Management and Conservation

Effects of Prescribed Burning on Avian Nest Survival in the Southern Great Plains

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ABSTRACT Shrubs, such as mesquite (Prosopis spp.) and cholla (Opuntia spp.), now dominate fire-suppressed grasslands in southwestern North America. Responses of birds to prescribed burning of the shortgrass prairie in this region are poorly understood. We examined daily survival rates of mourning dove (Zenaida macroura) and lark sparrow (Chondestes grammacus) nests in an experimental landscape (4,811 ha) of spatially replicated, inter-annual fire frequencies (burning every 2 yr, 4 yr, or 10 yr) near Amarillo, Texas. Herbaceous habitat structure was most developed in infrequently burned plots, but shrub densities were less variable among the burn treatments. We modeled daily nest survival (DSR) against burn frequency, shrub density at nest sites, and nest stage (incubation or nestling). Daily survival of mourning dove nests was not well-related to any measured covariate, but lark sparrow DSR was negatively related to the duration of inter-annual burn frequency. In semiarid grasslands heavily inundated with shrubs, prescribed burning may positively influence the nest success of some bird species. © 2012 The Wildlife Society.

KEY WORDS grassland birds, great plains, lark sparrow, mourning dove, nest survival, prescribed burning, shortgrass prairie, woody encroachment.

As found across the Great Plains, grassland bird species that occur in the southwestern United States, such as the northern bobwhite (Colinus virginianus), Cassin’s sparrow (Pecucaea cassini), lark sparrow (Chondestes grammacus), and western meadowlark (Sturnella neglecta), have exhibited range-wide population declines throughout the last century (Peterjohn and Sauer 1999, Butcher and Niven 2007). Habitat loss is likely the most influential ecological driver of these declines (Herkert et al. 1995, Vickery and Herkert 2001), but degradation of remnant grasslands (Brawn et al. 2001, With et al. 2008) may also limit grassland bird populations. Fire suppression and overgrazing have driven habitat conversion of semiarid grasslands in the southwestern United States, resulting in the encroachment and dominance of woody species, such as creosotebush (Larrea tridentata), cholla (Opuntia spp.), mesquite (Prosopis spp.), and salt cedar (Tamarix spp.; Archer 1989, Bahre 1991, Bahre and Shelton 1993, Bahre 1995, Van Aukcn 2000). Honey mesquite (P. glandulosa), the most prominently encroaching shrub species, is now estimated to cover more than 38 million ha of shortgrass prairie (Archer 1989, Van Aukcn 2000, Brockway et al. 2002).

We have limited understanding of how grassland birds respond to shrub encroachment and management used to reduce shrub density in the southwestern United States (Ford and McPherson 1996, Brawn et al. 2001, Bock and Block 2005). This is not the case for tallgrass prairie, where the biological consequences of fire suppression, subsequent woody encroachment, and avian responses to grassland restoration techniques have been well-studied (Zimmerman 1992, Herkert 1994a,b; Reinking 2005; Powell 2006; Rahmig et al. 2009). Depredation and brood parasitism of grassland bird nests are positively related to nest proximity to woody vegetation in tallgrass prairie (Johnson and Temple 1990, Winter et al. 2000, Jensen and Finck 2004, Jensen and Cully 2005), thus prescribed burning that reduces overall shrub density should favor grassland-nesting species. Following fire, vegetation composition in shortgrass prairie may also shift to favor greater abundances of grassland birds, suggesting that the reintroduction of fire as a management technique may be positive for these species (Bock and Bock 1992a,b; Kirkpatrick et al. 2002). However, prescribed burning in this region is limited by land manager uncertainties regarding seasonal and historic fire frequencies, nutrient turnover rates, slow re-growth of vegetation following fire events, and resistance to burning by private landowners (Wright and Bailey 1982, Brockway et al. 2002). Thus, such management is rarely attempted and, as a consequence, responses of grassland-dependent wildlife to prescribed burning in grasslands of the southwestern Great Plains are relatively unknown (Ford and McPherson 1996, Brawn et al. 2001, Bock and Block 2005).

