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# GRASSLAND BIRD COMMUNTIY RESPONSE TO LARGE WILDFIRES

ANTHONY J. ROBERTS,[1,6,7] CLINT W. BOAL,[2] DAVID B. WESTER,[3,4]
SANDRA RIDEOUT-HANZAK,[3] AND HEATHER A. WHITLAW[5]

ABSTRACT.—We studied breeding season communities of grassland birds on short-grass and mixed-grass prairie sites during the second and third breeding seasons following two large wildfires in March 2006 in the Texas panhandle, USA. There was an apparent temporary shift in avian community composition following the fires due to species-specific shifts associated with life-history traits and vegetation preferences. Species that prefer sparse vegetation and bare ground on short-grass sites, such as Horned Lark (*Eremophila alpestris*), benefited from wildfires, while others, such as Western Meadowlark (*Sturnella neglecta*), that prefer more dense vegetation, were negatively impacted. Mixed-grass sites had species-specific shifts in 2007, two breeding seasons after the fires; grassland bird communities on burned plots were similar by 2008 to those on unburned plots. Avian communities appeared to return to pre-burn levels within 3 years following wildfires. Many of the responses in our study of wildfire were similar to those reported following prescribed fires elsewhere. Prescribed fires appear to have similar effects on the avian community despite differences in intensity and environmental conditions during wildfires. *Received 8 November 2010. Accepted 10 October 2011.*

Fire is a driving ecological process of healthy grassland ecosystems. Following a history of fire suppression, the application of prescribed fire has become a contemporary method to manage grasslands. However, it is unclear how well prescribed fires can replicate the ecological values of natural fires. For example, studies of grassland bird response to fire have focused on effects of prescribed fire (e.g., Huber and Steuter 1984, Madden et al. 1999, Kirkpatrick et al. 2002, Grant et al. 2010); no published studies have examined the impact of wildfires on avian communities in Great Plains grasslands. Prescribed fires are generally conducted in low wind and high humidity conditions that promote controlled burning. Wildfires may be more intense than prescribed fires and cover larger spatial scales. Litter is decreased, woody plants are often killed or burned completely, and live plants are killed at high rates (Rideout-Hanzak et al. 2011). This removes structure from the landscape, affecting the poorly understood avian community in unknown ways (Smith 2000). There is potential for increased frequency and size of wildfires in the coming decades due to warmer temperatures and less precipitation, as predicted by current climate change models (North American Bird Initiative 2010). Numerous bird species of concern occur in the short-grass prairies of the southern Great Plains, an assemblage of some of the least ecologically understood of all grassland birds (Askins et al. 2007). The effects of wildfires on these and other species are unknown.

Two large wildfires ignited east of Amarillo in the Texas panhandle, USA on 12 March 2006. Together the fires burned 360,000 ha of predominantly private lands in what is known as the East Amarillo Complex (EAC) wildfires (Zane et al. 2006). Widespread vegetation loss on the mixed-grass and short-grass ecosystems had potentially large negative impacts on numerous species of grassland obligate songbirds that breed in the Southern High Plains. The EAC presented a rare opportunity to examine the effects of large-scale wildfires on grassland bird populations. The objectives of our study were to: (1) examine the changes in avian species densities after a wildfire, and (2) how avian community composition adjusts in the years following the wildfires.

## METHODS

This study was conducted on private ranches in Roberts, Gray, and Donley counties in the Texas panhandle. This area is in the Rolling Plains ecoregion of Texas, a transition zone between short-grass and mixed-grass prairie types in the Southern High Plains. The landscape

[1] Department of Natural Resources Management, Texas Tech University, Lubbock, TX 79409, USA.

[2] U.S. Geological Survey, Texas Cooperative Fish and Wildlife Research Unit, Texas Tech University, Lubbock, TX 79409, USA.

[3] Department of Animal, Rangeland, and Wildlife Sciences, Texas A&M University-Kingsville, Kingsville, TX 78363, USA.

[4] Caesar Kleberg Wildlife Research Institute, Texas A&M University-Kingsville, Kingsville, TX 78363, USA.

[5] U.S. Fish and Wildlife Service, West Texas Sub-Office, Texas Tech University, Lubbock, TX 79409, USA.

[6] Current address: Wildland Resources Department, 5230 Old Main Hill, Logan, UT 84322, USA.

