Research Article

Northern Bobwhite Breeding Season Ecology on a Reclaimed Surface Mine

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ABSTRACT Surface coal mining and subsequent reclamation of surface mines have converted large forest areas into early successional vegetative communities in the eastern United States. This reclamation can provide a novel opportunity to conserve northern bobwhite (*Colinus virginianus*). We evaluated the influence of habitat management activities on nest survival, nest-site selection, and brood resource selection on managed and unmanaged units of a reclaimed surface mine, Peabody Wildlife Management Area (Peabody), in west-central Kentucky, USA, from 2010 to 2013. We compared resource selection, using discrete-choice analysis, and nest survival, using the nest survival model in Program MARK, between managed and unmanaged units of Peabody at 2 spatial scales: the composition and configuration of vegetation types (i.e., macrohabitat) and vegetation characteristics at nest sites and brood locations (i.e., microhabitat). On managed sites, we also investigated resource selection relative to a number of different treatments (e.g., herbicide, disking, prescribed fire). We found no evidence that nest-site selection was influenced by macrohabitat variables, but bobwhite selected nest sites in areas with greater litter depth than was available at random sites. On managed units, bobwhite were more likely to nest where herbicide was applied to reduce sericea lespedeza (*Lespedeza cuneata*) compared with areas untreated with herbicide. Daily nest survival was not influenced by habitat characteristics or by habitat management but was influenced by nest age and the interaction of nest initiation date and nest age. Daily nest survival was greater for older nests occurring early in the breeding season (0.99, SE < 0.01) but was lower for older nests occurring later in the season (0.08, SE = 0.13). Brood resource selection was not influenced by macrohabitat or microhabitat variables we measured, but broods on managed units selected areas treated with herbicide to control sericea lespedeza and were located closer to firebreaks and disked native-warm season grass stands than would be expected at random. Our results suggest the vegetation at Peabody was sufficient without manipulation to support nesting and brood-rearing northern bobwhite at a low level, but habitat management practices improved vegetation for nesting and brood-rearing resource selection. Reproductive rates (e.g., nest survival and re-nesting rates) at Peabody were lower than reported in other studies, which may be related to nutritional deficiencies caused by the abundance of sericea lespedeza. On reclaimed mine lands dominated by sericea lespedeza, we suggest continuing practices such as disking and herbicide application that are targeted at reducing sericea lespedeza to improve the vegetation for nesting and brood-rearing bobwhite. © 2016 The Wildlife Society.

KEY WORDS brood, *Colinus virginianus*, habitat management, Kentucky, nest-site selection, nest survival, northern bobwhite, reclaimed surface mine.

Since the passage of the Surface Mine Control and Reclamation Act (SMCRA) in 1977, >600,000 ha of surface mine lands have been reclaimed in the eastern United States (U.S. Office of Surface Mining Reclamation and Enforcement 2008). These reclaimed mine lands may play an important role in the conservation of northern bobwhite (*Colinus virginianus*; bobwhite) populations, especially within

the Central Hardwood Bird Conservation Region of the central United States (Applegate et al. 2011). Reclaimed mine land presents an opportunity to manage bobwhite at a larger scale not often afforded by fragmented agricultural landscapes (DeVault et al. 2002, Hernández et al. 2013). Although regulations encourage reclaimed mine land be re-vegetated with native species, most reclamation has been completed with non-native invasive plant species because it is less expensive, growth is rapid, and bond requirements are satisfied (Koll 2002). The use of non-native species arrests succession leaving many reclaimed mine lands in a prolonged (>25 yr) early successional state (Chaney et al. 1995, Koll 2002, Gilland and McCarthy 2014). Such reclamation practices may limit the quality of these areas for northern bobwhite (Brooke et al. 2015, Peters et al. 2015, Unger et al. 2015).

Sericea lespedeza (*Lespedeza cuneata*) and tall fescue (*Schedonorus arundinaceus*) are 2 of the most common plants used in the reclamation process. Multiple studies have reported, unless controlled, tall fescue-dominated fields provide poor-quality habitat for bobwhite (Barnes et al. 1995, Harper and Gruchy 2009, Osborne et al. 2012). Extensive sericea lespedeza cover can limit the establishment of native herbaceous and woody plants that provide food and cover for bobwhite throughout the year (Adams et al. 1973, Wade 1989, Foster and Gross 1998). Additionally, sericea lespedeza may limit the abundance of insects important for broods and the seeds are indigestible by bobwhite (Bugg and Dutcher 1989, Blocksome 2006). Recent research on our study area suggests disking and herbicide application can reduce sericea lespedeza, influence habitat selection by bobwhite during the breeding season (Brooke et al. 2015), and increase breeding season survival (Peters et al. 2015). Additionally, grasshopper sparrow (*Ammodramus savannarum*) fecundity was lower on older reclaimed mine sites associated with increased sericea lespedeza cover compared with newly reclaimed areas (Wood and Ammer 2015). However, the influence of land management on nesting and brood-rearing bobwhite on reclaimed mines has not been reported.

