



## Research Article

# An Evaluation of Northern Bobwhite Translocation to Restore Populations

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**ABSTRACT** Northern bobwhite (*Colinus virginianus*) populations have been declining in Texas during the last 2 decades. This decline also is occurring in areas with apparently ample habitat such as the Rolling Plains of Texas, USA. The goal of our study was to examine the efficacy of translocating wild-trapped bobwhites into recently depopulated areas in the eastern Rolling Plains as a method of restoring population size. Our objectives were to document survival (spring-summer and annual), reproductive efforts, and site fidelity of translocated bobwhites and compare relative abundance of bobwhites between release and control sites to evaluate the efficacy of translocation. We translocated 409 wild bobwhites ( $n = 186$  radio-marked F) to 2 sites in Shackelford and Stephens counties, Texas, during March 2013 and 2014. Spring-summer (Mar–Sep) survival ranged between 0.32 and 0.38 ( $n = 186$  bobwhites), and annual survival ranged between 0.19 and 0.23 ( $n = 186$  bobwhites) during 2013–2015. Seventy-four percent of translocated females that entered the nesting season (i.e., May 1;  $n = 112$  bobwhites) produced a nest, resulting in 125 nests with an apparent nest success of 46.1% and a nesting rate of  $1.1 \pm 0.1$  (SE) nests/female. Site fidelity post-translocation was moderate to high; 67.7% ( $n = 44$  bobwhites) and 85.1% ( $n = 40$  bobwhites) of translocated bobwhites did not disperse  $>2$  km from their release point during the summers of 2013 and 2014, respectively. Despite this strong demographic performance, relative abundance of bobwhites did not differ between the release and control sites after 2 years as indexed by covey-call counts ( $2.7 \pm 0.6$  vs.  $3.0 \pm 1.2$  coveys/point, respectively;  $P = 0.71$ ) and helicopter surveys ( $0.9 \pm 0.2$  vs.  $0.7 \pm 0.2$  bobwhites/km, respectively;  $P = 0.64$ ). Thus, translocation of bobwhites failed to increase the bobwhite population beyond that of the control during this study. Future translocation research should aim to increase translocation success by investigating methods for increasing survival during the 1-month period following translocation (e.g., soft-release, vitamin E and selenium injections) and for improving site fidelity using conspecific attraction. Future research should incorporate genetic measures to assess genetic contribution of translocated individuals to improve evaluation of translocation success. © 2017 The Wildlife Society.

**KEY WORDS** *Colinus virginianus*, northern bobwhite, quail, relative abundance, reproduction, Rolling Plains, survival, translocation.

Northern bobwhite (*Colinus virginianus*) populations have been declining throughout their range for decades according to every large-scale population index (Brennan 1991, National Audubon Society 2010, Hernández et al. 2013, Sauer et al. 2014). According to the Breeding Bird Survey (BBS), bobwhites have been declining 4% annually in the United States during 1966–2013, and the rate of decline has increased in the most recent decade (Sauer et al. 2014). No

single factor can explain completely the decrease of bobwhites across their range, but habitat loss and fragmentation have been proposed as ultimate causes (Brennan 1991, Church et al. 1993, Hernández et al. 2013). Proximate factors potentially contributing to this decline may include the replacement of native plant communities with non-native vegetation, drought, predation, and disease (Brennan 1991, Nedbal et al. 1997, Rollins and Carroll 2001, Peterson 2007, Hernández et al. 2013).

Historically, bobwhite populations were relatively abundant in Texas, which Brennan (2007) attributed to contiguous habitat in the southern, western, and northern areas of the state. However, roadside count data during 1978–2013 from the Texas Parks and Wildlife Department (TPWD) recorded the 5 lowest mean numbers of quail/route for the Rolling

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Plains of Texas during 2009–2013 (TPWD 2013). This decline has been especially disconcerting to landowners, biologists, and quail hunters in the Rolling Plains because it has been recent (since 2006) and abrupt, and the Rolling Plains ecoregion has supported some of the best bobwhite hunting opportunities in the past (Rollins 2007). Although fragmentation due to urbanization may account for population reduction in the eastern regions of Texas (e.g., Pineywoods, Cross Timbers, and Prairies ecoregions), it may not be a factor limiting bobwhite populations in the Rolling Plains where fragmentation is less prominent (Wilkins et al. 2003).

Managing land to increase connectivity among bobwhite populations is an obvious goal in fragmented landscapes. What is less clear, however, is how to enhance low populations where land is apparently not fragmented and where habitat has been managed extensively for bobwhites (Brennan 1991). Brennan (1991) proposed that these circumstances provide a perfect opportunity to test the suitability of a given area using translocated, wild bobwhites.

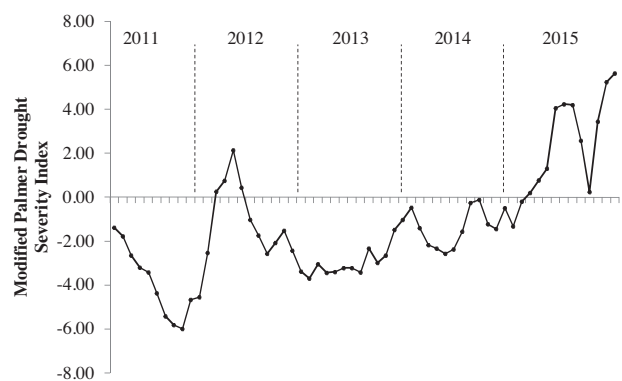
The translocation of wild bobwhites has received limited research attention, and the few studies that have been conducted have yielded contrasting results (Liu et al. 2000, Terhune et al. 2010, Scott et al. 2013). For example, Scott et al. (2013) reported lower survival and productivity of translocated bobwhites in central Texas compared to resident bobwhites. Conversely, other studies have reported similar survival and productivity between translocated and resident bobwhites (Liu et al. 2000; Terhune et al. 2006a, 2010). In addition, Terhune et al. (2006a) documented a population response 1 year after bobwhites were translocated in Georgia, whereas Scott et al. (2013) documented no population response to bobwhite translocation up to 2.5 years post-translocation. A collective review of these studies suggests that the likelihood of translocation success for bobwhites depends on a variety of factors such as distance translocated, holding time (i.e., length of time bobwhites were held after being trapped and prior to release), and habitat-connectivity surrounding release sites (Scott et al. 2013).

The continued, continental decline of bobwhites (Brennan 1991, Hernández et al. 2013, Sauer et al. 2014) warrants evaluation of habitat- and population-restoration techniques designed to ameliorate the species' decline. Griffith et al. (1989) recommended the development of successful restoration techniques in advance of the disappearance or decline of a species in an effort to ensure the availability of effective techniques prior to their necessity. Therefore, in an effort to contribute to future bobwhite restoration methods, our research goal was to examine the efficacy of translocating wild-trapped bobwhites to large (>2,500 ha), well-managed areas in the Rolling Plains of Texas that had recently (since 2006) experienced population declines (TPWD 2013). Specifically, our objectives were to document survival (spring-summer and annual), reproductive effort, and site fidelity of wild, translocated bobwhites, and compare relative abundance of bobwhites between release and control sites at 2 years post-release to evaluate the efficacy of translocation.

## STUDY AREA

We conducted this study during 2013–2015 on 2 privately owned ranches: a release ranch (2,960 ha) where we released bobwhites at 2 sites (250 ha and 400 ha) and a control ranch (1,855 ha) where we did not release bobwhites but collected data on 1 site (400 ha). These ranches are located on the Shackelford and Stephens county line, which lies on the eastern fringe of the Rolling Plains ecoregion of Texas, USA (Gould 1975). This ecoregion contains a mosaic of rangeland and cropland, with the majority (>65%) used for cattle grazing as cow-calf operations (Correll and Johnston 1979, Scifres 1980:35). The Rolling Plains in general, and the study counties specifically, are comprised of relatively intact rangeland. Rangeland comprises 94% of the area of both Shackelford (237,167 ha) and Stephens (238,716 ha) counties (Natural Resources Conservation Service [NRCS] 2014). Mean rangeland-patch area is  $989.7 \pm 716.9$  (SE) ha in Shackelford County and  $1,178.2 \pm 740.3$  ha in Stephens County. Mean nearest neighbor distance between rangeland patches is  $46.5 \pm 3.3$  m in Shackelford County and  $37.4 \pm 2.6$  m in Stephens County. For comparison, the rangeland in Fisher County in the Rolling Plains comprises only 68% of the county, yet is similar in area (233,875 ha) to Shackelford and Stephens counties. In addition, mean rangeland-patch area is considerably lower ( $156.2 \pm 85.3$  ha), and mean nearest neighbor distance between rangeland patches is larger ( $74.7 \pm 4.5$  m), in Fisher County compared to Shackelford and Stephens counties.