Past research regarding fire, woody encroachment, and birds in shortgrass prairie has lacked the use of spatially

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replicated experimental units (Bock and Bock 1988) or has been conducted in relation to single wildfire events (Bock and Bock 1992a, Reynolds and Krausman 1998, Kirkpatrick et al. 2002). Also lacking have been comprehensive examinations of avian demographic consequences of fire suppression and woody encroachment in shortgrass prairie (Bock and Block 2005). We examined daily survival of bird nests in relation to experimental burning of a shortgrass prairie heavily encroached by honey mesquite and cholla in the southern Great Plains. Historically, the avian assemblage in the region was likely dominated by grassland species common to the southwestern United States, such as Cassin’s sparrows and western meadowlarks (Seyffert 2001); however, the current avian community is more characteristic of shrub-steppe vegetation (Seyffert 2001, Long 2010). We, therefore, limited our investigation to 2 species that are facultative ground nesters in grasslands (Drobney et al. 1998, Martin and Parrish 2000): mourning doves (Zenaida macroura) and lark sparrows. These species provided relatively large sample sizes of nests (n = 56 and 318, respectively) and were thought to best represent the regional grassland bird community on our site. We predicted vegetation structure, including shrub density, to generally diminish in response to prescribed burning; and, based on patterns of nest success and woody vegetation observed elsewhere (Johnson and Temple 1990, With 1994, Klug et al. 2010), daily nest survival (DSR) was predicted to be highest in areas that were more frequently burned.

STUDY AREA

We conducted our research from May to July of 2008 and 2009 at the 4,811-ha Cross Bar Cooperative Management Area (CCMA) located approximately 20 km northwest of Amarillo, Texas (35°N, 101°W). The CCMA was historically semiarid shortgrass prairie composed of graminoid species, such as blue grama (Bouteloua gracilis) and buffalo grass (Buchloe dactyloides) and forb species, such as small soapweed yucca (Yucca glauca; Tanner 2004). In the past, honey mesquite and cholla were found intermittently at the CCMA; however, these 2 woody species now dominate the landscape (Tanner 2004). Average monthly temperatures and monthly precipitation from May to July, 1971–2000, ranged from 18.0° C to 26.0° C and 6.4–8.3 cm (National Weather Service 2009). Average monthly temperatures and precipitation ranged from 19.1° C to 25.2° C and 5.3–12.6 cm during data collection in 2008 (National Weather Service 2009). During the 2009 field season, average monthly temperatures and precipitation ranged from 17.7° C to 25.6° C and 1.1–9.6 cm (National Weather Service 2009).

The CCMA was originally purchased for use as a helium storage facility by the United States Bureau of Mines in 1931 (Tanner 2004). While being used for underground storage, the land was heavily grazed for >70 years (Tanner 2004). Grazing was discontinued on a majority of the property in 1993 and was completely eliminated in 1998 after the land was acquired by the United States Bureau of Land Management (BLM; Tanner 2004). Similar to other southwestern grasslands, years of overgrazing and fire suppression resulted in high levels of woody encroachment (Tanner 2004). In 2002, the BLM, in cooperation with West Texas A&M University (WTAMU), established and burned 9 experimental units (hereafter plots; 120–220 ha) on the CCMA to examine the effect of prescribed burning on shrub density. Each plot was assigned a burn treatment (i.e., fire every 2 yr, 4 yr, or 10 yr) in a stratified, random design, with 3 replicates of each burn treatment. Most prescribed burns on the CCMA were conducted in January and February.

METHODS

Habitat Measures

We measured habitat structure to 1) characterize habitat variation among the burn units, and 2) provide covariates that might explain variation in nest success. We modified our habitat measurements from the Breeding Biology Research and Monitoring (BBIRD) Grassland Field Protocol (Martin et al. 1997). We made measurements across each plot (22 Jun–31 Jul 2008; 22 Jun–31 Jul 2009) from the center of 13, 5-m radius circles located within 100-m radius circles. These 100-m radius circles were initially established systematically across each plot to examine the effects of prescribed burning on avian abundance and community structure (see Long 2010 for specific methodology and estimates of abundance, species richness, and diversity obtained using point-count sampling). The number of circles established per plot (range: 4–11) was determined by the total area of the plot, resulting in 30 circles located within plots burned every 2 years, 34 circles located within plots burned every 4 years, and 44 circles located within plots burned every 10 years. The vegetation sampling points were located at the center of each circle and at 30 m, 60 m, and 90 m from the center of the circle in each cardinal direction. We also made the same habitat measurements at each nest site upon the completion of each nesting attempt, where sampling points were centered on nest locations.