Bobwhite are a short-lived and highly productive species and populations can be limited by their reproductive potential (Folk et al. 2007). Bobwhite depend on their prolific reproductive output (e.g., renesting and double clutching) to compensate for high annual mortality (Roseberry and Klimstra 1984, Burger et al. 1995). To properly manage reclaimed mine land for bobwhite, it is important to understand how characteristics of the vegetation (i.e., structure, composition) and composition and arrangement of vegetation types on the landscape may influence bobwhite fecundity. Such metrics have been linked to bobwhite resource selection and survival (Seckinger et al. 2008, Holt et al. 2009, Janke et al. 2015), but few studies have linked these metrics to nest survival (Taylor et al. 1999*b*, Potter et al. 2011). Furthermore, few studies have assessed the influence of direct, active habitat manipulation on nest success and those that have suggested there was no evidence that habitat management influenced nest success across multiple spatial scales (Potter et al. 2011).

We conducted an experiment to identify factors influencing bobwhite nest-site selection, nest survival, and brood resource selection in the context of habitat management practices on a reclaimed surface mine in west-central Kentucky, USA. Our objectives were to 1) identify factors influencing bobwhite nest-site selection, nest survival, and brood resource selection at the macrohabitat scale (i.e., the selection of locations based on the configuration and composition of vegetation types on the landscape), and the microhabitat scale (i.e., based on the composition and structure of the vegetation at nest sites and brood locations; Kopp et al. 1998); and 2) estimate the influence of management on nest-site selection, nest survival, and brood resource selection.

STUDY AREA

We conducted our study from 2010 to 2013 on the Ken (1,853 ha) and Sinclair (1,471 ha) areas of the Peabody Wildlife Management Area (Peabody; 37°14′ 39″, 87°1′15″), an 18,000-ha property owned and managed by the Kentucky Department of Fish and Wildlife Resources in Ohio and Muhlenberg counties, west-central Kentucky (Fig. 1). Peabody consisted of reclaimed surface mine land dominated by planted, non-native, invasive species including sericea lespedeza, tall fescue, sweet clover (*Melilotus officinalis*), field brome (*Bromus arvensis*), smooth brome (*Bromus inermis*), and Kentucky bluegrass (*Poa pratensis*). Annual precipitation averaged 123.1 cm and annual temperature averaged 13.9°C from 1996 to 2015 (National Oceanic and Atmospheric Administration 2016).

We delineated 4 major vegetation types, representing 91% of our study area. The most dominant vegetation type, open herbaceous, comprised 36% of the study area and was dominated by sericea lespedeza, tall fescue, goldenrod (*Solidago canadensis*), common ragweed (*Ambrosia artemisiifolia*), and field brome. Shrub cover comprised 25% of the study area and was dominated by trees and shrubs including black locust (*Robinia psuedoacacia*), sumac (*Rhus* spp.), autumn olive (*Elaegnus umbellata*), brambles (*Rubus* spp.), and coralberry (*Symphoricarpos orbiculatus*). Forest cover comprised 22% of the study area and was characterized by having a semi-open canopy dominated by eastern cottonwood (*Populus deltoides*), green ash (*Fraxinus pennslyvanica*), and red maple (*Acer rubrum*) with an understory dominated by Japanese honeysuckle (*Lonicera japonica*) and brambles. Planted native warm-season grass (NWSG) comprised 8% of the study area and was dominated by big bluestem (*Andropogon gerardii*), indiangrass (*Sorghastrum nutans*), and switchgrass (*Panicum virgatum*), with sericea lespedeza dominating the forb component. The remaining 9% of the study areas included water, wetlands, roads, firebreaks, and man-made structures.