Mean annual rainfall in the Rolling Plains ecoregion ranges from 55 cm to 75 cm, and the growing-season length consists of 185–235 frost-free days (Correll and Johnston 1979). During our study and the 2 years prior to it (2011–2014), the study ranches averaged 16–37 cm less rainfall than the 30-year mean annual precipitation for Stephens County, Texas (72 cm; 1981–2010; Northwest Alliance for Computational Science & Engineering 2015). The study ranches therefore experienced drought conditions during the 2 years prior to (2011–2012) and during (2013–2014) this study (Fig. 1).



**Figure 1.** Modified Palmer Drought Severity Index data for the Rolling Plains of Texas, USA, 2011–2015. Data are from the National Climate Data Center.

Predominant soil types on the release ranch included Truce fine sandy loam, Bluegrove loam, Bonti-Exray complex, and Owens-Harpersville complex soils (NRCS 2013a). The majority of the area on which our study occurred comprised sandy loam soils, which supported 1,350–4,500 kg/ha total dry-weight herbaceous forage production during normal years (NRCS 2013a). Predominate soil types on the control ranch included Thurder clay loam, Bluegrove flaggy fine sandy loam, Hensly stony clay loam, and Lindy silt loam (NRCS 2013a). Dry-weight forage production on these soil types ranged from 2,240 kg/ha to 5,600 kg/ha during a normal year (NRCS 2013a). Wildlife management on both ranches was extensive rather than intensive. Mesomammals were killed opportunistically if encountered. In addition, neither ranch had an intensive wildlife feeding program, but timed spin-cast feeders (dispensing corn) were used to bait deer for hunting (control site had 1 feeder, release site 1 had 2 feeders, release site 2 did not have feeders). Wildlife habitat was managed via grazing management. Thus, grazing was ceased on the release ranch in March 2011 and on the control ranch in April 2013 to maintain existing ground cover and avoid grazing during drought.

The woody vegetation community at both ranches was dominated by honey mesquite (*Prosopis glandulosa*), live oak (*Quercus virginiana*), gum bumelia (*Sideroxylon lanuginosum*), lotebush (*Ziziphus obtusifolia*), algerita (*Mabonia trifoliolata*), tasajillo (*Cylindropuntia leptocaulis*), common prickly ash (*Zanthoxylum americanum*), netleaf hackberry (*Celtis reticulata*), and prickly pear species (*Opuntia* spp.; plant nomenclature follows NRCS 2013b). Common grasses included little bluestem (*Schizachyrium scoparium*), sideoats grama (*Bouteloua curtipendula*), King Ranch bluestem (*Bothriochloa ischaemum*), and Texas wintergrass (*Nassella leucotricha*; NRCS 2013b). The forb community included various crotons (*Croton* spp.), western ragweed (*Ambrosia psilostachya*), snow-on-the-mountain (*Euphorbia marginata*), and Indian blanket (*Gaillardia pulchella*).

Historically, this ecoregion supported some of the best bobwhite hunting in Texas, but local reports and TPWD roadside count data indicate bobwhite numbers had been low since 2006 (Rollins 2007, TPWD 2013). Hunting reports for 1 of our study sites on the release ranch contained a minimum of 56 coveys (11.6 ha/covey) during 1985 (J. R. Jones, release ranch landowner, personal communication); however, quail hunts ceased during the 1990s because of low quail encounters. The control ranch also supported productive bobwhite populations (25 covey encounters/half-day hunt) until 1998, when bobwhites began a noticeable decline (A.V. Jones, control ranch landowner, personal communication). Quail hunts were discontinued at this ranch after 2003 because of low bobwhite encounters. We attempted to trap resident bobwhites at the release ranch prior to translocation during February 2013 (7 days;  $n = 30$  traps), but our efforts were unsuccessful and no resident quail were trapped. In addition, we did not detect bobwhites on the control site during pre-treatment surveys. Thus, both the release and control site did not possess a viable resident bobwhite population.

## METHODS

This study was a completely randomized design with 2 treatments and repeated measures. The treatments were translocation of bobwhites or no translocation of bobwhites. There were 3 experimental units, 2 of which received treatment and 1 served as an experimental control. Thus, we estimated the experimental error from the translocation treatment (Steel and Torrie 1980:147). The release ranch contained 2 treatment experimental units, which served as release sites for translocated bobwhites. Release site 1 (R1; 400 ha) was located 2 km north of release site 2 (R2; 250 ha). The control ranch contained a control site (C; 400 ha), which was located about 21 km south of the release sites. All 3 sites (R1, R2, and C) were managed similarly. We measured bobwhite relative abundance on all 3 sites pre-treatment (spring 2013), during treatment (autumn 2013, spring 2014, autumn 2014), and post-treatment (spring 2015;  $n = 5$  time periods). These time periods comprised the repeated-measures component of the design.

### Habitat Evaluation

We conducted vegetation surveys on the 3 sites during May–August 2014 to evaluate the suitability of the sites for bobwhites. Vegetation surveys assessed the primary components of bobwhite habitat (i.e., bare ground, herbaceous cover, nesting cover, and woody cover) and focused on vegetation characteristics that are diagnostic of bobwhite habitat in the Rolling Plains. We assessed habitat using 8 variables: bare ground (% bare ground, % litter), herbaceous cover (% herbaceous cover, radius of obstruction), nesting cover (bunchgrass density, prickly pear density), and woody cover (% brush cover, cone of vulnerability; Rice et al. 1993, Kopp et al. 1998, Hernández et al. 2003). To uniformly sample vegetation across the sites, we created a 346 × 346-m grid using ArcMap 10.1 (Environmental Systems Research Institute, Redlands, CA, USA) and randomly overlaid this grid on the study sites. We then identified the center of each grid cell that was within the sites' boundaries to serve as a vegetation sampling point (R1:  $n = 30$  points, R2:  $n = 20$  points, and C:  $n = 30$  points). We sampled about 8 points/week during 30 May–12 August 2014 and alternated surveying days between sites to capture possible changes in plant phenology through time.

At each point, we established 4, 15-m transects radiating in the 4 cardinal directions. We used the line-intercept method to measure percent canopy cover of woody plants (Canfield 1941), and the point-center quarter method as modified by Mitchell (2007) to estimate the density of bunchgrass and prickly pear. We defined bunchgrasses as clumps of grass that were large enough (20 cm wide × 20 cm tall) to serve as a suitable bobwhite nesting site (Lehmann 1984, Hernández et al. 2003). We measured the distance from the survey point to the closest suitable bunchgrass and prickly pear in each of the 4 quarters (northeast, southeast, southwest, and northwest). In addition, we visually estimated percent exposure of bare ground, herbaceous cover (forbs and grass), and litter cover using a 20 × 50-cm Daubenmire frame centered at the 5-, 10-, and 15-m mark along each of the 4

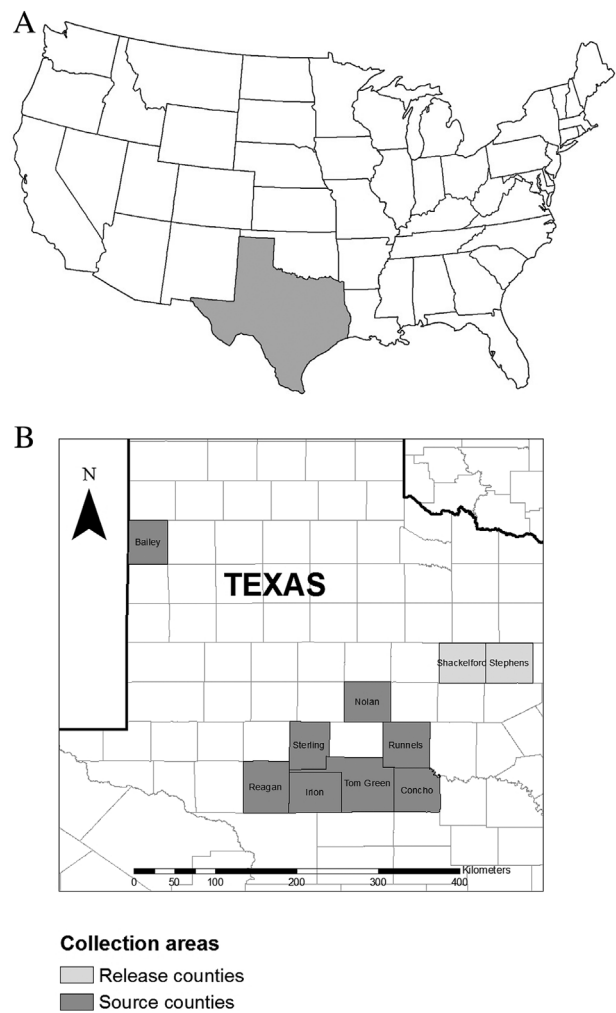
transects (Daubenmire 1959). We measured radius of obstruction (horizontal cover) at 4 cardinal and 4 ordinal directions by placing a 2.5-cm-diameter  $\times$  2-m-long pole at the central survey point, and then incrementally backing away until the lower most strata (0–15 cm) was 100% visually obstructed from 1 m in height (Kopp et al. 1998). We averaged the 8 distances for each point to calculate the mean distance of obstruction (Kopp et al. 1998). We determined angle of obstruction and cone of vulnerability (vertical cover) by placing a 2-m pole at the central survey point and then directing it to the top of the vegetation structure causing the most vertical obstruction within 50 m along the 4 cardinal and 4 ordinal directions (Kopp et al. 1998). We used a clinometer to measure the angle between the ground and the pole (i.e., the angle of obstruction) and calculated the mean angle of obstruction for each point.