[7] Corresponding author;
e-mail: tony.roberts@aggiemail.usu.edu

is characterized by rolling hills leading to flat plains interspersed with ephemeral wetland depressions known as playas (Williams and Welker 1966). Elevation ranges from 420 to >600 m. The climate is characterized by hot summers and cold to mild winters, but temperatures fluctuate extensively within seasons (Williams and Welker 1966). The mean summer temperature (May–Jul) for 2007 and 2008 was 22.3°C (U.S. Department of Commerce 2009). Historically the Texas panhandle received an average of 53 cm of precipitation a year (Williams and Welker 1966); the study area received ~61 cm of precipitation annually during the study period.

*Study Plot Selection.*—We selected 20 survey plots for study based on access to private property, burn history, and vegetation community type of either short-grass or mixed-grass prairie (centered at 14 S 342427 E, 3928890 N). The latter distinction was based on soil type and dominant vegetation. Burn type, either burned or unburned area, was established by talking with landowners and local officials, and visual examination of any woody vegetation in the area such as prickly pear cactus (*Opuntia* spp.), catclaw mimosa (*Mimosa aculeaticarpa*), or sand sagebrush (*Artemisia filifolia*). We were unable to initiate this study and select sites until the fall following the EAC fires and were unable to examine avian communities during the breeding season directly after the fires occurred. The 20 plots were equally distributed among areas identified as short-grass burned, short-grass unburned, mixed-grass burned, and mixed-grass unburned. Individual plots were at least 1 km apart and 0.5 km from a known fire boundary, roads, or other vegetation or topographic changes. We analyzed each vegetation type separately to examine within-type differences among five replicates each of burned and unburned plots. All study sites were on private property and we were unable to alter grazing regimes; sites ranged from no grazing to moderate stocking levels.

*Breeding Season Surveys.*—We used fixed-radius point counts to survey avian populations within the 20 plots, May-June, 2007 and 2008. We conducted surveys using a three by three grid of nine 75-m radius point counts for a total survey area of ~16 ha at each plot. We used three observers throughout the study, all of whom had either prior experience with grassland bird identification or were trained prior to surveys. Point-count centers were placed 200 m apart to minimize risk of recounting individuals (Ralph et al. 1993). We conducted surveys between 0.5 hrs before and 3 hrs after sunrise. We did not survey points during inclement weather such as rain, or in winds >16 km/hr (Ralph et al. 1993). Observers recorded all birds flushed within 75 m of the point when walking to a point. All birds heard or seen in the 75-m radii within a 7-min window were recorded and the distance from the observer was estimated (Reynolds et al. 1980). Birds seen while walking between points and birds that flew overhead but not using the area within the radii were not recorded.

*Data Analysis.*—We used both the May and June surveys to create a measure of average abundance and species diversity measures over the breeding season. Average abundance was calculated by averaging the count of each species between the two survey periods. We calculated species diversity using the Shannon diversity index (H′; Shannon and Weaver 1963). Taylor (1978) suggested that diversities calculated over a variety of samples are normally distributed, but we used the more robust *t*-test suggested by Hutcheson (1970) to compare H′ across burn conditions. Hutcheson's *t*-test uses the number of individuals to calculate degrees of freedom, often resulting in large numbers. We derived evenness (E) from H′ using the ratio of observed to maximum diversity as described by Magurran (1988).

We only used singing males from June surveys to calculate density estimates. We were only interested in the number of breeding territories supported in each habitat type in contrast to richness and diversity measures. June surveys were during the height of the breeding season when territories were established, and indicative of the density of breeding pairs. We used actual counts of singing males rather than estimates of density derived using detection probabilities. Species-specific estimates of detection probabilities introduce additional sources of variability (Johnson 2008), and we believed detection probability in a grassland landscape is close to one and similar across burn conditions, especially given our survey was based on territorial males that are announcing their presence (Thompson et al. 2009). Density estimates are an average of the five plots in each treatment. We compared densities among years and burn treatments using a *t*-test with an alpha level of 0.05. Our interest

TABLE 1. Avian community measures for breeding season birds among burned and unburned plots associated with the East Amarillo, Texas Complex wildfires of 2006.