We measured herbaceous plant height (cm) from the center of each 5-m radius circle at 1 m, 3 m, and 5 m in each cardinal direction using a tape measure. We made a modification to the Robel method (Robel et al. 1970), otherwise suggested by BBIRD protocol, because of our limited resolution in estimating visual obstruction in the short-statured vegetation of recently burned plots. We used median height at each vegetation sampling point to reduce the effect of extreme values in summarizing vertical vegetation structure. We visually estimated percent horizontal coverage of grasses, forbs, and bare ground within 4 quadrants of each 5-m vegetation sampling point. We then used mean percent coverages to summarize horizontal cover for each cover type.

We measured the density of shrubs to examine potential influences of shrub structure on nest success. We used the point-quarter method (Cottam and Curtis 1956) to estimate densities (stems/ha) of shrub species categories across the plots. We used a rangefinder to measure the distance from vegetation sampling point centers to the closest shrub, per species category, in each quadrant per habitat sampling point. Shrub species categories included the following: large...
(≥1.5 m) and small (<1.5 m) mesquite and cholla; we further categorized mesquite size classes as live and top-killed growth forms (i.e., basal re-growth following fire). We limited our measurements of shrub density to these species because of their dominance on the landscape of the CCMA. Because of the varied sizes and growth forms of these shrub species, we estimated densities of several shrub categories, including all shrubs (across species and size classes), large shrubs (both species), small shrubs (both species), all mesquite (both size classes and growth forms), and all cholla (both size classes).

We recorded additional habitat data specific to nest sites, including nesting plant species, nest location (on-ground or in-shrub), nest height (cm from ground to the lowest portion of the nest), and distance from nearest grazed land.

**Nest Monitoring**

In 2008 (18 May–31 Jul) and 2009 (17 May–31 Jul), we searched for nests on each plot every 3–4 days for approximately 3–4 hours/plot. We used a combination of visual and auditory cues to locate each nest (Martin and Geupel 1993). Because of coverage constraints related to plot size, we rotated the location from which searching began to ensure uniform search effort within plots. We marked most nest locations by placing flagging approximately 5–10 m from each nest and relocated these nests using Global Positioning System (GPS) units. We did not mark some nest locations with flagging, but rather marked them using only GPS coordinates for an ancillary study on investigator effects on nest success. However, monitoring technique did not affect daily nest mortality (Jacobson et al. 2011). Field personnel trained in proper monitoring techniques monitored most nests every 3–4 days (Martin and Geupel 1993) and rarely every 8–13 days for nests included in Jacobson et al. (2011). We recorded nest contents until nest completion.

We determined nest fate using evidence at the nest site. We considered nests unsuccessful if found empty before estimated fledging dates or if abandoned (i.e., >3 visits to nests with eggs where incubation was not evident). We considered nests found empty on monitoring visits proximate to predicted fledging dates successful if fledglings were observed around the nest, if fecal sacs were present, or if parental behavior indicated the presence of unseen fledglings (Martin and Geupel 1993).

**Statistical Analysis**

We averaged herbaceous plant height, percent ground cover (grasses, forbs, and bare ground), and shrub density (all categories) across point-count circles per plot replicate. We then used 2-way analyses of variance (ANOVA) and Fisher’s least significant difference follow-up tests in Proc GLM in SAS (SAS 9.2; SAS Institute, Cary, NC) to compare habitat variable means among the burn treatments and to test for burn × year interactions. When the interaction between burn treatment and year was not significant, we used 1-way ANOVA with the single burn effect to assess variation in habitat structure among the burn treatments. When burn × year was significant (P < 0.10), 1-way ANOVAs were used to test the burn effect separately for 2008 and 2009. We considered differences statistically significant when P < 0.05 and marginally significant (i.e., potentially biologically significant) when 0.05 < P < 0.10, as alpha is arbitrary (Johnson 1999).