### Trapping and Translocation

Our goal was to trap and translocate 200 wild bobwhites/year during March 2013 and 2014 to maximize the number of females released just prior to the beginning of the breeding season (Apr–Aug). We trapped bobwhites on 9 privately owned and 2 publicly owned properties that volunteered to donate coveys. Donor ranches donated 6–69 wild bobwhites/year for translocation. Donor ranches were located 54–360 km from the release sites in Bailey, Concho, Irion, Nolan, Reagan, Runnels, Shackelford, Sterling, and Tom Green counties (Fig. 2).

Prior to trapping, we baited trap sites weekly with milo during January–March 2013 and 2014. We used 2 teams (2 people/team) to trap wild bobwhites. Teams used baited funnel traps covered with natural vegetation to provide protective cover (Stoddard 1931:442) and trapped 2–4 days on a given donor ranch ( $n \approx 30$  traps/ranch). We placed traps in suspected loafing areas, where bobwhites had been seen prior, or near existing deer feeders. We checked traps once around mid-morning and once at dusk. We banded and weighed captured bobwhites and classified them by age and sex. We collected a tissue sample (4–6 breast feathers) from each individual for possible future DNA analysis to investigate the impact of translocation on the resident population. We fitted all females weighing  $\geq 150$  g with a 6-g necklace-style radio-transmitter with a mortality sensor (American Wildlife Enterprises, Monticello, FL, USA). Because translocation success is contingent upon successful reproduction, we radio-marked only females to more accurately capture reproductive potential. However, we banded all trapped bobwhites (both M and F) with individually numbered leg bands.

We translocated trapped bobwhites to either R1 or R2 during March 2013 and 2014. We released bobwhites caught in the same trap together as a covey. We coalesced partial coveys ( $< 6$  bobwhites) with other partial coveys to form a complete covey ( $\geq 6$  bobwhites). We held coveys initially in breathable (i.e., cotton) pillowcases and then placed them in transport cages just prior to translocation (G.Q.F. Manufacturing Company, Savannah, GA, USA). We released bobwhites trapped mid-morning that same afternoon, whereas we released bobwhites trapped at dusk the following afternoon. Therefore, we held



**Figure 2.** Location of (A) Texas, USA and (B) distribution of counties that served as source populations for northern bobwhites and release sites in Texas, USA, during March 2013–2014. We trapped wild bobwhites within source counties and then translocated them to release counties. We trapped 4 bobwhites on the western edge of Shackelford County and translocated them to the eastern edge during 2013.

some bobwhites for about 8 hours and others for 20 hours, but we did not hold birds  $> 24$  hours. We used the same grids established for vegetation sampling as release points for translocated coveys. Grid cells represented 12-ha blocks (the approximate home range size of a covey; Dixon et al. 1996) whose centers served as release points for coveys ( $n = 50$  release points for R1;  $n = 30$  release points for R2). We released the first translocated covey at the release point closest to the center of each grid (i.e., release site). Thereafter, we selected release points by spiraling clockwise out from the center of the release site. This resulted in bobwhites being released in the core interior of the release site to minimize the probability of translocated bobwhites dispersing off site. We alternated covey releases between R1 and R2. We released translocated coveys using a hard-release technique (i.e., bobwhites released immediately after arrival to release site). We opened the door of the transport cage and released bobwhites. We conducted this study under the approval of Texas Parks and Wildlife Department (Scientific permit no. SPR-0690–152)

and the Agriculture Animal Care and Use Committee (AACUC) at Texas A&M University (AUP no. 2013–004A).

### Radio-Telemetry

We monitored translocated, radio-marked female bobwhites year-round. We classified the monitoring of radio-marked females as location or detection. A location occurred when a researcher homed in on radio-marked females and obtained global positioning system (GPS) coordinates using a Trimble Juno (Trimble Navigation Limited, Sunnyvale, CA, USA) equipped with CyberTracker software (CyberTracker, Cape Town, Africa). A detection occurred when a researcher simply verified the status (i.e., alive or dead) and general whereabouts of a radio-marked female without homing. The incorporation of detections was necessary to monitor radio-marked bobwhites during a period when time and manpower was limited because trapping, processing, and translocation were still in progress across a large spatial extent within a restricted trapping deadline. We used detections to monitor survival of bobwhites but not to estimate bobwhite dispersal. Monitoring for radio-marked females began within 24 hours of release, but monitoring intensity varied throughout the year. During 1 March–31 March, radio-marked females were located 2 times/week and detected on all non-location days. As trapping concluded and the reproductive season approached, monitoring intensity increased, and radio-marked females were located 3–4 times/week and detected 2–3 times/week during 1 April–15 August. Monitoring intensity subsequently decreased when the reproductive season terminated, and bobwhites were located only 1–2 times/week during 16 August–28 February.

When we detected a mortality signal, we located and retrieved the transmitter and determined possible cause of detachment (mortality or collar failure). We classified causes of mortalities as avian, mammalian, snake, unknown predation, or unknown death in accordance with Carter et al. (2002). We considered radio-marked females to be nesting when they were located at the same location on  $\geq 2$  consecutive tracking dates. We obtained a GPS location within 15 m of the nest and recorded vegetation characteristics to aid in future location of the nest. We monitored nests from afar ( $\geq 50$  m from the nest) 3–4 times per week. Once a clutch hatched or failed, we recorded nest fate as successful, depredated, or abandoned in accordance with Burger et al. (1995). We recorded clutch size when possible for all nests, and egg hatchability (i.e., no. eggs that hatched) for successful nests.

### Population Response

*Covey-call counts.*—We estimated bobwhite relative abundance using covey-call counts during early spring (Feb–Mar 2013–2015) and autumn (Oct–Nov 2013–2014). A team of 3–8 people collected data. In an attempt to minimize observer variability (Rusk et al. 2009), we trained all covey-call count team members prior to conducting counts, and used the same researchers each year when possible. We conducted covey-call counts simultaneously at the release and control sites on days when sufficient observers were present. On days when observers were limited, covey-call

counts alternated between study sites (e.g., release sites on day 1, control site on day 2, release sites on day 3). We completed the entire covey-call count survey within a 3-week period.

We carried out covey-call counts at permanent survey points that we established throughout each site ( $n=7$  points at R1,  $n=3$  points at R2,  $n=7$  points at C). All survey points were at least 0.8 km apart and occurred at a density of approximately 1/55 ha. Observers arrived at survey points 45 minutes before sunrise. The survey began when observers heard the first bobwhite covey call and continued for the next 20 minutes (DeMaso et al. 1992). If observers did not hear calls, they listened until sunrise. Observers recorded the number of different coveys heard calling and the approximate location of coveys. We surveyed each point twice per time period. We did not repeat surveys within 3 days of each other, and the same observer did not survey the same survey point within a given time period. We estimated bobwhite relative abundance (no. coveys heard/point) for each time period for each site. We calculated estimates using the greatest count of number of coveys heard at each point during each time period because this value represented the minimum, known number of coveys at a given point.

*Helicopter counts.*—We also estimated bobwhite relative abundance using helicopter surveys (Rusk et al. 2007) conducted during spring (Mar 2013–2015) and autumn (Nov 2013–2014). We used a geographic information system to establish transects spaced 200 m apart on each site and the surrounding area. We oriented transects north-to-south and varied their length (0.17–5.19 km) to keep transects within ranch boundaries. This resulted in 14 transects (39 km) for R1, 17 transects (30 km) for R2, and 23 transects (68 km) for the control site. Survey protocol involved flying a third of the transects at a given ranch, flying to the other ranch to survey a third of the transects, returning to the first ranch to survey the next third of transects, and repeating the process until all transects had been surveyed. We completed the entire helicopter survey in 1 day.

