Response of northern bobwhite movements to management-driven disturbance in a shrub dominated ecosystem

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Abstract

Despite inhabiting fire-adapted grasslands and shrublands across much of their continental distribution, northern bobwhite (*Colinus virginianus*; hereafter bobwhite) behavior relative to disturbance (e.g., fire) is poorly understood, especially in the western fringe of their distribution. We assessed bobwhite movement and space use following dormant season burning (Jan-Mar; 2013-2014) in a sand shinnery oak (*Quercus havardii*; hereafter shinnery oak) plant community. We captured and radio-marked bobwhites ($n = 369$) and monitored them via radio-telemetry across burn treatments (averaging 254 ha) ranging from 0-12, 13-24, 25-36, and >36 months post fire (hereafter, time since fire; TSF) at the Packsaddle Wildlife Management Area in western Oklahoma, USA. Mean covey home range size was $76.6 \text{ ha} \pm 5.9 \text{ [SE]}$ (range; 12-270 ha) ($n = 61$ coveys), which is substantially larger than covey home ranges reported for other regions. Prescribed fire affected space use of coveys ($F_{4, 54} = 2.95$, $p < 0.05$) which was driven by smaller home range sizes of coveys using 25-36 TSF ($52.07 \text{ ha} [\pm 6.6]$) than 0-12 ($85.0 \text{ ha} [\pm 15.53]$), 13-24 ($86.7 \text{ ha} [\pm 20.7]$) and >36 TSF ($78.9 \text{ ha} [\pm 6.54]$). Generalized linear mixed models demonstrated that neither spring dispersal (movements or area traversed) were correlated with TSF, age, or sex ($n=114$), further demonstrating a minimal effect of prescribed fire; however, dispersal areas were greater in 2013 than in 2014 ($p < 0.05$). Our research shows that prescribed fire applied at a landscape scale had limited effects on short-term bobwhite movement and space use. These findings also suggest that in shinnery oak vegetation communities land managers can utilize prescribed fire across large spatial extents without substantially altering the space use or movement of bobwhites.

Key words: *Colinus virginianus*, dispersal, home range, movement, prescribed burning, prescribed fire.

Introduction

Northern bobwhite (*Colinus virginianus*; hereafter bobwhite) have undergone a widespread decline in abundance throughout a large portion of their historical distribution for more than 50 years (Stoddard 1931, Brennan 1991, Sauer et al. 2011). Habitat loss and fragmentation related to land conversion, anthropogenic development, and large scale fire suppression are considered major contributors to declining populations of bobwhites (Brennan 1991, Fies et al. 2002, Peterson et al. 2001, Brennan and Kuvlesky 2005). Consequently, the preservation and management of remaining bobwhite habitat is increasingly important.

Few species have received as much research or management attention as bobwhites (Hernandez and Guthery 2012). However, despite inhabiting fire-adapted grasslands and shrublands across much of their continental distribution, topics related to bobwhite ecology and fire are poorly understood within their western distribution. For example, more than 35 years have passed since Renwald et al. (1978) noted the lack of knowledge regarding bobwhite responses to prescribed fire in the western portion of its distribution; yet, our understanding of bobwhite ecology and management relative to fire is still limited. Although prescribed fire has been shown to contribute to improved bobwhite habitat in the southeastern United States (Cram et al. 2002, Jones and Chamberlain 2004, Burke et al. 2008), direct management implications from the few studies that have assessed bobwhite responses to fire in Oklahoma and Texas have

69 been less definitive (Baumgartner 1946, Wilson and Crawford 1979, Carter et al. 2002, Ransom
70 and Schulz 2007). This uncertainty is especially problematic because prescribed fire inhibits
71 woody encroachment and therefore is identified as a key component for the long term
72 conservation of rangelands in North America (Fuhlendorf and Engle 2004, Engle et al. 2008).
73 As a result, land managers aiming to maintain rangeland ecosystems (i.e., long term) and provide
74 high quality bobwhite habitat (i.e., short term and long term) are faced with substantial
75 ambiguity given the scarcity of information on the temporal impacts of prescribed fire on
76 bobwhite behavior, especially on highly dynamic landscapes. For example, Wellendorf and
77 Palmer (2009) provided an informative study that examined bobwhite home ranges from two
78 different burn scale extents (i.e., 2 and 8 ha) in Florida, but little empirical data exists on
79 bobwhite space use (e.g., home range) for much larger burns scales. This is especially true in the
80 more xeric Great Plains where burning is often applied on the landscape at extents in excess of
81 100 ha (Renwald et al. 1978, Powell and Busby 2013, Holcomb et al. 2014).