| | Short-grass | | Mixed-grass | |
|---|---|---|---|---|
| | Burned ($n$ = 5) | Unburned ($n$ = 5) | Burned ($n$ = 5) | Unburned ($n$ = 5) |
| Species abundance (#/ha) | | | | |
| 2007 | 1.806 | 1.425 | 1.164 | 1.702 |
| 2008 | 1.714 | 1.220 | 1.570 | 1.446 |
| Species richness | | | | |
| 2007 | 11 | 12 | 9 | 11 |
| 2008 | 14 | 10 | 13 | 13 |
| Shannon diversity index (H') | | | | |
| 2007 | 1.165 ± 0.119 | 1.336 ± 0.092 | 1.470 ± 0.075 | 1.442 ± 0.067 |
| 2008 | 1.669 ± 0.071 | 1.582 ± 0.070 | 1.605 ± 0.078 | 1.804 ± 0.061 |
| Species evenness (E) | | | | |
| 2007 | 0.589 | 0.693 | 0.826 | 0.782 |
| 2008 | 0.798 | 0.835 | 0.753 | 0.853 |

was the effect of wildfire, not vegetation type, and we compared densities only between burned and unburned plots within short- and mixed-grass vegetation types, not between the two vegetation types. Effect sizes were calculated using Cohen's $d$ (Cohen 1988) with a value $\geq$0.8 considered a large effect, and a value $\leq$0.2 a small effect. We present Cohen's $d$ supplemental to $t$-tests to facilitate assessments of the effects of year and burn treatment. Results may not be statistically significant, but the effect size can provide insights on biological importance.

## RESULTS

*Short-grass.*—We detected 13 species on short-grass plots in 2007 with 11 occurring on burned and 12 on unburned plots (Table 1). Fourteen species were detected on burned plots and 10 on unburned plots in 2008 (Table 1). Avian diversity was higher ($t_{239}$ = 2.06, $P$ = 0.04) on unburned areas in 2007 but was similar across burn conditions in 2008 (Table 1). Diversity on burned plots was lower ($t_{223}$ = 4.99, $P$ < 0.001) in 2007 but there was no difference between years on unburned plots.

We performed density analyses for the four most common species detected: Grasshopper Sparrow (*Ammodramus savannarum*), Lark Sparrow (*Chondestes grammacus*), Horned Lark (*Eremophila alpestris*), and Western Meadowlark (*Sturnella neglecta*). On average, Grasshopper Sparrows were present at the highest density of all species recorded during point-count surveys, ranging from 0.32 to 0.44 birds/ha. Differences between burn conditions ($t_8$ = 0.10, $P$ = 0.46) or between years ($t_8$ = 0.53, $P$ = 0.30) were not significant. Estimated densities of Horned Lark were 0.21/ha on unburned and 0.24/ha on burned plots in 2007. Densities on burned plots were not different between 2007 and 2008 (0.21/ha; $t_8$ = 0.20, $P$ = 0.42). Despite a 50% decrease in density (0.11/ha) among unburned plots in 2008, there was no evidence of statistical or biological significance to the data ($t_8$ = 0.84, $P$ = 0.21, $d$ = −0.6). However, moderate effect sizes suggest densities in 2008 were higher ($t_8$ = 0.94, $P$ = 0.19, $d$ = 0.60) on burned plots. Estimated densities of Western Meadowlarks in 2007 were higher ($t_8$ = 1.41, $P$ = 0.098, $d$ = 1.0) on unburned (0.45/ha) compared to burned plots (0.36/ha). The abundance of Western Meadowlarks decreased to 0.39/ha in 2008 on unburned plots ($t_8$ = 0.85, $P$ = 0.21, $d$ = −1.4), but was not significant (Table 2). Lark Sparrows occurred in greater numbers on burned plots in both years and accounted for >10% of all species detected overall (Table 2). Effect sizes suggest Lark Sparrow densities were greater ($t_8$ = 1.18, $P$ = 0.14, $d$ = 0.73) on burned (0.35/ha) compared to unburned (0.14/ha) plots in 2007; differences between years within plot types were not significant (Table 2). Lark Sparrow densities were similar in 2008 on both burned (0.28/ha) and unburned (0.15/ha) plots. Moderate effect sizes suggests densities were higher on burned plots in 2008 ($d$ = 0.57).