Surveys involved 1 pilot and 3 observers. We conducted surveys using a Robinson R44 helicopter (Robinson Helicopter Company, Torrance, CA, USA) and traversed along a transect 7–10 m above the ground at a speed of 40 kph (Rusk et al. 2007). When a bobwhite detection occurred, the helicopter hovered momentarily, and researchers recorded the number of bobwhites observed and the location of the helicopter using a Trimble Juno equipped with CyberTracker software. We calculated bobwhite relative abundance (no. bobwhites observed/km) for each transect during each time period.

### Statistical Analysis

We evaluated bobwhite habitat within each site. We calculated bunchgrass density (clumps/ha) and prickly pear density (plants/ha) for each site using the point-center quarter method as modified by Mitchell (2007). We calculated mean percent bare ground exposed; percent litter, herbaceous cover, and woody cover; radius (m) and angle ( $^{\circ}$ ) of obstruction; and cone of vulnerability ( $m^3$ ) for each survey

point, and we then averaged these metrics across sites. We evaluated the suitability of bobwhite habitat at a particular site by comparing the study-site estimates of these variables to published, habitat-suitability bounds of bobwhites (Lehmann 1984, Guthery 1986, Kopp et al. 1998, Arredondo et al. 2007). We considered a given habitat variable to have met bobwhite needs if its value fell within the reported habitat-suitability bounds.

We estimated spring-summer (Mar–Sep 2013–2014) and annual (Mar–Feb 2013–2015) survival rates for radio-marked, translocated female bobwhites. The spring-summer interval was the 183 days following the release of translocated bobwhites (9 Mar 2013–6 Sep 2013 and 8 Mar 2014–5 Sept 2014). We defined annual survival as the 365 days following the release of translocated bobwhites (9 Mar 2013–8 Mar 2014 and 8 Mar 2014–7 Mar 2015). Similar to Terhune et al. (2006a), we did not eliminate translocated bobwhites that died during the traditional censor periods (i.e.,  $\leq 14$  days; Pollock et al. 1989, Cox et al. 2004) from analysis because these mortalities could have been related to translocation. We used bobwhites lost to dispersal or radio failure to estimate survival up to the day they went missing or the radio detached, upon which time we censored the individuals from the analysis (Pollock et al. 1989, Hernández et al. 2005).

Specifically, we calculated spring-summer and annual survival distributions of translocated female bobwhites using the staggered-entry Kaplan–Meier estimator (Pollock et al. 1989) in program STAGKAM (Kulowiec 1989) and compared them between sites, years, and age classes using a log-rank chi-square test (Pollock et al. 1989). We also used the known-fate model with a logit link function in Program MARK (version 7.2) to assess factors possibly affecting weekly survival during the spring-summer interval (Mar–Sep; White and Burnham 1999). We modeled survival of translocated bobwhites as a function of 6 variables (yr, release site, distance translocated, body mass at capture, length of holding time, and age; Table 1). We constructed 30 biologically relevant, *a priori* defined models and used Akaike’s Information Criterion corrected for small sample size ( $AIC_c$ ) to rank them (Burnham and Anderson 2002). We evaluated the relative weight of evidence for each model using Akaike weights ( $w_i$ ), in which the greatest weight signified the most supported model in the set of models constructed.

**Table 1.** *A priori* survival models for radio-marked, translocated female northern bobwhites in Stephens and Shackelford counties, Texas, USA, March–September 2013–2014.

| Model        | Biological explanation                                       |
|--------------|--|
| S.           | Constant survival  |
| T            | Survival varies linearly through time                        |
| Year         | Survival varies between 2013 and 2014                        |
| Release site | Survival varies between release sites (R1 and R2)            |
| Mass         | Survival varies with body mass (g) at time of capture        |
| Distance     | Survival varies with distance (km) bobwhite was translocated |
| Holding time | Survival varies with amount of time bobwhite was held        |
| Age          | Survival varies between juveniles and adults                 |

We calculated apparent nest success, clutch size, and egg hatchability. We also calculated potential and realized estimates of percentage of nesting females and nesting rate (no. nests/F; Hernández et al. 2005, Scott et al. 2013). We calculated potential estimates using only translocated radio-marked females that survived the entire nesting season (1 May–31 Aug) and therefore the estimates captured the entire nesting history of females. Potential estimates represented the complete reproductive potential of a translocated bobwhite. Conversely, realized estimates were based on translocated radio-marked females that were alive at the beginning of the nesting season (1 May) regardless of subsequent fate. Realized estimates included bobwhites that died during the nesting season before having an opportunity to nest or re-nest and therefore were lower than potential estimates. We evaluated the cumulative effect of reproductive variables (realized % of nesting F, realized nesting rate, nest success, clutch size, egg hatchability) by multiplying these variables to estimate the number of chicks produced by females each year. We used Fisher’s exact test to compare nesting effort and nest success between years (Siegel 1956). We compared mean nesting rate, mean clutch size, and mean egg hatchability between years using a 1-way analysis of variance (PROC ANOVA; SAS 9.2). We conducted statistical analyses using SAS 9.2 (SAS Institute. Cary, NC, USA).

We estimated site fidelity of radio-marked, translocated female bobwhites during spring-summer (Mar–Aug) based on the distance traveled from release point to the farthest observed radio-telemetry location. We did not calculate distances for individuals that survived  $< 30$  days post translocation or had  $< 20$  locations so the analysis would consist only of individuals that had sufficient time to disperse. We also excluded mortality locations from the analysis to remove any potential bias in distance traveled because of predators moving carcasses. We classified translocated bobwhites as dispersers or non-dispersers based on distance traveled. A disperser was a bobwhite that traveled  $\geq 2$  km from its release point, whereas a non-disperser was a bobwhite that remained within 1 km from its release point (Townsend et al. 2003, Scott et al 2013). We used relative cumulative frequency distributions of distance traveled to quantify the percent of bobwhites that had traveled a given distance.

We analyzed relative abundance of bobwhites based on covey-call counts and helicopter surveys separately using SAS 9.2. We averaged samples (points or transects) in each site for each treatment and time period using the means procedure in SAS (PROC MEANS; SAS 9.2). We used a generalized linear model procedure (PROC GLM; SAS 9.2) to estimate type I means and associated standard errors and to test the hypotheses that mean relative abundance ( $\bar{x}$  max. call counts or  $\bar{x}$  bobwhites/km) did not differ between treatments and were not affected by a time  $\times$  treatment interaction. We analyzed differences in means using a repeated measures analysis of variance (PROC ANOVA; SAS 9.2), and evaluated means that differed significantly using the protected least significant difference (LSD) test

**Table 2.** Estimates of habitat variables diagnostic of northern bobwhite habitat at control (C) and release sites (R1 and R2) in Stephens and Shackelford counties, Texas, USA, May–August 2014. We released translocated bobwhites at the release sites (R1 and R2).

| Variable                                | Site     |        |          |        |           |        | Habitat suitability |  |
|---|----------|--------|----------|--------|-----------|--------|---------------------|--|
|   | C        |        | R1       |        | R2        |        | Bounds              | Reference  |
|   | Estimate | SE     | Estimate | SE     | Estimate  | SE     |                     |  |
| Sampling plots ( <i>n</i> )             | 30       |        | 30       |        | 20        |        |                     |  |
| Bunchgrass density (clumps/ha)          | 202      | 18.6   | 238      | 21.7   | 395       | 45.7   | >618                | Lehmann (1984:62), Guthery (1986:2), Arredondo et al. (2007) |
| Prickly pear density (colonies/ha)      | 68       | 6.4    | 114      | 10.5   | 102       | 11.5   | <sup>a</sup>        |  |
| Herbaceous cover (%)                    | 25       | 2.8    | 25       | 2.0    | 38        | 2.2    | 0–35                | Kopp et al. (1998)   |
| Litter cover (%)                        | 33       | 3.5    | 33       | 3.6    | 23        | 3.5    | <sup>a</sup>        |  |
| Bare ground (%)                         | 43       | 4.2    | 42       | 3.6    | 40        | 3.5    | 10–60               | Kopp et al. (1998)   |
| Brush cover (%)                         | 24       | 3.5    | 25       | 3.8    | 12        | 2.9    | 5–90                | Lehmann (1984:248), Kopp et al. (1998)                       |
| Radius of obstruction (m)               | 8        | 0.7    | 10       | 0.8    | 11        | 1.2    | 0–13                | Kopp et al. (1998)   |
| Angle of obstruction (°)                | 46       | 3.9    | 46       | 3.7    | 33        | 3.3    | <sup>a</sup>        |  |
| Cone of vulnerability (m <sup>3</sup> ) | 693,567  | 80,638 | 693,498  | 69,155 | 1,022,679 | 97,922 | <750,000            | Kopp et al. (1998)   |

<sup>a</sup> No reported measurements of variable.