METHODS

Study Design
We divided each study area (i.e., Ken and Sinclair) into 2 experimental units to assess the influence of habitat manipulations on bobwhite nest-site selection, nest survival,
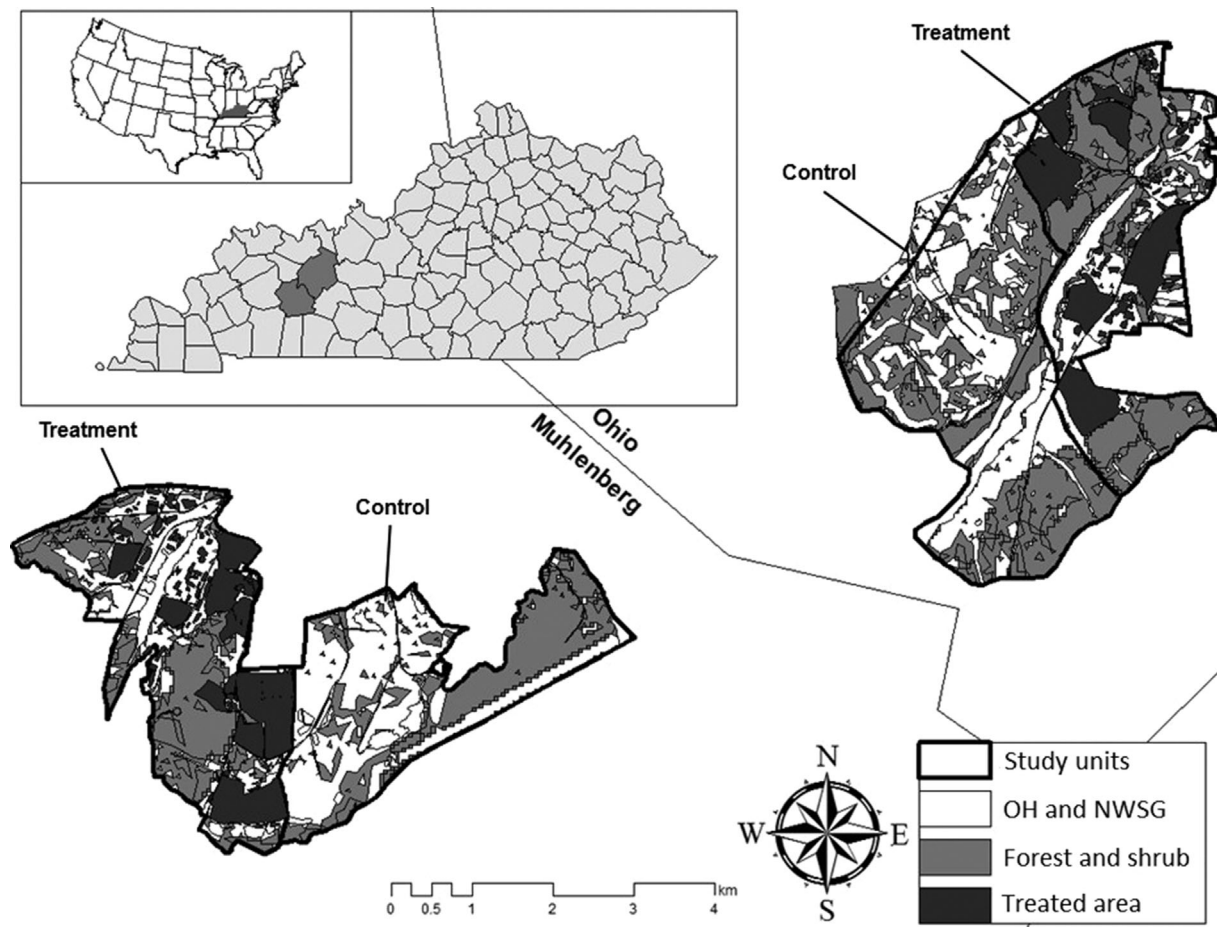
Figure 1. Location of study sites to determine the influence of habitat manipulations on northern bobwhite nest-site selection, nest survival, and brood resource selection on a reclaimed surface mine, Peabody Wildlife Management Area, 2010–2013, Kentucky, USA. Vegetation types included open herbaceous (OH), native warm-season grass (NWSG), forest, and shrub.

and brood resource selection. We randomly assigned 1 unit of each study area to remain undisturbed (control) and manipulated the other unit using various management techniques (treatment). The control units of Sinclair and Ken were 673 ha and 1,043 ha, respectively, whereas the treatment units were 798 ha and 810 ha, respectively. We focused habitat manipulations to open areas (open herbaceous and NWSG) on the treatment units; we manipulated 50% of open areas on treatment units during our study.