82 The population and management implications of dispersal are understudied aspects of
83 bobwhite ecology (Fies et al. 2002, Cook et al. 2009). Movements appear to be more frequent
84 and in many cases much larger during the spring following covey break-up (Townsend et al.
85 2003, Fies et al. 2002, Cook et al. 2009) and these movements ultimately influence the spatial
86 distribution of breeding bobwhites across the landscape (Errington 1933, Lehmann 1946,
87 Jackson 1960; Appendix C). Management-driven disturbances such as prescribed fire
88 temporarily alter vegetation structure (Peterson and Reich 2001, Harrell et al. 2001) and
89 therefore may impact how bobwhites perceive and traverse the landscape. For example,
90 management prescription applied at landscape scales could influence bobwhite space use during
91 the covey period (i.e., prior to dispersal) and movements during dispersal by altering forage

availability or vegetation structure (i.e., predation cover, thermal cover, and nesting cover). Additionally, the covey period and dispersal phenology of bobwhites temporally overlaps with the application of dormant season prescribed fire (i.e., winter and early spring), and therefore provides a direct opportunity to assess bobwhite dispersal relative to prescribed fire.

Bridging the gap between management practices and future conservation goals requires an understanding of how bobwhites respond to habitat alterations at scales that are similar to the scale of management practices applied on the landscape (Fies et al. 2002, Wellendorf and Palmer 2009, Cook et al. 2009). Our objectives were to, 1) measure bobwhite covey home ranges to assess spatial responses to time since fire at a scale similar to the scale of management on the study area, and 2) calculate spring dispersal in order to explore how landscape level habitat manipulation impacts bobwhite behavior and space use.

Methods

Study Area

We investigated bobwhite covey home range size and dispersal across a shrub and mixed grass landscape at the Packsaddle Wildlife Management Area (WMA) in western Oklahoma, USA, during winter and spring 2013–2014. Packsaddle WMA (7,956 ha) is owned and managed by the Oklahoma Department of Wildlife Conservation. The vegetation at the study area mostly consists of shrubs and grasses and is dominated by sand shinnery oak (*Quercus havardii*; hereafter shinnery oak). In addition to shinnery oak, other shrubs include sand sagebrush (*Artemisia filifolia*), sand plum (*Prunus angustifolia*) and aromatic sumac (*Rhus aromatica*) (DeMaso et al. 1997, Vermeire and Wester 2001). Herbaceous species such as little bluestem (*Schizachyrium scoparium*), big bluestem (*Andropogon gerardii*), sideoats grama (*Bouteloua*

curtipendula), blue grama (*Bouteloua gracilis*), western ragweed (*Ambrosia psilostachya*), Texas croton (*Croton texensis*) and prairie sunflower (*Helianthus petiolaris*) are common (DeMaso et al. 1997, Peterson and Boyd 1998). Terrain is flat to rolling, with the predominant soil types being sandy Brownfield and Nobscott types (Wiedeman and Penfound 1960).

Regionally, mean annual precipitation ranges from 241.8 - 746.6 mm, averaging 631.4 mm from 1981-2010 and 558.8 mm per year from 1994 – 2014 (Arnett Oklahoma Mesonet Site; Oklahoma Mesonet 2014). Rainfall was below average for 2012 (425.90 mm) (i.e., the year prior to initiation of our study), 2013 (531.90 mm) and 2014 (432.31 mm) (Arnett Oklahoma Mesonet Site; Oklahoma Mesonet 2014). Notably, this study occurred in the years following historic drought conditions that occurred during 2011 when average yearly precipitation was 241.8 mm (Arnett Oklahoma Mesonet Site; Oklahoma Mesonet 2014).

Prescribed fire and grazing are the two primary habitat management practices on the study area with wildlife conservation and management as the primary goal. Prescribed fires have been a consistent management practice on the study area since 2004 and are almost entirely conducted during the dormant season. Approximately 1,372 ha and 1,821 ha of the study area were burned by ODWC personnel from January to late March 2013 and 2014, respectively. The mean ($\pm$ SE) size of burned pastures was 254.3 ha ($\pm$ 33.7) ha and ranged from 8.5-626.5 ha; however, 80% and 50% of burns were > 100 ha and > 200 ha; respectively. We targeted our study on a focal area where prescribed burning and grazing was planned as an ongoing management practice (Fig. 1) which also provided a spatial extent for bobwhite trapping sites. Stocking rate of livestock was 1.6 ha per AUM was uniformly applied on the focal study area from April 1 – August 15 during each year of the study and was not stocked during other times of the year.