(Zar 1999). We considered statistical tests for every analysis to be significant at  $P \leq 0.05$ .

## RESULTS

### Habitat Evaluation

Of the 8 habitat characteristics measured to quantify bobwhite habitat on the 3 sites, most (6–7 variables) occurred within reported habitat-suitability bounds (Table 2). The 1–2 variables falling outside of reported habitat-suitability bounds at a given site were bunchgrass density and cone of vulnerability. Bunchgrass density was below the recommended values on all 3 sites. The estimated cone of vulnerability for R2 was greater than recommended values, which suggests a possible lack of vertical cover (Table 2). Considering all variables collectively, however, our habitat assessment suggested all 3 sites represented suitable bobwhite habitat.

### Survival

We trapped and translocated 409 wild bobwhites ( $n = 202$  and 207 bobwhites during Mar 2013 and 2014, respectively). Only 2 bobwhites died in transit to the release sites during the study ( $n = 1$  mortality/year). We translocated a similar number of males and females (214 M vs. 195 F); however, juveniles comprised the majority of translocated bobwhites

( $n = 292$  juveniles and 117 adults). We attempted to release an equal number of bobwhites into each release site; however, because of a need one day to release translocated bobwhites at the nearest release site (R1) to avoid a transportation delay, R1 received more bobwhites ( $n = 238$  bobwhites) than R2 ( $n = 171$  bobwhites).

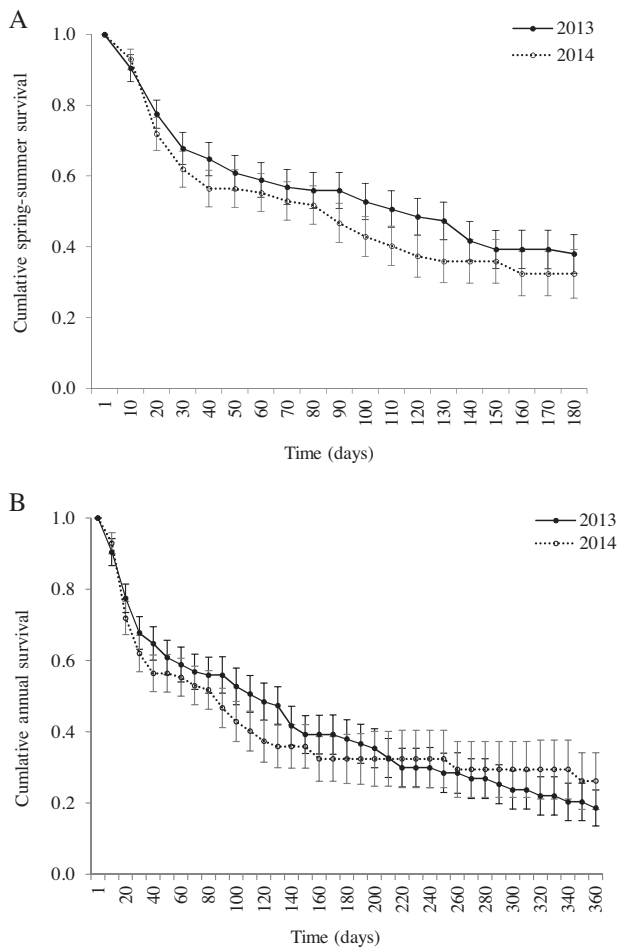
Of the 409 translocated bobwhites, we radio-marked 95 and 91 females in 2013 and 2014, respectively. We documented no difference in spring-summer survival between release sites in 2013 ( $P = 0.21$ ) or 2014 ( $P = 0.82$ ; Table 3). We also documented no difference in annual survival between release sites in 2013 ( $P = 0.12$ ) or 2014 ( $P = 0.79$ ; Table 3). Thus, we pooled across sites to estimate spring-summer and annual survival (Fig. 3). Spring-summer survival ( $\hat{S}$ ) was similar between 2013 ( $\hat{S} = 0.38 \pm 0.05$ ) and 2014 ( $\hat{S} = 0.32 \pm 0.07$ ;  $P = 0.33$ ; Table 3). Annual survival also was similar between 2013 ( $\hat{S} = 0.19 \pm 0.05$ ) and 2014 ( $\hat{S} = 0.23 \pm 0.08$ ;  $P = 0.65$ ; Table 3). However, we documented a difference in survival by age. Juveniles experienced lower spring-summer survival ( $\hat{S} = 0.32 \pm 0.05$ ) than adults ( $\hat{S} = 0.57 \pm 0.11$ ;  $P = 0.007$ ).

Of the 30 *a priori* models built to evaluate factors potentially influencing spring-summer survival, 3 models contained substantial support (i.e.,  $\Delta AIC_c < 2$ ; Table 4). The most parsimonious model indicated a linear time trend effect ( $\beta = 0.065$ , 95% CI = 0.033–0.096) with an additive age

**Table 3.** Spring-summer (Mar–Sep) and annual (Mar–Feb) survival rates ( $\hat{S}$ ) of translocated, radio-marked female northern bobwhites on 2 release sites (R1 and R2) in Stephens and Shackelford counties, Texas, USA, 2013–2015.

| Variable and site | 2013     |           |      |       | 2014     |           |      |       | 2013 vs. 2014 |
|-------------------|----------|-----------|------|-------|----------|-----------|------|-------|---------------|
|                   | <i>n</i> | $\hat{S}$ | SE   | $P^a$ | <i>n</i> | $\hat{S}$ | SE   | $P^a$ | <i>P</i>      |
| Spring-summer     |          |           |      |       |          |           |      |       |               |
| R1                | 54       | 0.46      | 0.08 |       | 49       | 0.32      | 0.10 |       |               |
| R2                | 41       | 0.30      | 0.07 | 0.211 | 42       | 0.32      | 0.09 | 0.821 |               |
| Pooled            | 95       | 0.38      | 0.05 |       | 91       | 0.32      | 0.07 |       | 0.326         |
| Annual            |          |           |      |       |          |           |      |       |               |
| R1                | 54       | 0.24      | 0.07 |       | 49       | 0.21      | 0.09 |       |               |
| R2                | 41       | 0.14      | 0.08 | 0.116 | 42       | 0.26      | 0.13 | 0.792 |               |
| Pooled            | 95       | 0.19      | 0.05 |       | 91       | 0.23      | 0.08 |       | 0.646         |

<sup>a</sup>  $P$ -value for comparison of R1 versus R2.



**Figure 3.** Spring-summer (Mar-Sep; A) and annual (Mar-Feb; B) survival curves ( $\pm$ SE) of translocated, radio-marked female northern bobwhites in Stephens and Shackelford counties, Texas, USA, 2013–2014 and 2014–2015.

effect ( $\beta = 0.896$ , 95% CI = 0.262–1.530) on spring-summer survival. Based on this model, spring-summer survival was considerably lower for juveniles (0.28) compared to adults (0.59). This best model was 2.6 times more likely than the second-best model. The second and third best models included an additive effect of distance translocated and length of holding time, respectively (Table 4), but the beta

estimates for these additive effects overlapped zero. Thus, distance translocated and length of holding time did not explain more of the variation in spring-summer survival than the best model.

We recorded 126 bobwhite mortalities during this study. Avian predation ( $n = 61$  mortalities, 48%) accounted for the majority of the mortalities, followed by mammalian predation ( $n = 41$  mortalities, 33%), unknown predation ( $n = 16$  mortalities, 13%), and unknown cause ( $n = 7$  mortalities, 6%; Table 5). Snake predation accounted for 1 mortality. When evaluating mortality by month, the largest number of mortalities occurred during the first 30 days after translocation of bobwhites (Fig. 3). The primary predator of translocated bobwhites during the month following release (Mar–Apr) changed from 2013 to 2014. Avian predation ( $n = 15$  mortalities, 45%) accounted for the majority of mortalities during March–April 2013, whereas mammalian predation accounted for the majority ( $n = 15$  mortalities, 60%) of mortalities during March–April 2014. After this initial peak in mortality during the first 30 days post-translocation, bobwhite mortalities decreased and remained low throughout the year during 2013–2015 (Fig. 3).

### Reproduction

We located 125 nests during this study. All of the 5 reproductive variables measured (i.e., % nesting, nesting rate, nest success, clutch size, egg hatchability) were similar between 2013 and 2014 (Table 6); therefore, we pooled estimates across years. We observed relatively high values for percentage of females nesting ( $74.1\%$ ), nesting rate ( $1.1 \pm 0.1$  nests/F), nest success ( $46.2\%$ ), clutch size ( $12.6 \pm 0.3$  eggs), and egg hatchability ( $89.0 \pm 2.4\%$ ; Table 6). Based on pooled values and a deterministic model of reproduction, we estimated that 258 chicks were produced during 2013 and 198 chicks were produced during 2014 (Table 6).