We used the logistic exposure method of Shaffer (2004), using Proc GENMOD in SAS, to model DSR in relation to environmental covariates. For overall period survival estimates, from incubation through fledging of young, we assumed 29-day and 23-day nest periods for mourning doves and lark sparrows, respectively (Baicich and Harrison 1997). We used an information–theoretic model selection procedure to determine the predictive ability for DSR of inter-annual burn frequency and the density of all shrub species around nests. We modeled inter-annual burn frequency as a categorical covariate. We included the density of all shrubs at nest sites in models to generally represent woody encroachment. Other shrub species, and their growth forms, were correlated with the density of all shrubs and thus not included in candidate models. Additionally, we included developmental stage of offspring (nest stage; i.e., incubation or nestling) to models with burn treatment or shrub density to test for the improvement of this covariate upon model fit. We did not model nest stage separately from burn treatment or shrub density because we were interested in testing the particular effect of the latter variables on DSR, not exploring best predictors of DSR. Only those variables that were not correlated (Pearson’s correlation, P < 0.05) among nests of particular species were included in the same model. We considered other covariates for DSR (e.g., year, date of season, and nest height) but these potential covariates were correlated with both burn frequency and shrub density and thus not included in candidate model sets. The candidate sets of models per species also included a constant (null) model of DSR from which to compare support for models containing covariates. All other models containing covariates contained at least the covariates for burn frequency or shrub density.

We ranked model fit among the candidate set per species using Akaike’s Information Criterion (AICi; Burnham and Anderson 2002) corrected for small sample size (AICc; Sugiura 1978). We then used differences in AIC, (ΔAICi) and Akaike weights (wi) to determine model support (Burnham and Anderson 2002). We considered models with ΔAICi ≤ 2 competitive models. We calculated odds ratios (for categorical burn frequency = ei/bi for burn intervals i and j) and covariate coefficients (for continuous covariates), and their 95% confidence intervals, to determine the effect of each covariate on DSR included best candidate models. Where we found multiple plausible models (ΔAICi, ≤ 2.0), we calculated model-averaged estimates of covariate coefficients across these top models. We estimated period survival (Shaffer and Thompson 2007) using model-averaged slope estimates 1) overall per species, using mean levels of continuous covariates and observed proportions of observations per categorical burn variable (if included in top models), and 2) per burn frequency, where burn was determined to have a significant effect on DSR, while
RESULTS

Habitat Measures
Mean herbaceous plant height, mean percent grass cover, and percent bare ground were the only habitat variables that showed statistically significant interactions between burn treatment and year (Table 1; i.e., the differences among the burn treatments depended upon the year of study). In 2008, mean herbaceous plant height was marginally different across the 3 burn treatments, with mean herbaceous plant height higher in plots burned every 10 years than in plots burned every 2 and 4 years (Table 1). In 2009, mean herbaceous plant height was significantly higher in plots burned every 4 and 10 years than in plots burned every 2 years (Table 1). Mean percent grass cover was significantly greater in plots burned every 4 years and 10 years than in plots burned every 2 years (Table 1). Mean percent bare ground was significantly greater in plots burned every 10 years than in plots burned every 2 and 4 years (Table 1). In 2009, mean percent bare ground was significantly greater in plots burned every 2 and 4 years than in plots burned every 10 years (Table 1). We found no additional differences in any other shrub density category among burn frequencies (Table 1).

Nest Survival
Nest sample sizes per burn treatment for mourning doves totaled 22 nests in 2-year burns, 20 nests in 4-year burns, and 14 nests in 10-year burns. Lark sparrow nests were distributed among burn treatments as follows: 150 nests in 2-year burns, 102 nests in 4-year burns, and 66 nests in 10-year burns. Ten percent of lark sparrow nests were parasitized by brown-headed cowbirds (Molothrus ater), with only 1 nest containing >1 cowbird egg.

Effects of overall shrub density and nest stage were included in best fit models (ΔAICc ≤ 2.0) of mourning dove DSR (Table 2). Both of these top models were reasonably well fit (c = 1.15–1.16). The model-averaged estimate of overall DSR from these best models was 0.921 (95% CI: 0.813–0.969) yielding an overall period survival (incubation to fledging) estimate of 9.3% (95% CI: 0.2–40.5%). Confidence limits of coefficients for both shrub density (model-averaged β < 0.001, 95% CI: −0.0001 to 0.0002) and nest stage (β < 0.001, 95% CI: −0.242 to 0.322) included 0, indicating no substantial effect of these covariates. A single model including the covariates burn and all-shrub density was superior (ΔAICc ≤ 2.0, wj = 0.734) among the candidate set in explaining variation in DSR of lark sparrows (Table 2; model fit was reasonable: c = 1.26). The estimate of overall DSR from this top model was 0.890 (95% CI: 0.799–0.943) yielding an overall period survival estimate of 6.9% holding any other covariates in those models at their mean values. An index of overdispersion ($\hat{\imath}$) was calculated by dividing model deviance by model degrees of freedom.