TABLE 2. Avian abundance (percent composition) on burned and unburned short-grass plots associated with the East Amarillo, Texas Complex wildfires of 2006.

| Species | Short-grass | | | | Totals |
|---|---|---|---|---|---|
| | 2007 | | 2008 | | |
| | Burned ($n$ = 5) | Unburned ($n$ = 5) | Burned ($n$ = 5) | Unburned ($n$ = 5) | |
| Northern Bobwhite (*Colinus virginianus*) | 0 | 0 | 1 (0.8) | 0 | 1 (0.2) |
| Scaled Quail (*Callipepla squamata*) | 0 | 2 (1.9) | 0 | 0 | 2 (0.4) |
| Killdeer (*Charadrius vociferous*) | 1 (0.7) | 0 | 1 (0.8) | 1 (0.8) | 3 (0.6) |
| Mourning Dove (*Zenaida macroura*) | 0 | 0 | 4 (3.1) | 5 (4.3) | 9 (1.8) |
| Common Nighthawk (*Chordeiles minor*) | 8 (5.8) | 1 (0.9) | 7 (5.3) | 5 (4.3) | 21 (4.3) |
| Scissor-tailed Flycatcher (*Tyrannus forficatus*) | 2 (1.4) | 3 (2.9) | 2 (1.5) | 4 (3.4) | 11 (2.2) |
| Horned Lark (*Eremophila alpestris*) | 31 (22.4) | 15 (14.4) | 25 (19.1) | 15 (12.9) | 86 (17.6) |
| Cliff Swallow (*Petrochelidon pyrrhonota*) | 0 | 1 (0.9) | 1 (0.8) | 0 | 2 (0.4) |
| Barn Swallow (*Hirundo rustica*) | 1 (0.7) | 1 (0.9) | 2 (1.5) | 0 | 4 (0.8) |
| Dickcissel (*Spiza americana*) | 2 (1.4) | 1 (0.9) | 0 | 0 | 3 (0.6) |
| Cassin's Sparrow (*Peucaea cassinii*) | 4 (2.9) | 2 (1.9) | 6 (4.6) | 6 (5.2) | 18 (3.7) |
| Grasshopper Sparrow (*Ammodramus savannarum*) | 33 (23.9) | 33 (31.7) | 27 (20.6) | 30 (25.9) | 124 (25.3) |
| Lark Sparrow (*Chondestes grammacus*) | 26 (18.8) | 11 (10.6) | 21 (16.0) | 12 (10.3) | 70 (14.3) |
| Western Meadowlark (*Sturnella neglecta*) | 28 (20.3) | 33 (31.7) | 26 (19.8) | 33 (28.4) | 120 (24.5) |
| Brown-headed Cowbird (*Molothrus ater*) | 2 (1.4) | 1 (0.9) | 6 (4.6) | 5 (4.3) | 14 (2.9) |
| Common Grackle (*Quiscalus quiscula*) | 0 | 0 | 2 (1.5) | 0 | 2 (0.4) |
| Totals | 138 | 131 | 104 | 89 | 490 |

*Mixed-grass.*—We observed 12 species on mixed-grass plots in 2007 during the breeding season with nine detected on burned and 11 on unburned plots (Table 1). We detected 17 species on mixed-grass plots in 2008. Barn Swallow (*Hirundo rustica*), Eastern Kingbird (*Tyrannus tyrannus*), Killdeer (*Charadrius vociferous*), and Lesser Prairie-Chicken (*Tympanuchus pallidicinctus*) were all observed in low numbers on burned plots, but were not detected on unburned plots (Table 3). There was no difference in diversity between burned and unburned plots in 2007. However, diversity on unburned mixed-grass plots in 2008 increased ($t_{268}$ = 3.45, $P < 0.001$) from that in 2007 and was higher ($t_{235}$ = 2.59, $P$ = 0.01) among burned plots (Table 1). No change was detected in diversity between years on burned plots.

We analyzed Western Meadowlark, Cassin's Sparrow (*Peucaea cassinii*), Lark Sparrow, and Grasshopper Sparrow densities on mixed-grass plots. Western Meadowlark densities ranged from 0.31 to 0.45 birds/ha; no differences were detected between burn conditions ($t_8$ = 0.39, $P$ = 0.35) or years ($t_8$ = 0.17, $P$ = 0.43). Estimated densities of Cassin's Sparrows in 2007 were 0.36/ha on unburned compared to 0.29/ha on burned plots ($t_8$ = 0.35, $P$ = 0.36, $d$ = 0.3). Densities did not change significantly on burned plots ($t_8$ = 0.07, $P$ = 0.47, $d$ = 0.1) or unburned plots ($t_8$ = 0.39, $P$ = 0.35, $d$ = −0.3); Cassin's Sparrows occurred at identical abundance by 2008 on both plot types (0.30/ha). Grasshopper Sparrows occurred at significantly higher ($t_8$ = 2.2, $P$ = 0.031, $d$ = 1.5) densities on unburned (0.45/ha) than burned plots (0.12/ha) in 2007. Densities numerically decreased ($t_8$ = 1.26, $P$ = 0.12, $d$ = −0.9) in 2008 on unburned plots (0.35/ha) and effect size suggest densities increased ($t_8$ = 1.17, $P$ = 0.13, $d$ = 0.8) on burned plots (0.32/ha). Lark Sparrow densities were nearly identical on burned (0.23/ha) and unburned (0.24/ha) plots in 2007. The following year densities on both burn conditions were 0.15/ha; although the numerical decreases were not significant, it appears there was a noticeable effect of year on both burned ($t_8$ = 0.83, $P$ = 0.22, $d$ = 0.53) or unburned ($t_8$ = 0.85, $P$ = 0.21, $d$ = 0.55) plots.