### Site Fidelity

We evaluated site fidelity for 65 and 47 translocated, radio-marked bobwhites during March–August 2013 and 2014, respectively. The farthest distance that we documented a translocated, radio-marked bobwhite from its release point was 13 km in 2013 compared to 7 km in 2014 (Fig. 4). In addition, 32% ( $n = 21$  bobwhites) of translocated bobwhites were dispersers (i.e.,  $>2$  km) in 2013 compared to only 15%

**Table 4.** Model selection results of analysis evaluating factors affecting spring-summer (Mar–Sep) survival of translocated, radio-marked female northern bobwhites in Stephens and Shackelford counties, Texas, USA, 2013–2014. T represents a linear time trend effect. S. represents constant survival.

| Rank | Model   | $K^a$ | $AIC_c^b$ | $\Delta AIC_c^c$ | $w^d$ |
|------|---|-------|-----------|------------------|-------|
| 1    | T + age   | 3     | 860.98    | 0.00             | 0.53  |
| 2    | T + age + distance  | 4     | 862.89    | 1.90             | 0.20  |
| 3    | T + age + holding time  | 4     | 862.96    | 1.98             | 0.20  |
| 4    | T + mass + distance + holding time + year + release site + age  | 8     | 867.14    | 6.16             | 0.02  |
| 5    | T   | 2     | 868.51    | 7.53             | 0.01  |
| 19   | S. + mass + holding time + distance + year + release site + age | 7     | 882.86    | 21.88            | 0.00  |
| 20   | S.  | 1     | 888.42    | 27.44            | 0.00  |

<sup>a</sup> Number of parameters.

<sup>b</sup> Akaike's Information Criterion corrected for small sample size.

<sup>c</sup> Difference between a model and the best performing model. We present models with  $\Delta AIC_c < 8.00$ , the global model, and the null model (S.).

<sup>d</sup> Akaike model weight.



**Table 5.** Cause-specific mortalities of translocated, radio-marked female northern bobwhites released in Stephens and Shackelford counties, Texas, USA, during March 2013–March 2015.

| Year and site <sup>a</sup> | Radio-marked    | Mortalities | Avian predation |      | Mammalian predation |      | Snake predation |     | Unknown predation |      | Unknown death |     |
|----------------------------|-----------------|-------------|-----------------|------|---------------------|------|-----------------|-----|-------------------|------|---------------|-----|
|                            | <i>n</i>        | <i>n</i>    | <i>n</i>        | %    | <i>n</i>            | %    | <i>n</i>        | %   | <i>n</i>          | %    | <i>n</i>      | %   |
| 2013                       |                 |             |                 |      |                     |      |                 |     |                   |      |               |     |
| R1                         | 54              | 35          | 15              | 42.9 | 14                  | 40.0 | 1               | 2.9 | 3                 | 8.6  | 2             | 5.7 |
| R2                         | 41              | 32          | 17              | 53.1 | 8                   | 25.0 | 0               | 0.0 | 4                 | 12.5 | 3             | 9.4 |
| Pooled                     | 95              | 67          | 32              | 47.8 | 22                  | 32.8 | 1               | 1.5 | 7                 | 10.4 | 5             | 7.5 |
| 2014                       |                 |             |                 |      |                     |      |                 |     |                   |      |               |     |
| R1                         | 49              | 32          | 16              | 50.0 | 12                  | 37.5 | 0               | 0.0 | 3                 | 9.4  | 1             | 3.1 |
| R2                         | 42              | 27          | 13              | 48.1 | 7                   | 25.9 | 0               | 0.0 | 6                 | 22.2 | 1             | 3.7 |
| Pooled                     | 91 <sup>b</sup> | 59          | 29              | 49.2 | 19                  | 32.2 | 0               | 0.0 | 9                 | 15.3 | 2             | 3.4 |

<sup>a</sup> 2013: 9 March 2013–8 March 2014, 2014: 8 March 2014–7 March 2015.

<sup>b</sup> Individuals released in 2013 that survived over a year were not included.

( $n = 7$  bobwhites) in 2014. Although we recorded some long-distance locations, particularly during 2013, these extreme movements were made by a small number of individuals. Most, 45% ( $n = 29$  bobwhites) and 57% ( $n = 27$  bobwhites), of bobwhites were non-dispersers (remained  $< 1$  km from release point) during 2013 and 2014, respectively (Fig. 4).

### Population Response

Pre-treatment covey-call surveys detected 0.0 coveys/point in the control site and 0.6 coveys/point at the release sites (Fig. 5). This measure of relative abundance increased to a high of 6.4 coveys/point in the control site and 5.2 coveys/point in the release site during the course of the study (Fig. 5). Mean maximum coveys/point did not differ between the control ( $3.0 \pm 1.2$  coveys/point) and release sites ( $2.7 \pm 0.6$  coveys/point;  $F_{1,4} = 0.15$ ,  $P = 0.714$ ; Fig. 5). We did not observe a treatment  $\times$  time interaction ( $F_{4,4} = 0.38$ ,  $P = 0.817$ ). Thus, relative abundance did not statistically differ between treatments during this study.

Pre-treatment helicopter surveys detected 0.26 bobwhites/km at the control site and 0.12 bobwhites/km at the release sites, and estimates increased to a high of 1.35 bobwhites/km and 1.71 bobwhites/km, respectively, during the course of the study (Fig. 6). Mean bobwhites observed/km did not differ between the control ( $0.7 \pm 0.2$  bobwhites/km) and

release sites ( $0.9 \pm 0.2$  bobwhites/km;  $F_{1,4} = 0.25$ ,  $P = 0.640$ ; Fig. 6). We did not document a treatment  $\times$  time interaction ( $F_{4,4} = 0.49$ ,  $P = 0.749$ ). Therefore, relative abundance of bobwhites estimated using helicopter surveys was similar between treatments during this study.

## DISCUSSION

Overall, we documented high annual survival, high reproductive effort, and moderate site fidelity for translocated bobwhites during our study. Despite these positive demographic performances, we did not document a population response resulting from translocation of wild bobwhites.

### Habitat Evaluation

Our habitat measurements indicated that all 3 sites in our study comprised suitable bobwhite habitat. Almost all (6–7 variables) of 8 habitat characteristics diagnostic of bobwhite habitat fell within reported habitat-suitability ranges. The cone of vulnerability for R2 was larger than volumes recommended by Kopp et al. (1998). Therefore, R2 may have lacked the preferred vertical cover to provide bobwhites concealment from avian predators. However, survival estimates of bobwhites translocated to this area indicated predation was similar to the other release site, which did have vertical cover falling within the reported bounds of

**Table 6.** Reproductive measures of translocated, radio-marked female northern bobwhites in Stephens and Shackelford counties, Texas, USA, May–September, 2013–2014. The *P*-value compares 2013 and 2014.

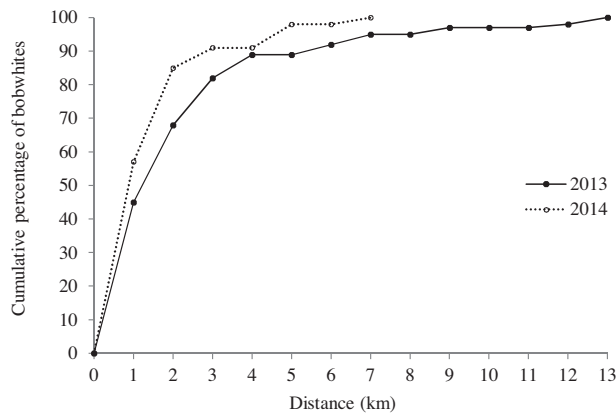
| Variable  | 2013     |          |     | 2014     |          |     | Pooled   |          |     | <i>P</i> |
|---|----------|----------|-----|----------|----------|-----|----------|----------|-----|----------|
|   | <i>n</i> | Estimate | SE  | <i>n</i> | Estimate | SE  | <i>n</i> | Estimate | SE  |          |
| Nesting females (%; realized estimate) <sup>a</sup>     | 62       | 77.4     |     | 50       | 70.0     |     | 112      | 74.1     |     | 0.394    |
| Nesting females (%; potential estimate) <sup>b</sup>    | 27       | 100.0    |     | 15       | 93.3     |     | 42       | 97.6     |     | 0.357    |
| Nesting rate (nests/F; realized estimate) <sup>a</sup>  | 62       | 1.2      | 0.1 | 50       | 0.9      | 0.1 | 112      | 1.1      | 0.1 | 0.162    |
| Nesting rate (nests/F; potential estimate) <sup>b</sup> | 27       | 1.7      | 0.2 | 15       | 1.5      | 0.2 | 42       | 1.6      | 0.1 | 0.268    |
| Nest success (%)  | 72       | 41.7     |     | 47       | 53.2     |     | 119      | 46.2     |     | 0.261    |
| Clutch size (eggs/nest)                                 | 41       | 12.4     | 0.4 | 28       | 12.9     | 0.4 | 69       | 12.6     | 0.3 | 0.426    |
| Egg hatchability (%) <sup>c</sup>                       | 25       | 86.6     | 3.7 | 18       | 91.4     | 2.5 | 43       | 89.0     | 2.4 | 0.332    |
| Simulated production (no. chicks) <sup>d</sup>          |          | 257.9    |     |          | 197.6    |     |          | 236.5    |     |          |

<sup>a</sup> Based on females alive 1 May of a given year.