Table 1. Comparison of mean habitat variables (mean ± SE) across 3 prescribed burn treatments (fire every 2 yr, 4 yr, and 10 yr) at the Cross Bar Cooperative Management located near Amarillo, Texas, during 2008–2009. Means are listed separately per year where interactions between burn treatment and year were statistically significant.

<table>
<thead>
<tr>
<th>Habitat variable*</th>
<th>2</th>
<th>4</th>
<th>10</th>
</tr>
</thead>
</table>
| **Herbaceous plant height**
  2008** | 11.0A (1.00) | 11.3B (2.84) | 18.7B (2.73) |
  2009* | 9.2A (0.73) | 32.7A (4.81) | 34.3B (6.57) |
| **Percent grass cover**
  2008* | 61.6A (0.19) | 76.8A (0.80) | 77.4A (3.25) |
  2009* | 64.8A (1.48) | 63.0A (3.13) | 74.4B (1.62) |
| **Percent forb cover**
  2008* | 10.5 (1.60) | 12.3 (0.63) | 16.2 (0.74) |
| **Percent bare ground**
  2008* | 35.2A (1.48) | 37.0A (1.67) | 25.6A (0.52) |
  2009* | 38.3A (0.21) | 22.9A (0.84) | 22.1B (3.35) |
| **All shrub density**
  2008** | 683.3 (123.90) | 532.4 (105.13) | 627.8 (159.87) |
| **Large shrub density**
  2008** | 95.4AB (9.76) | 65.7A (9.86) | 118.2B (21.62) |
| **Small shrub density**
  2008** | 490.6 (87.09) | 385.1 (71.79) | 355.2 (63.02) |
| **Mesquite density**
  2008** | 313.8 (31.73) | 256.0 (28.16) | 316.0 (66.11) |
| **Large mesquite density**
  2008** | 8.0 (1.98) | 8.5 (3.68) | 17.0 (4.58) |
| **Small mesquite density**
  2008** | 43.8 (6.31) | 52.3 (4.79) | 60.0 (11.14) |
| **Large top-killed mesquite density**
  2008** | 48.8A (6.26) | 29.8A (4.79) | 57.4B (11.89) |
| **Small top-killed mesquite density**
  2008** | 91.3A (14.75) | 86.3 (16.27) | 62.7 (5.80) |
| **Cholla density**
  2008** | 138.2 (31.83) | 100.7 (36.89) | 111.5 (26.02) |
| **Large cholla density**
  2008** | 18.5 (5.83) | 14.3 (1.65) | 21.0 (2.85) |
| **Small cholla density**
  2008** | 93.8 (21.92) | 81.0 (31.36) | 73.3 (12.29) |

* Significantly different means (\(\hat{\imath}\)) are indicated with different letters when overall P-values indicate a statistically significant difference (\(P < 0.05\)) across the burn treatments. Marginally significant means (\(\hat{\imath}\)) are also indicated with different letters when \(0.05 \leq P \leq 0.10\).
Table 2. Models of daily nest survival for mourning doves and lark sparrows at the Cross Bar Cooperative Management Area, located near Amarillo, Texas, 2008–2009. Shown are the numbers of parameters per model (K), Akaike’s Information Criterion, corrected for small sample size (AICc), differences in AICc values (ΔAICc), and model weights (w). Constant indicates a model without explanatory variables. Explanatory variables in candidate models are categorical burn frequency of habitat, density of all shrubs (stems/ha) at nest sites, and nest stage (developmental stage of offspring: eggs or nestlings).