Capture and Radio-monitoring

During autumn and winter (October-March) 2013 and 2014, we trapped bobwhites using baited Stoddard style funnel traps (Stoddard 1931). Traps were distributed across the study area in order to maximize the potential to capture bobwhites, but also to obtain adequate coverage of the study area. Traps were checked daily in the morning and evening. We weighed each captured bobwhite and determined sex and age using sexually dimorphic plumage identifiers and age specific characteristics of wing features (Leopold 1939, Leopold 1945, Rosene 1969). Each captured bobwhite weighing > 130 g ($n=369$) was fitted with a 6 g necklace radio-collar equipped with an 8 hr mortality sensor (Advanced Telemetry Systems, Isanti, Minnesota, USA) and banded with a uniquely numbered aluminum leg band (size 7; National Band and Tag, Newport, Kentucky, USA).

We monitored bobwhites via radio-telemetry using the homing technique (White and Garrott 1990) to determine covey locations during autumn and winter and individual bird locations during spring (i.e., April and May). At each bird location, we circled radio-marked individuals at a distance of 15-20 m and recorded an estimated distance and bearing from the observer to the covey (i.e., winter) or individual (i.e., spring). We obtained 2-3 locations per week for coveys during the covey period (i.e., 1 November thru spring dispersal) and 4-7 locations per week for individuals during the spring. For each radio-monitored covey, a single location was recorded and other radio-marked members of the covey were confirmed to be in association with the covey. Locations of each individual bird were recorded during the dispersal period. We used movement data to determine covey break-up dates and considered covey break-up to have occurred when one or more individuals of a covey were no longer affiliated with their respective covey (Liberati and Gates 2012). We searched for missing birds using a combination

of techniques including intensively searching a 3 km circle surrounding the most recent bird location, checking for signals from high terrain points on the study area, and using a truck-mounted dipole omni-directional antenna (Advance Telemetry Systems, Isanti, Minnesota) on local roads. The spring dispersal period was defined as the period between covey break-up and nest initiation for nesting birds or median nest initiation date for birds that were not observed nesting (22 May for 2013 and 25 May for 2014). We located nests by searching the locations of both males and females when radio-telemetry indicated that they were in the same site for two consecutive locations. To estimate nest initiation dates, clutch size was recorded at each nest site following the onset of incubation. We assumed that 1 egg was laid per day (Rosene 1969, Lehmann 1984) and used the clutch size at the onset of incubation to back-date estimates of nest initiation date.

Covey Home Range

We estimated bobwhite covey home ranges for 22 and 39 coveys in 2013 and 2014, respectively. To determine covey home range, we used the Geospatial Modeling Environment (Beyer 2014) to calculate fixed kernel density estimates (Worton 1987, Worton 1989) for coveys with ≥ 20 locations (Taylor et al. 1999, Singh et al. 2011, Peters et al. 2015). For coveys used in home range analysis, radio-locations per covey ranged from 20-75 locations and averaged ($\pm$ SE) 42.2 ($\pm$ 2.1) locations during the covey period, totaling 942 and 1,634 independent radio-locations in 2013 and 2014, respectively. We used maximum likelihood cross-validation to estimate smoothing parameters given that it is considered an appropriate estimation method when sample sizes are relatively small (Horne and Garton 2006) and calculated 95% isopleths (ha) for each covey. We used a mapped spatial layer of burn treatments to compare covey home ranges among different time since fire (hereafter, TSF) (0-12 months, 13 – 24 months, 25 – 36,

and >36 months post fire). We designated a covey or individual into a TSF if >50% of radio-locations occurred within a TSF treatment. This seemed to be an ecologically relevant categorization for our dataset given that only 2 coveys did not fit into any specific category; specifically, ~3% of those monitored. Thus, these two coveys (12 individuals) were removed from TSF comparisons because proportions of locations were each less than 50% of the total locations and were distributed across 3 different TSF intervals (i.e., 0-12, 13-25 and >36 months post fire). However, > 90% and > 60% of locations were observed within a single TSF interval for 69% and 97% of coveys, respectively.

