National Wildlife Refuge fire risk and management: characterization of woodland fuels and possible risk to Golden-cheeked Warbler habitat. Final Report for the USGS Cooperative Agreement 02CRAG0002, 110 p.