<table>
<thead>
<tr>
<th>Common name and candidate models</th>
<th>K</th>
<th>AICc</th>
<th>ΔAICc</th>
<th>w</th>
</tr>
</thead>
<tbody>
<tr>
<td>Mourning Dove</td>
<td></td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td>All shrub density</td>
<td>2</td>
<td>172.82</td>
<td>0.00</td>
<td>0.710</td>
</tr>
<tr>
<td>All shrub density + Nest stage</td>
<td>3</td>
<td>174.70</td>
<td>1.88</td>
<td>0.277</td>
</tr>
<tr>
<td>Constant</td>
<td>1</td>
<td>181.44</td>
<td>8.63</td>
<td>0.009</td>
</tr>
<tr>
<td>Burn</td>
<td>3</td>
<td>183.81</td>
<td>11.00</td>
<td>0.003</td>
</tr>
<tr>
<td>Burn + Nest stage</td>
<td>4</td>
<td>185.22</td>
<td>12.40</td>
<td>0.001</td>
</tr>
<tr>
<td>Lark Sparrow</td>
<td></td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td>Burn + All shrub density</td>
<td>4</td>
<td>988.36</td>
<td>0.00</td>
<td>0.734</td>
</tr>
<tr>
<td>All shrub density</td>
<td>2</td>
<td>990.39</td>
<td>2.03</td>
<td>0.266</td>
</tr>
<tr>
<td>Burn + Nest stage</td>
<td>4</td>
<td>1033.72</td>
<td>45.36</td>
<td>&lt;0.001</td>
</tr>
<tr>
<td>Burn</td>
<td>3</td>
<td>1042.56</td>
<td>54.21</td>
<td>&lt;0.001</td>
</tr>
<tr>
<td>Constant</td>
<td>1</td>
<td>1044.73</td>
<td>56.38</td>
<td>&lt;0.001</td>
</tr>
</tbody>
</table>

*The candidate set for lark sparrow differs from that of mourning dove because of differences in correlations between prospective covariates (and thus allowable, tandem covariates) for each bird species.

(95% CI: 0.6–26.1%). The model indicated that lark sparrow DSR was positively related to burn frequency. Odds ratios indicated that daily survival was 1.55 times (95% CI: 1.10–2.18) more likely in 2-year than in 10-year burn plots and 1.40 times (95% CI: 0.97–2.01) more likely in 4-year than in 10-year burn plots. The odds ratio between 2-year and 4-year burn treatments was 1.11 (95% CI: 0.83–1.49), indicating a smaller difference in lark sparrow DSR between these burn treatments. Period survival estimates of lark sparrow nests among the 3 burn frequencies were 9.0% (95% CI: 0.7–31.5%) in 2-year burns, 7.0% (95% CI: 0.4–28.6%) in 4-year burns, and 2.7% (95% CI: 0.8–5.6%) in 10-year burns. We did not find a substantial relationship between DSR and shrub density at lark sparrow nest sites (0 < β > −0.001, 95% CI: −0.0001 to 0.0001).

**DISCUSSION**

As predicted, lark sparrow DSR was lower in less frequently burned plots. However, we are not clear if this pattern was due to predator responses to variation in shrub density among the burn treatments. Because shrub density was similar in plots burned every 2 years and 10 years, greater lark sparrow DSR in frequently burned (biennial and quaternnial) plots may instead reflect reductions in herbaceous cover for predators in those areas (e.g., snakes; Cavitt 2000). Also, direct mortality of predators is possibly more common in frequently burned plots. For our other species of interest, the mourning dove, burn treatment did not explain variation in DSR. The limited differences in shrub density across the burn treatments may have contributed to this pattern. Our ability to detect a significant pattern in DSR across the burn treatments for mourning doves may have also been limited by nest sample sizes. Otherwise, we are unclear why burn treatment explained variation in lark sparrow—but not mourning dove—nest survival.

We found few patterns between DSR and shrub density at nest sites. However, such patterns might exist in landscapes exhibiting more pronounced variation in shrub density. Nest predation tends to be greater near woody vegetation in tallgrass prairie (Johnson and Temple 1990, Winter et al. 2000, Klug et al. 2010) and short-grass prairies (With 1994). Shrubland bird species, such as the greater roadrunner (*Geococcyx californianus*; Hughes 1996) and northern mockingbird (*Mimus polyglottos*; Derrickson and Breitwisch 1992) are potential nest predators in our study region. Small mammals also depredate bird nests (Nour et al. 1993, With 1994) and recent research conducted at our study site (R. Matlack, WTAMU, unpublished data) indicated that small mammal abundance and shrub density are positively correlated. Nest predation and snake occurrence coincided with elevated shrub cover in tallgrass prairie (Klug et al. 2010). In semiarid grasslands, Mendelsson and Jennings (1992) similarly found that the abundance of some snakes (i.e., potential nest predators) was correlated with shrub density. Perhaps rodents, in addition to being predators, might also attract snakes near shrubs.