## DISCUSSION

Grasshopper Sparrows were the most abundant species among short-grass plots, and were present in consistent numbers regardless of year or burn condition. This species was significantly more abundant on unburned plots after a wildfire in Montana shrubsteppe (Bock and Bock 1987), suggesting preference for denser vegetation in arid western landscapes. In contrast, lower numbers of Grasshopper Sparrows were detected on burned areas for 2 years following a grassland

TABLE 3. Avian abundance (percent composition) on burned and unburned mixed-grass plots associated with the East Amarillo, Texas Complex wildfires of 2006.

| | Mixed-grass | | | | |
|---|---|---|---|---|---|
| | 2007 | | 2008 | | |
| Species | Burned ($n$ = 5) | Unburned ($n$ = 5) | Burned ($n$ = 5) | Unburned ($n$ = 5) | Totals |
| Northern Bobwhite | 1 (1.1) | 2 (1.5) | 2 (1.7) | 6 (4.3) | 11 (2.3) |
| Lesser Prairie-Chicken (*Tympanuchus pallidicinctus*) | 0 | 0 | 1 (0.8) | 0 | 1 (0.2) |
| Killdeer | 0 | 0 | 2 (1.7) | 0 | 2 (0.4) |
| Mourning Dove | 0 | 3 (2.3) | 2 (1.7) | 16 (11.4) | 21 (4.4) |
| Common Nighthawk | 4 (4.5) | 7 (5.3) | 12 (10.0) | 7 (5.0) | 30 (6.2) |
| Eastern Kingbird (*Tyrannus tyrannus*) | 0 | 0 | 1 (0.8) | 0 | 1 (0.2) |
| Scissor-tailed Flycatcher | 3 (3.4) | 0 | 2 (1.7) | 3 (2.1) | 8 (1.7) |
| Horned Lark | 0 | 0 | 0 | 1 (0.7) | 1 (0.2) |
| Cliff Swallow | 0 | 0 | 0 | 3 (2.1) | 3 (0.6) |
| Barn Swallow | 0 | 5 (3.8) | 1 (0.8) | 0 | 6 (1.2) |
| Blue Grosbeak (*Passerina caerulea*) | 0 | 1 (0.8) | 0 | 0 | 1 (0.2) |
| Dickcissel | 3 (3.4) | 3 (2.3) | 0 | 0 | 6 (1.2) |
| Cassin's Sparrow | 22 (24.7) | 28 (21.1) | 31 (25.8) | 27 (19.3) | 108 (22.4) |
| Grasshopper Sparrow | 9 (10.1) | 35 (26.3) | 23 (19.2) | 22 (15.7) | 89 (18.5) |
| Lark Sparrow | 18 (20.2) | 19 (14.3) | 12 (10.0) | 12 (8.6) | 61 (12.7) |
| Western Meadowlark | 25 (28.1) | 29 (21.8) | 29 (25.2) | 30 (21.4) | 113 (23.4) |
| Eastern Meadowlark (*Sturnella magna*) | 0 | 0 | 0 | 3 (2.1) | 3 (0.6) |
| Brown-headed Cowbird | 4 (4.5) | 1 (0.8) | 2 (1.7) | 8 (5.7) | 15 (3.1) |
| Bullock's Oriole (*Icterus bullockii*) | 0 | 0 | 0 | 2 (1.4) | 2 (0.4) |
| Totals | 89 | 120 | 133 | 140 | 482 |

wildfire in Arizona (Bock and Bock 1992). High effect sizes indicate the higher densities of Western Meadowlarks on burned plots in 2007 may have been biologically relevant, but were not statistically significant in our study. The aboveground plant biomass among burned and unburned plots was similar by 2008 (Rideout-Hanzak et al. 2011), likely providing similar nesting sites and foraging opportunities across burn conditions. Horned Larks prefer areas with a high percentage of bare ground across their range (Beason 1995). Consistent with this general habitat association, Horned Larks decreased in density as burned areas were revegetated and more litter formed.