<sup>b</sup> Based on females alive 1 May and surviving through 31 August of a given year.

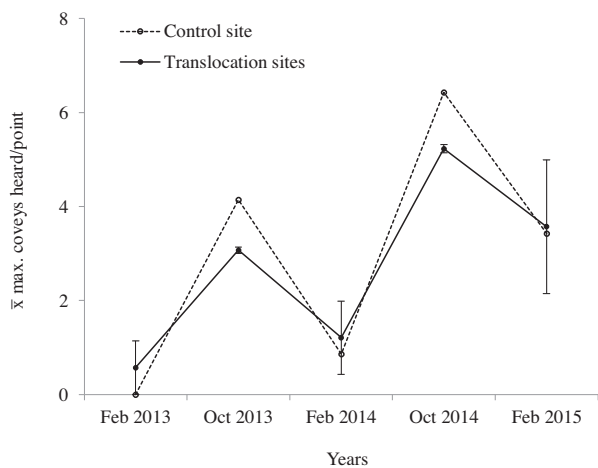
<sup>c</sup> Number of hatched eggs/clutch size  $\times 100\%$ ; only based on successful nests.

<sup>d</sup> Simulated cumulative effect of the 5 reproductive variables; realized estimates used when applicable.

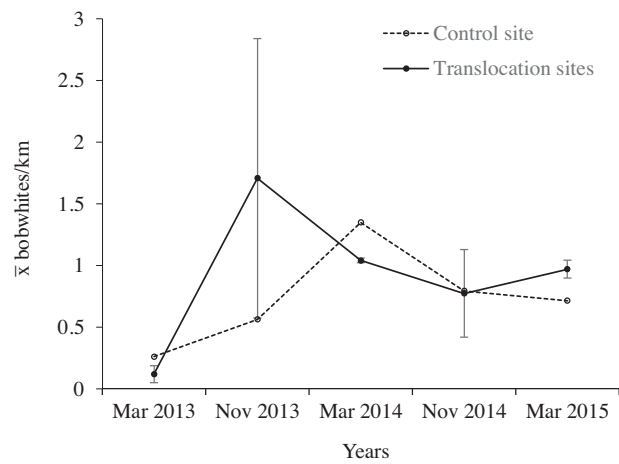


**Figure 4.** Relative cumulative frequency distribution of distance (km) from release point to farthest observed location of translocated, radio-marked female northern bobwhites in Stephens and Shackelford counties, Texas, USA, March–September, 2013–2014.

suitability. Bunchgrass density on all 3 sites was lower than recommended values reported by Lehmann (1984), Guthery (1986), and Arredondo et al. (2007). This finding may have resulted from bunchgrass density actually being lower on our study sites, possibly caused by the drought of 2011–2014, or from differences in methods used to measure bunchgrass density. Lehmann (1984) and Guthery (1986) recommended a bunchgrass density of 618 nesting clumps/ha, but their estimates were speculations and lacked empirical data. Arredondo et al. (2007) provided empirical estimates obtained using continuous-selection functions (Kopp et al. 1998) and reported that bobwhites selected for nest sites in areas containing >730 suitable nest clumps/ha. However, this empirical estimate of 730 nest clumps/ha may be high because of bias associated with the point-center quarter method. Mitchell (2007) discussed biases in point-center quarter



**Figure 5.** Relative abundance ( $\bar{x} \pm SE$  max. coveys heard/point) of northern bobwhites at the control site (C) and release sites (R1 and R2) based on covey-call counts in Stephens and Shackelford counties, Texas, USA, February and October, 2013–2015. We released translocated bobwhites at release sites after February 2013 and February 2014 surveys. We do not report standard error for the control site because only 1 replicate existed.



**Figure 6.** Relative abundance ( $\bar{x} \pm SE$  bobwhites/km) of northern bobwhites at the control site (C) and release sites (R1 and R2) based on helicopter surveys in Stephens and Shackelford counties, Texas, USA, March and November, 2013–2015. We released translocated bobwhites at release sites after March 2013 and March 2014 surveys. We do not report standard error for the control site because only 1 replicate existed.

methodology and provided modifications to methodology to yield unbiased estimates of plant density. We used the revised methodology suggested by Mitchell (2007) to estimate bunchgrass density and therefore differences in bunchgrass density between our study and Arredondo et al. (2007) could be caused by differences in the point-center quarter methodology.

Despite the reason, the observation that bunchgrass density was not within the recommended habitat-suitability bounds does not necessarily indicate that nesting cover was limited at the study sites. Bobwhites in the Rolling Plains of Texas readily nest in multiple nesting substrates (besides the typical nesting substrate of bunchgrass) such as prickly pear cactus, catclaw (*Acacia* spp.), and yucca (*Yucca* spp.). For example, Hernández et al. (2003) documented that 30% of bobwhite nests were located in prickly pear cactus and 12% in catclaw in the Rolling Plains ( $n = 81$  nests). Similarly, Carter et al. (2002) documented that 14 of 21 bobwhite nests were located in either prickly pear or brush ( $n = 21$  nests). In our study, prickly pear and shrub cover was common on all 3 sites (68–114 prickly pear colonies/ha and 12–25% shrub cover), and thus nesting substrates commonly used by bobwhites in this ecoregion (i.e., prickly pear, shrubs, and bunchgrasses) were readily present on all study sites. Bobwhites made use of this general availability of nesting substrates given that 64% of bobwhite nests ( $n = 124$ ) were located in prickly pear, 26% in bunchgrasses, and 6% in shrubs (Downey 2015).

### Survival and Reproduction

Survival of translocated bobwhites decreased most rapidly during the spring (Mar–May), after which mortality lessened and leveled. Specifically, about a third of our radio-marked female bobwhites died within 30 days after the first covey was released. Despite this initial high mortality, translocated bobwhites in our study survived the 183-day spring-summer season (0.32–0.38) similar to (0.27–0.41; Burger et al. 1995; Terhune et al. 2006a, 2010; Sisson et al. 2009) or better than

bobwhites in other studies (0.09–0.18; Liu 1995, Peters et al. 2015). Higher bobwhite spring–summer survival (0.56–0.64) was reported for translocated bobwhites by Jones (1999), but Jones (1999) estimated survival over a shorter interval (131 or 164 days) than in our study so higher survival estimates are partly resulting from a shorter time interval. Thus, comparison of survival among translocation studies is limited because studies define survival over different time intervals or use different criteria for censoring. A more meaningful comparison among studies is a comparison of annual survival, which standardizes the time interval (i.e., 365 days). We documented relatively high annual survival rates (0.19 and 0.23) for translocated bobwhites. Reports of annual survival for translocated bobwhites are limited. The 1 translocation study (Liu 1995) that documented annual survival rates reported values lower for translocated bobwhites (0.08 and 0.13) and even resident bobwhites (0.14) than those that we documented. Our estimates of translocated bobwhites are similar (0.20) to resident bobwhites studied by Sisson et al. (2009) but greater than values (0.04–0.07) documented by most studies involving radio-marked bobwhites (Burger et al. 1995, Suchy and Munkel 2000, Cox et al. 2004, Lohr et al. 2011, Peters et al. 2015). Guthery (1997) estimated that 20–30% annual survival was necessary for sustainable bobwhite populations. The survival rates that we documented were within or very close to this survival threshold necessary for population persistence. Thus, considering that bobwhites in our study were radio-marked and experienced the added stress of translocation, the annual survival rates that we documented were encouraging.