Similar to other studies conducted in shortgrass prairie (Howard et al. 2001, Skagen et al. 2005, Yackel Adams et al. 2007), nest period survival estimates at our study site were low (<20%). Additionally, estimated nest success rates for 6 other species studied on the site were estimated to be less than 10% (Long 2010). If low nest success is a general condition for grassland birds in the region, this is of special concern as many species of this disturbance-dependent ecosystem have experienced significant widespread population declines in the recent past (Samson and Knopf, 1994, Peterjohn and Sauer 1999, Askins et al. 2007, Butler and Niven 2007). Low nest success facilitated by changes in land use patterns, including increased shrub density, may be driving these declines. Nest monitoring by biologists may also affect nest success if observers leave visual or olfactory cues for predators (Westmoreland and Best 1985, Whelan et al. 1994). We discount this possibility for our study. In 2009, we conducted an ancillary investigation of observer effects on nest success on our site and found that nest survival rates were not significantly affected by the presence of marker flags or varying visitation frequency by human observers (Jacobson et al. 2011). This suggests that some environmental—rather than methodological—feature is responsible for low nest success at our study site.

After 7 years of prescribed burning at the CCMA, herbaceous cover responded negatively to prescribed burning though prescribed burning had few, clear effects on the density of particular shrub species. Increased shrub density in semiarid grasslands is globally widespread (Adamoli et al. 1990, Burrows et al. 1990, Archer et al. 1995, Moreira 2000, Roques et al. 2001) and many ecological (Allen 1995, Samson et al. 2004, Brown et al. 2008) and social limitations (Samson and Knopf 2001, Yoder et al. 2003, Samson et al. 2004) restrict the successful restoration and rehabilitation of native vegetation in grassland ecosystems. Previous research in semiarid grasslands has shown that prescribed burning can
reduce the density of shrubs (Cable 1967, Britton and Wright 1971, Heirman and Wright 1973, Williams et al. 1999, Roques et al. 2001) and increase overall grass production (Heirman and Wright 1973, Medina and Silva 1990). However, management to successfully reduce shrub density depends upon several factors including extent of shrub invasion, fire frequency (Humphrey 1958, Heirman and Wright 1973, Williams et al. 1999, Roques et al. 2001), herbaceous fuel load (i.e., fire intensity; Cable 1967, Britton and Wright 1971, Heirman and Wright 1973, Wright et al. 1976), and shrub age (i.e., stem diameter; Glandening and Paulson, 1955, Dwyer and Pieper 1967, Wright et al. 1976). Substantial effects of prescribed burning on shrub density may be delayed in grasslands with advanced levels of shrub encroachment (Allen 1995, Van Auken 2000), conditions found consistently across shortgrass prairies of the southwestern Great Plains.

MANAGEMENT IMPLICATIONS

Intact, shortgrass prairie landscapes of the southwestern Great Plains offer tremendous conservation value for grassland wildlife (Samson et al. 2004). Prescribed burning does appear to have a positive influence on the nesting success of at least some bird species in shrub-encroached, shortgrass prairie. We expect nest success of other grassland birds in the southern Great Plains to respond positively to prescribed burns of ungrazed habitat during the dormant season. However, the reduction in herbaceous cover from frequent fires (e.g., biennial burns) might not accommodate habitat requirements of all grassland-dependent species (lark sparrow commonly noted above the ground in cholla cactus and its occurrence might be less affected by reductions in herbaceous cover). Further improvements in nesting success might be achieved if other management techniques induce more pronounced reductions in shrub density than we observed resulting from dormant-season fires. Such reductions could be accomplished by means of chemical or mechanical shrub removal. Burning in conditions that enhance fire intensity (e.g., high biomass, drought), or burns during the growing season, may also increase shrub mortality, but should be done judiciously. Conversely, burning under lower fuel loads (e.g., heavily grazed pastures), may be ineffectual in reducing shrub density. Large-scale adoption of prescribed burning of shortgrass prairie rangelands may be socially resisted. However, conservation officials in the southern Great Plains should continue to promote the benefits of prescribed burning and other forms of brush removal, where appropriate. Coordination of such management efforts with scientifically rigorous research on wildlife responses would help guide the restoration of southern Great Plains rangelands.

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