We examined differences in covey home range size using a 2-factor, factorial analysis of variance (ANOVA) (Zar 1984) with year and TSF as categorical factors in program R (Zuur et al. 2009, R Development Core Team 2012). Because home range estimate distributions were right-skewed, we applied a $\log_n$ transformation prior to analysis (Singh et al. 2011). When significant mean differences were reported by ANOVA results, we tested for mean differences for year and TSF category combinations using a Tukey-Kramer honestly significant difference test (Zar 1984, Sokal and Rohlf 1995) to control for Type I error rate. We deemed differences significant at the $p < 0.05$ alpha level.

Spring Dispersal

We censored birds that were captured after the earliest observed covey break-up (17 March for 2013 and 21 March for 2014) ($n = 75$ individuals) because it is possible that dispersal movements had already occurred as well as birds that died or had collar failures during the dispersal period ($n = 162$ individuals) (Liberati and Gates 2012). Movement data were also censored for birds that were lost for >3 consecutive days during the dispersal period ($n = 14$

individuals), or lost permanently (n = 4 individuals). Thus, we monitored spring dispersal movements for 114 (n = 49 in 2013, and n = 65 in 2014) individuals represented by 38 individual coveys. Bobwhites that spent < 50% of their time in any given TSF (12 individuals) were not included in dispersal among TSF analysis. Maximum dispersal distances were calculated by measuring the Euclidean distance between individual bird locations at the last measured location prior to the onset of covey break-up to the farthest location observed during the dispersal period (i.e., movements prior to nest initiation for birds that nested or movements prior to median nest initiation date for birds that were not observed to nest) using ArcGIS 10.0 (ESRI, Redlands, California, USA). Dispersal distance distributions were non-normal and thus, were log_e transformed prior to analysis (Liberati and Gates 2012). To assess the extent of the study area that was traversed by dispersing bobwhites (i.e., dispersal area), we used the same methodological approach that was used to estimate covey home ranges. Specifically, we used radio-locations obtained during the dispersal period to calculate fixed kernel density estimates (Worton 1987, Worton 1989) and 95% isopleths (ha) using the Geospatial Modeling Environment (Beyer 2014).

We used a full model approach with the inclusion of *a priori* variables (Whittington et al. 2006, Eglington et al. 2008, Mundry and Nunn 2009) for maximum dispersal distance and dispersal area separately. Specifically, we fitted generalized linear mixed models (GLMM) using the “lme” package in program R (Zuur et al. 2009, R Development Core Team 2012) that included covey id as a random effect and TSF, age, sex and year as *a priori* explanatory variables that we considered as factors that may influence bobwhite dispersal distance (i.e., response variable).

Annual Variation

Because of the high inter-annual weather variability in the Southern Great Plains (Joyce et al. 2001) (i.e., between year effects affecting plants, animals, and the landscape), we examined normalized differential vegetation index values (NDVI) in order to further parse out possible variation between years. NDVI is strongly correlated with primary productivity and thus serves as a proxy for vegetation biomass (Hobbs 1995, Pettorelli et al. 2005) and structural configuration (Markon et al. 1995, Pettorelli et al. 2011), which we hypothesized could potentially influence bobwhite space use between years on a landscape (i.e., due to resource or cover availability) subjected to large scale burning. NDVI can also serve as a coarse proxy for weather variation (e.g., drought) in the Great Plains of North America (Wang et al. 2001) especially given that weather patterns are key drivers of primary productivity (Pettorelli et al. 2011; Figure 1). We derived NDVI from a multispectral image (2 m spatial resolution) collected for the study area on cloud free days during June 2012 and 2013. This sampling timeframe corresponded to the growing season prior to each year of the study when bobwhite coveys were monitored and thus provided an index of vegetation and a way to test for possible differences in vegetation between years, especially in the context of drought. We derived NDVI using the red and infrared (NIR) bands calculated as $(\text{NIR} - \text{Red}) / (\text{NIR} + \text{Red})$ (Rouse et al. 1974) with the raster calculator in ArcGIS 10.0 (ESRI, Redlands California). The zonal statistics tool (ArcGIS 10.0; ESRI, Redlands, California) was used to calculate mean NDVI as well as coefficient of variation (CV) of NDVI for each covey home range and dispersal area in each year. We utilized CV of NDVI as an additional vegetation metric for depicting spatial heterogeneity (Wang et al. 2006). Mean differences in NDVI and CV of NDVI were tested using a Welch's t-test ($p < 0.05$ significance level) in program R (Zuur et al. 2009, R Development Core Team 2012).