We documented that raptors and mammals accounted for most of the mortalities (48.4% and 32.5%, respectively) of translocated bobwhites. Although these values appear high, they are similar to values reported by other studies for avian predation (41.2–61.3%) and mammalian predation (32.4–35.2%; Terhune et al. 2006a, Sisson et al. 2009, Lohr et al. 2011). In general, predation accounted for 95% of bobwhite mortalities. It is difficult to ascertain if stress may have been the ultimate cause in some cases and indirectly influenced vulnerability to predation. Stress has been documented to affect the health, behavioral abilities, and cognitive pathways of translocated animals (Teixeira et al. 2007). If stress is linked to mortality, then future studies may consider injecting bobwhites with vitamin E and selenium, which is hypothesized to reduce muscle damage caused by stress and increase survival of translocated bobwhites (Abbott et al. 2005). Given this high mortality 1-month post-release, we recommend that future studies investigate the benefits of housing bobwhites in a protective area (i.e., soft-release) until 1 May to minimize the time that translocated bobwhites are exposed to predators prior to the start of the nesting season. A combination of these 2 techniques (injection of vitamin E and selenium and housing bobwhites until 1 May) may provide the best method to improve survival 1-month post-translocation in this ecoregion and help ensure the greatest number of females enter the nesting season.

Bobwhites in our study reproduced as well or better than what has been reported in past translocation research (Jones

1999, Parsons et al. 2000, Terhune et al. 2006a, Scott et al. 2013). The realized nesting effort in our study (74.1% F nested) was similar (73.0–78.6%) or higher (36.9–56.5%) than efforts made by translocated and resident bobwhites in other translocation studies (Jones 1999, Terhune et al. 2006a, Scott et al. 2013). The realized nesting rate of bobwhites in our study (1.1 nests/F) also was similar (0.6–1.6 nests/F) or higher (0.2–0.4 nests/F) than rates reported by other translocation studies (Parsons et al. 2000; Terhune et al. 2006a, b; Scott et al. 2013). In addition, bobwhite nest success (hatch rate) during our study (46.2%) was similar to estimates recorded by Terhune et al. (2006a) and Scott et al. (2013) (41.7–51.2%). Other reproductive variables such as clutch size and egg hatchability were similar between bobwhites in our study and previous studies (Jones 1999, Scott et al. 2013). Thus, translocated bobwhites in our study exhibited a reproductive effort aligned with the reproductive potential of the species as documented in prior research.

### Site Fidelity

We documented moderate site fidelity of translocated bobwhites. Specifically, 15–32% of translocated bobwhites were dispersers (i.e., traveled >2 km from release point) in our study. Liu et al. (2002) reported that  $\geq 10\%$  ( $n \geq 50$ ) of translocated bobwhites traveled  $\geq 2.5$  km, and Scott et al. (2013) recorded that 41% ( $n = 18$ ) of translocated bobwhites traveled >2 km from their release point. Both our study and these 2 studies translocated bobwhites to areas that had few resident bobwhites, which may serve as a possible explanation for dispersal of translocated bobwhites. Scott et al. (2013) suggested that the lack of a resident bobwhite population at their release site may explain the high (41%) dispersal of translocated bobwhites they observed. Ahlering et al. (2006) hypothesized that the mere presence of other conspecifics may facilitate higher site fidelity through a phenomenon known as conspecific attraction. Conspecific attraction occurs when individuals settle in areas where conspecifics reside, and this phenomenon has been documented in several avian species (Ahlering et al. 2006). Our findings provide some support for this phenomenon. Relative abundance on our release sites was low prior to translocation (0.12 bobwhites/km; spring 2013) and increased after translocation (1.7 bobwhites/km; spring 2014). Correspondingly, we documented a decrease in dispersers from 2013 (32%) to 2014 (15%). Indeed, studies that have translocated bobwhites to areas containing a resident population have documented high site fidelity (89–98%; Jones 1999; Terhune et al. 2006b, 2010). Jones (1999) reported that translocated bobwhites were integrating with resident coveys. Hence, it is possible that higher site fidelity may be achieved through conspecific attraction by translocating bobwhites to areas before the resident population becomes too low. A potential technique to increase site fidelity if a resident population is not present may involve playing bobwhite-call recordings in the release area via automated audio units. We recommend that conspecific density be incorporated in future studies as a covariate in modeling site.

It has also been proposed that habitat fragmentation or poor habitat can result in bobwhite dispersal (Fies et al.

2002). In fact, Scott et al. (2013) suggested that the high (41%) dispersal observed in their study may have been attributed to the fragmented nature of their study area. However, translocated bobwhites during this study were released into counties that consisted of 94% rangeland involving relatively large contiguous patches of rangeland that are closely spaced (NRCS 2014). Thus, fragmented habitat likely is not the reason why some translocated bobwhites dispersed from our release sites because our release sites were composed of suitable bobwhite habitat occurring on a landscape consisting of mostly contiguous rangeland (NRCS 2014).

### Population Response

We failed to document a population response resulting from the translocation of wild bobwhites beyond that of a control. Bobwhite relative abundance was similar between control and release sites during all 5 time periods. Our failure to document a population response post-translocation of bobwhites beyond that of a control was similar to Jones (1999) and Scott et al. (2013). Only Terhune et al. (2006a) documented a population response, but this response was not consistent. The study by Terhune et al. (2006a) involved 3 release sites: 2 small (285–340 ha) and 1 large site (405 ha). Terhune et al. (2006a) reported a statistically significant population response on the 2 smaller sites (57% and 109% increase in coveys detected/hr hunting) but not a statistically significant response on the largest site (18% increase in coveys detected/hr hunting). In addition, Terhune et al. (2006a) translocated bobwhites over a considerably short distance (~1.2 km) compared to the long distances (95–221 km) (Jones [1999], Scott et al. [2013]), and our study translocated bobwhites.

Several hypotheses may explain the lack of a population response post-translocation including minimum number of translocated individuals, lag effect, and strong influence of precipitation on bobwhite populations. According to Griffith et al. (1989), a minimum number of individuals need to be translocated to have a successful population increase in the long-term. This minimum number of individuals is unknown for bobwhites. However, releasing a larger number of individuals *per se* may not increase the probability of a successful translocation. Griffith et al. (1989) suggested that translocation success may be more probable for translocation programs occurring over longer time periods. In our study, the release of a large number of bobwhites (~200 bobwhites/yr) for 2 years was not sufficient to result in a population response. Thus, future bobwhite translocation efforts may be improved by releasing a minimum number of individuals needed each year for >2 years. Another possible explanation is that the effect of translocation has a latent period or accumulates over time. Black-tailed prairie dog (*Cynomys ludovicianus*) populations grew 3.2 times 1 year post-translocation, but exhibited a growth of 9.2 times 4 years post-translocation (Dullum et al. 2005). Therefore, it is possible that a detectable population response has yet to occur on our study sites but may occur in the future or that few or no bobwhites would exist today on our release sites if bobwhite translocations had not occurred.

Finally, it also is possible that the lack of precipitation prior to, and during, our study (2011–2014) negatively affected the bobwhite population and limited the population response to translocation. Precipitation has a profound influence on all aspects of bobwhite demographics (i.e., survival, reproduction, abundance) and often accounts for such a large portion of the variation in bobwhite population (Rice et al. 1993, Bridges et al. 2001, Parent et al. 2016). For example, Hernández et al. (2005) documented that a lower percentage of females nested during a dry period (52.6%) versus a wet period (100%) in southern Texas. The nesting rate also was lower during the dry period ( $1.2 \pm 0.3$  nests/F) compared to the wet period (2.3 nests/F), and the nesting season was shorter (69 days vs. 159 days, respectively). In addition, Hernández et al. (2005) documented that fall-winter survival was lower during the dry period (0.30) compared to the wet period (0.60), a finding similar to Tri et al. (2016) who documented that bobwhite survival decreased with increasing aridity. Given this strong influence of weather, bobwhite population response to management often is limited during dry years in the southwestern region of the bobwhite's range (Bridges et al. 2001, Hernández et al. 2005, Parent et al. 2016). Consequently, the negative influence of low rainfall may have overridden any potential positive demographic response associated with translocation.

### MANAGEMENT IMPLICATIONS

Translocation of wild bobwhites is a proposed technique to bolster declining populations. Past research recommends that translocation efforts begin prior to population declines or low densities to increase likelihood of translocation success. Our findings support this general recommendation. We propose the following management recommendations to potentially increase the probability of bobwhite translocation success on Texas rangelands. First, future research should investigate the benefits of using soft-release (i.e., sequestering translocated bobwhites in a protected, captive area) during March–April, in conjunction with vitamin E and selenium injections to possibly ameliorate translocation stress (Abbott et al. 2005), to increase bobwhite survival during the 1-month period immediately following translocation. Second, we recommend releasing translocated bobwhites on sites that have not reached very low-densities, releasing translocated bobwhites near known locations of resident bobwhites, playing recordings of bobwhites calling, or housing call-back bobwhites on release sites to possibly increase site fidelity using conspecific attraction. Lastly, future research should incorporate a genetic metric of evaluation (in addition to demographic metrics) to permit assessment of the genetic contribution of translocated bobwhites to the resident population and allow for improved evaluation of translocation success.

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