The composition of avian communities differed on mixed-grass plots in 2007 despite similar diversity and evenness measures across burn conditions. Abundances of Cassin's and Grasshopper sparrows were higher on unburned than burned plots in mixed-grass areas. However, densities of these species on burned plots were similar to unburned plots by 2008. Lark Sparrows on both short- and mixed-grass plots decreased on burned areas from 2007 to 2008. Long-term Lark Sparrow abundance decreased along with the decrease in woody vegetation after a fire in sagebrush (*Artemisia* spp.) grasslands in Washington State (Earnst et al. 2009). This suggests the avian community may have returned to densities similar to unburned areas by the third breeding season after the wildfires. A homogeneous landscape with plant biomass and structure similar to unburned areas 3 years after the wildfires (Rideout-Hanzak et al. 2011) may have promoted similar avian communities across the landscape.

Historically, fire was a major ecological factor in both short- and mixed-grass ecosystems until intense livestock grazing and fire suppression altered vegetation and fuels so fires could not burn with historic frequency or intensity (Wright and Bailey 1982). Some areas of the Great Plains have seen fire return to the landscape in the form of prescribed fire. The EAC wildfires occurred during high winds and low humidity, different conditions than proscribed for prescribed fires. Prescribed fires appear to have similar influences on the avian community as the EAC wildfires despite differences in intensity and environmental conditions during the wildfire.

Many of the responses measured in our study of wildfire were similar to those observed following prescribed fires. Wintering Cassin's Sparrows in Arizona responded negatively to burning after a prescribed fire, but Grasshopper Sparrows showed no response (Gordon 2000). Abundance of Grasshopper and Cassin's sparrows increased after a prescribed fire in Texas mesquite (*Prosopis* spp.) savanna (Lee 2006). Prescribed fire in North Dakota mixed-grass prairie decreased populations of most species during the breeding season following the fires (Grant et al. 2010); however, 3 years after the prescribed fires, avian populations had increased and stabilized, recovering in a similar time span as in our study. Many species, including Grasshopper Sparrows and Western Meadowlarks, were positively correlated with use of prescribed fire in mixed-grass prairies (Madden et al. 1999). Western Meadowlarks in prairie Canada declined in abundance during the breeding season following prescribed fire, but had comparable densities on burned and unburned areas 3 years post-fire (Pylypec 1991).

We found an apparent temporary shift in avian community composition following wildfires due to species-specific shifts associated with life-history traits and vegetation preferences. The avian community appeared to be similar to that on unburned plots of similar grass types 3 years following the wildfires. This was consistent with vegetation recovery (Rideout-Hanzak et al. 2011). A homogeneous landscape in grasslands decreases the diversity of grassland birds (Fuhlendorf et al. 2006) and the grassland bird community reaches peak densities with increased periodic disturbance in short-grass and mixed-grass landscapes. Two of the most common species detected on short-grass plots, Grasshopper Sparrow and Horned Lark, are among 20 common North American birds experiencing the steepest population declines (Butcher and Niven 2007). The area burned by the EAC wildfires may not only provide important habitat for continued persistence of species of concern, but fire may be an integral component of habitat health for the avian community.

Persistence of a diverse and abundant avian community is dependent on periodic disturbances such as wildfire or prescribed fire, grazing, and drought to provide patches of habitat in varying stages of growth after disturbance (Fuhlendorf and Engel 2001). Prescribed fire has been used to mimic wildfire effects and reduce wildfire potential (Pattison 1998). Rideout-Hanzak et al. (2011) suggest the EAC fires may not have created drastically different conditions than a prescribed fire in this ecosystem; this was corroborated by the avian community response. The wildfires may have been ecologically beneficial in providing similar services to plants and soil as historic fire regimes on the Southern High Plains. The combination of varying grazing regimes and periodic prescribed fire in the Texas panhandle would facilitate development of a mosaic of grassland patches in varying stages of recovery from disturbance, and offer a wide variety of niches for grassland birds.

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