In addition to assessing vegetation biomass with NDVI, we also investigated fine scale vegetation structure and composition between years and among TSF. Specifically, we sampled vegetation at 98 stratified random points (repeated in each year) during 2012 and 2013 (i.e., respective growing seasons prior to bobwhite monitoring). Points were distributed in proportion to available vegetation types on the study area to provide a proxy of vegetation structure available to bobwhites. The distribution of points was based on a supervised classification and was verified using 215 ground-truthed points. Vegetation sampling was conducted within the growing season (June-August). At each point, we used a modified Daubenmire frame (0.5 X 0.5m) (Daubenmire 1954) to estimate percent bare-ground, litter, woody, herbaceous and forb cover. Vertical visual obstruction was assessed using a Nudds board (Nudds 1977). We used one-way multivariate analysis of variance (Johnson and Wichern 2007) (MANOVA: Pillai's Trace statistic) in program R (Zuur et al. 2009, R Development Core Team 2012) to test mean differences in vegetation characteristics between years and among TSF intervals. We used the Pillai's Trace test given that it has been acknowledged as being comparatively robust (Olson 1976, Quinn and Keough 2002), however, we also assessed results from other tests (i.e., Willks, Roy and Hotelling-Lawley) and found that the Pillai Trace test generally provided the most conservative p-values. Prior to analysis, we used Pearson's correlation tests to assess possible collinearity of vegetation variables. Collinearity was low (Pearson's $r < 0.5$) among all variables except for bare ground and litter cover (-0.65) and thus percent bare ground was excluded from analyses.

To obtain an index of bobwhite abundance between years, we conducted covey counts during autumn 2012 and 2013. Count locations ($n = 35$) were spaced at distance of 800m to avoid double counting (Davis 1979) and to obtain adequate coverage of the study area. Call

count surveys were conducted once at each location during the month of October in each year from 45 minutes prior to sunrise (DeMaso et al. 1992) until 30 minutes after sunrise. The number of coveys, calls/covey and covey location were recorded and sketched. Although covey counts only provide rough estimates of bobwhite density (DeMaso et al. 1992), they offered an approximate index of changes in bobwhite abundance from year to year.

Results

Covey Home Range

Overall, bobwhite covey home range size was $76.6 \text{ ha} \pm 5.8 \text{ [SE]}$ and ranged from 12 to 270 ha during both years. Mean covey home ranges were 36% larger in 2013 ($100.54 \text{ ha} \pm 11.83 \text{ [SE]}$) than in 2014 ($63.02 \text{ ha} \pm 5.32 \text{ [SE]}$) ($F_{4, 54} = 4.72$, $p < 0.05$) and differed among TSF categories ($F_{4, 54} = 2.95$, $p < 0.05$). Mean home range size [SE] was $52.07 \text{ ha} \pm 6.62$ in 25-36 TSF compared to $85.00 \text{ ha} [\pm 15.53]$, $86.70 \text{ ha} [\pm 20.66]$, and $78.91 \text{ ha} [\pm 6.54]$ in 0-12, 13-24 and >36 TSF, respectively (Fig. 2), and statistical differences were only found between 25-36 and >36 TSF categories.

Spring Dispersal

Observed spring dispersal distances among individual bobwhites during both years of the study ranged from 227.8 to 7,070.8 m (Table 1), however, 96.5% of observed dispersal movements were $<5,000 \text{ m}$ and 51.8% were $<1,000 \text{ m}$ (Fig. 3). In each year, more birds exhibited dispersal distances of $0-<1,000 \text{ m}$ than for any other distance category comprising 38.8% and 61.5% of dispersal distances in 2013 and 2014, respectively (Fig. 3). GLMM demonstrated that neither TSF, demographic (i.e., age and sex) or year parameters had model support for explaining mean differences in maximum bobwhite dispersal distances. Observed

dispersal area was $204.0 \text{ ha} \pm 30.2 \text{ [SE]}$ (range; 24-741 ha) in 2013 and $112.04 \text{ ha} \pm 19.32 \text{ [SE]}$ (range; 13-906 ha) in 2014. The GLMM of dispersal area indicated no differences among TSF, age, or sex, but did detect differences between years (estimate [SE] = -0.81 (0.28), $p < 0.05$) (i.e., larger mean dispersal areas in 2013 than in 2014).

Annual Variation

The ecological complexities associated with yearly differences are complicated and somewhat unsurprising (Johnson 1999) especially given that our study was conducted on a dynamic landscape with highly variable annual and seasonal weather patterns which are known to drive drastic population fluctuations of our study species (Jackson 1962, Stanford 1972, Roseberry and Klimstra 1984). However, to address potential drivers of the year effects in our study, we summarized several observations and comparisons of landscape/habitat metrics, vegetation structure and composition, as well as bobwhite abundance (Table 2 and Table 3). We found that mean NDVI was less for 2013 than in 2014 (for the growing season prior to covey monitoring; 2012 and 2013) within bobwhite covey home ranges and dispersal areas ($t = -8.18$, $df = 56.62$, $p < 0.001$, and $t = -11.91$, $df = 56.48$, $p < 0.001$, respectively), as was CV of NDVI ($t = -8.39$, $df = 84.95$, $p < 0.001$, and $t = -3.55$, $df = 39.05$, $p < 0.01$, respectively), (Table 2).

Additionally, we found no mean differences in vegetation variables sampled during the growing seasons prior to covey and dispersal monitoring (i.e., percent cover of functional groups and visual obstruction) between years (Pillai's Trace = 0.014, $F_{1, 193} = 0.44$, $p = 0.85$) (Table 3).

However, mean differences in vegetation variables differed among TSF (Pillai's Trace = 0.21, $F_{3, 191} = 2.36$, $p = 0.0013$), in particular, for percent litter cover ($F_{3, 191} = 5.34$, $P = 0.0015$) and visual obstruction ($F_{3, 191} = 3.21$, $p = 0.024$) (Table 3). Specifically, mean percent litter cover (15.83%) and visual obstruction (3.83) were least in 0-12 months post fire treatments and greatest in > 36

months post fire treatments (32.12% and 5.22, respectively) (Table 3). No other vegetation variables exhibited statistical differences at the $p < 0.05$ level.

Although drought persisted throughout much of the study (10 of 12 and 7 of 12 months of at least moderate drought for 2012 and 2013, respectively) (Palmer Drought Severity Index; NOAA 2016), annual growing season and May-July precipitation were greater in 2013 than in 2012 (Table 2). Moreover, there was 3.8 X's less rainfall from May-July in 2012 than 2013 (Table 2), and notably, only 0.25 mm of rainfall was recorded during May 2012. Based on autumn call counts, we also observed that mean coveys/count location was more than 2-fold less in 2012 ($1.51[\pm 0.24]$ coveys/location) than in 2013 ($3.51 [\pm 0.36]$ coveys/location) (Table 2).

Discussion

We found that prescribed fire applied across large portions of the landscape appeared to have a minimal effect on bobwhite space use. Importantly, individuals seemed to be neither attracted too nor repelled by varying TSF occurring on the landscape and we found no difference in spring dispersal movements or area among time since fire treatments. However, prescribed fire affected bobwhite space use of coveys which was driven by smaller home range sizes in 25-36 months' TSF which has also been identified as the time period associated with the recovery (i.e., within 2-4 growing seasons) of shinnery oak to pre-fire structure (McIlvain 1954, Harrell et al. 2001). Overall, the minimal effects of fire on bobwhite behavior that we observed are similar to the observations of Ransom and Schulz (2007) who found that fire had little influence on bobwhite density in Texas.

The mean covey home range size (76.6 ha) that we observed far exceeded those typically attributed to "excellent" (<5 ha) and "good" (<40 ha) bobwhite habitat (Wiseman and Lewis

1981, Dixon et al. 1996, Guthery et al. 2004, Singh et al. 2011), except for the 25-36 TSF treatments (50.2 ha). Larger home range size has often been attributed to limited forage availability and poor habitat quality for numerous species (Zabel et al. 1995, Pasinelli 2000, Moorcroft et al. 2006, Whitaker et al. 2007), including bobwhites (Singh et al. 2011). For example, larger eastern wild turkey (*Meleagris gallopavo*) home ranges have been associated with reduced forage availability resulting from lower site productivity (Wigley et al. 1986). Therefore, one potentially important additional contributor to the large home range sizes and dispersal movements that we observed may be the comparatively low site productivity and highly variable precipitation that is characteristic of more xeric regions in the bobwhite distribution (Spears et al. 1993), such as Western Oklahoma. Similar space use patterns have been observed for female lesser prairie chickens (*Tympanuchus pallidicinctus*); specifically, larger pre-nesting home ranges suggested to be a result of reduced forage availability and increased foraging movements due to drought in shinnery oak communities in New Mexico (Merchant 1982). Thus, similar mechanisms could have driven the comparatively large home range sizes that we observed in our study (also on a shinnery oak landscape), especially in the context of drought.

Our finding that 25-36 TSF was the only variable that influenced bobwhite covey home range size (i.e., generally resulting in smaller mean covey home range size) could have important implications for bobwhite ecology and management relative to fire. Specifically, 25-36 months post fire treatments may have provided the greatest benefits to bobwhite given that the advantages conferred by smaller home ranges can include lower energetic costs associated with movement and lower risk of encountering predators (Powell 2000, Yoder et al. 2004, Whitaker et al. 2007). Although 25-36 TSF could serve as a management target, it is important to note that

burning must occur regularly (e.g., yearly) on the larger contiguous landscape (e.g., study area) in order to consistently have pastures with 25-36 TSF. Thus, the importance of the otherwise minimal effect of TSF on covey home range and dispersal is increased further.

Although our study met our main objective of assessing the influence of prescribed fire on bobwhite movement and space use, and suggested minimal impact, there was variation in bobwhite space use that was beyond the scope of our study. For example, we found that covey home ranges and dispersal areas differed between years. Additionally, dispersal distances did not differ between years but we observed that a smaller percentage of bobwhites traveled <1000 m during spring dispersal in 2013 (38.8%) than in 2014 (61.5%) and it is important to note that 2013 directly followed 2 years of historic drought (NOAA 2016). We observed much lower amounts of annual, growing season, and May-June precipitation in 2012 than in 2013 and these yearly differences were reflected by reduced vegetation biomass (NDVI) and spatial variability of NDVI (CV of NDVI) which were statistically significant. Notably, the number of coveys detected during autumn covey count surveys were more than 2-fold in 2013 than in 2014 and further demonstrates the complexities (e.g., density dependent factors) associated with year-to-year variation in dynamic animal populations on dynamic landscapes. Interestingly, we found no differences in vegetation structure or composition between years which may be due to the dominance of shinnery oak which is a re-sprouting fire-evolved species (Peterson and Boyd 1998). Despite the variability in our study (e.g., drought, population dynamics etc.), we observed a minimal effect of TSF on bobwhites which is notable given that, unlike in the eastern United States, fire has been met with more scrutiny in drier regions as an option for bobwhite habitat management (Kuvlesky et al. 2007).

The extent to which dispersal occurs in bobwhite populations can be linked to the proximity and availability of preferred habitat types which are often the result of management practices (Cook et al. 2009). In this study, bobwhite movement and space use was mostly unimpacted by prescribed burning applied across large portions of the landscape. The lack of differences in bobwhite dispersal movements among all TSF demonstrates either, 1) behavioral flexibility by bobwhites to adjust space use under different landscape configurations (Jackson 1960, Guthery 1999, Guthery 2000), 2) similarities in resources and cover among TSF relative to the habitat needs of bobwhites, or 3) a combination of both given that bobwhites have evolved in grassland and shrubland ecosystems prone to disturbance. In shinnery oak communities, fire is acknowledged as a primary historical disturbance mechanism and shinnery oak exhibits rapid resprouting and fast structural recovery following fire (McIlvain 1954, Harrell et al. 2001). Therefore, the fire tolerance and recovery of this native clonal plant species (Boyd and Bidwell 2001, Harrell et al. 2001) could have contributed the lack of differences in bobwhite space use that we observed if stand replacement mediated the ability of birds to obtain cover or resources over time (i.e., months, years) even in the face of drought conditions. Moreover, we found no differences in vegetation composition or structure between years based on sampling and only differences in percent litter cover and visual obstruction among TSF. These observations are particularly important given that woody cover has been shown to be a critical component of non-breeding bobwhite habitat (Wiseman and Lewis 1981, Roseberry and Klimstra 1984, Hiller et al. 2007) and can buffer bobwhite populations from drought effects (Parent et al. 2016).

It is well understood that the long term conservation of rangelands, as well as the species that inhabit them, can be achieved partly through the use of prescribed fire to prevent the encroachment of woody species (Fuhlendorf and Engle 2004, Engle et al. 2008). However,

uncertainty regarding the usefulness of prescribed fire for both reducing woody encroachment and promoting bobwhite habitat has been a long-standing question and management concern in the western fringe of the bobwhite distribution (Renwald et al. 1978, Carter et al. 2002, Ransom and Schulz 2007). Our findings shows that prescribed fire applied at a landscape scale had limited influence on short-term bobwhite behavior and space use in a shinnery oak system. Moreover, even in the case of the most recent burns (0 - 12 months post fire), significant departures of birds from the treatment areas or differences in dispersal movements were not observed.

Implications

Although attempting to burn at the smallest feasible scale may be the best approach for improving habitat quality for bobwhites in some landscapes (Ransom and Schulz 2007, Wellendorf and Palmer 2009), many small burns may be more logistically difficult to achieve than fewer large burns. These findings suggest that land managers in shinnery oak vegetation communities can utilize prescribed fire across large spatial extents without substantially altering the space use or movement of bobwhites.

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687 **Table 1.** Means and ranges of dispersal movements ($\pm$ SE) of individual radio-marked northern
 688 bobwhites (n = 114) at the Packsaddle Wildlife Management Area in western Oklahoma, USA,
 689 2013 – 2014.

Year	Sex	N	Dispersal Distance (m)	Dispersal Range (m)
2013	Male	22	1,636.0 ($\pm$ 242.2)	483.6 – 4,420.5
	Female	27	1,863.1 ($\pm$ 295)	425.3 – 6,421.4
2014	Male	32	1,273.6 ($\pm$ 206.2)	350.8 – 4,705.3
	Female	32	1,694.7 ($\pm$ 320.4)	227.83 – 7,070.8
Pooled	Male	54	1,421.3 ($\pm$ 157.5)	350.8 – 4,705.3
	Female	60	1,771.8 ($\pm$ 218.5)	227.83 – 7,070.8

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699 **Table 2.** Summary of differences in abiotic and biotic factors observed at the Packsaddle
700 Wildlife Management Area in western Oklahoma, USA. Precipitation values represent those
701 recorded at a nearby weather station (i.e., <30km) during the year and months prior to bobwhite
702 monitoring.

Abiotic or biotic factor	2012	2013
Annual precipitation (mm)	425.90	531.90
Growing season (April –September) precipitation (mm)	196.09	381.25
May-July precipitation (mm)	41.65	159.76
Mean NDVI on study area [SE] ^A	0.28 [±0.06]	0.36 [±0.13]
CV of NDVI (%) ^B	22	35
Mean NDVI [SE] within covey home ranges	0.28 [±0.005]	0.36 [±0.008]
CV of NDVI [SE] within dispersal areas	0.19 [±0.006]	0.32 [±0.008]
Coveys per count location [SE]	1.51 [0.24]	3.51 [0.36]

703 ^A Mean of normalized differential vegetation index (NDVI).

704 ^B Percent coefficient of variation (CV) of NDVI.

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709 **Table 3.** Summary of means [SE] for vegetation variables collected at 98 random stratified
 710 points at the Packsaddle Wildlife Management Area in western Oklahoma, USA, 2012 – 2014.
 711 Vegetation data were collected during the growing season prior to bobwhite monitoring.

	Percent Cover ($\pm$ SE)					
Time Since Fire (months)	Bare	Litter	Grass	Forb	Woody	Visual Obstruction
0-12 months	55.83 (± 4.62)	15.83 (± 2.55)	19.60 (± 2.15)	17.75 (± 3.03)	18.50 (± 4.70)	3.83 (± 0.38)
13-24	48.41 (± 6.28)	17.84 (± 3.03)	26.60 (± 3.42)	21.82 (± 4.06)	18.75 (± 5.61)	4.24 (± 0.46)
25-36	44.86 (± 6.23)	32.36 (± 6.34)	28.33 (± 4.04)	17.22 (± 3.56)	21.94 (± 7.02)	4.04 (± 0.43)
≥ 36	39.15 (± 2.82)	32.12 (± 2.32)	26.65 (2.23)	18.41 (± 1.44)	14.54 (± 2.08)	5.22 (± 0.26)

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726 **Figure Legend**

727 **Figure 1.** Map of Packsaddle Wildlife Management located in Western Oklahoma, USA. The
728 bolded boundary line indicates the focal area scheduled to receive prescribed fire treatments
729 prior to and during northern bobwhite monitoring (2012 - 2014). Different time since fire is
730 depicted by filled circles (2012 burns), horizontal lines (2013 burns) and vertical lines (2014
731 burns).

732 **Figure 2.** Mean covey home ranges ($\pm$ SE) (n=59) of northern bobwhites among 0-12 (n=6), 13-
733 24 (n=6), 25-36 (n=15), and >36 (n=32) time since fire (TSF) at the Packsaddle Wildlife
734 Management Area in western Oklahoma, USA, 2013 – 2014.

735 **Figure 3.** Frequency distribution of distance traveled by radio-marked northern bobwhites from
736 covey break-up through nest initiation during the spring dispersal period at the Packsaddle
737 Wildlife Management Area in western Oklahoma, USA 2013 – 2014.

Figure 1

Figure 2

Figure 3