Habitat manipulations included prescribed fire, linear and block-shaped disking, and herbicide application to reduce sericea lespedeza. We conducted prescribed burns (329 ha) annually during the dormant season with individual burn units averaging 12.1 ha. We conducted disking (349 ha) with an offset disk to remove and incorporate standing vegetation into the soil, and used a finish disk and cultipacker afterward to smooth the soil surface. Disked blocks averaged 0.54 ha and disking occurred throughout the year with a majority (64%) occurring from August through March. We performed disking on a 3-year return interval. We disked approximately 56 km of 7- to 9-m-wide firebreaks annually, planted firebreaks to winter wheat (*Triticum aestivum*) in late summer, and firebreaks remained fallow during the following growing season. We used aerial application

(142 ha) of metsulfuron methyl (19.05 g active ingredient [a.i.]/ha; Escort XP®, DuPont, Wilmington, DE) to reduce sericea lespedeza in August 2010. Disking and herbicide applications were effective at reducing sericea cover, increasing plants important as bobwhite food, and increasing openness at ground-level, but prescribed fire did not reduce sericea, increase bobwhite food plants, or increase openness at ground-level (Brooke et al. 2015).

Capture and Radio Telemetry
We captured bobwhite from August 2009 to September 2013 using Stoddard (1931) funnel traps. We recorded body mass, sex, and age (juvenile or adult) of all captured individuals (Rosene 1969). Birds >90 g in mass received 2 aluminum leg bands and we fitted individuals >120 g with a necklace-style very high-frequency (VHF) radio-transmitters (∼6 g, American Wildlife Enterprises, Monticello, FL). Transmitters were equipped with a 12-hour mortality sensor. Trapping, handling, and banding protocols were approved by the University of Tennessee Institutional Animal Care and Use Committee (permit no. 2042-0911).

We tracked radio-marked bobwhite ≥3 days/week to locate nests using the homing method (White and Garrot 1990). We assumed an individual located in the same general

area for 2 to 3 consecutive days was nesting. We conducted a nest search once the individual left the area to locate the nest. We monitored nests daily and visually assessed nests at least once per week when incubating adults were away from the nest (Taylor et al. 1999b). We assumed we found all nests on day 1 of incubation when estimating nest initiation and nest age (Potter et al. 2011) unless we knew the actual starting date of nest initiation. Mean error for this assumption was 4.88 days (95% CI = 3.60–6.27). Upon nest completion and after the radio-marked bird and brood had vacated the nest site, we examined nests to determine hatching success (DeMaso et al. 1997). We determined the number of chicks hatched/nest from egg shell remains at the nest site (DeVos and Mueller 1993). We flushed brooding individuals weekly starting 2 weeks after nest completion to determine if the brood was still present. We tracked broods until flushing confirmed no brood was present.

Microhabitat and Macrohabitat Variables
We collected vegetation data for all nesting attempts during the 2012 and 2013 breeding season to characterize bobwhite nest-site selection at the microhabitat scale. We also collected vegetation data at a random subset of brood locations during the same time period to characterize brood resource selection. For nest-site and brood resource selection, we collected vegetation data at used locations and at 1 random location within 165 m in the same vegetation type. Therefore, our inference at the microhabitat scale is limited to the availability within a given vegetation type. We used 165 m because it was the greatest average daily movement of adult bobwhite during any season in our study area (Unger et al. 2015).

After each nesting attempt, we recorded the presence of individual plant species at 0.25-m intervals along 1-m point transects in all 4 cardinal directions centered on the nest. We estimated the percent cover of each species as the total number of times the species was present divided by the 16 transect points. We categorized plant species into groups: NWSG (planted and volunteering), forbs, cool-season grasses, other grasses, and sericea lespedeza. We measured litter depth 0.5 m from the nest in each cardinal direction and averaged those to obtain a single litter depth value for each nest. We measured the density (stems/ha) of shrubs and saplings by counting the number of stems >1.37 m in height and ≤11.4 cm diameter at breast height within a 5-m-radius plot centered on the nest. Finally, we measured the visual obstruction (VO) at the nest using a modified Nudds (1977) cover board divided into 8 25 × 25-cm strata; we recorded the percent of each stratum covered by standing vegetation (0 = 0%, 1 = 1–20%, 2 = 21–40%, ..., 5 = 81–100%). We placed the board at the nest and read it 5 m from the nest in all cardinal directions. We averaged the 4 cover board estimates for each stratum to obtain 8 visual obstruction measurements (VO1–VO8) per nest where VO1 represented the stratum 1.75–2.00 m and VO8 represented 0–0.25 m above the ground surface. We estimated the same variables at 1 paired random location per nest.

We measured similar vegetation parameters at brood locations. We used a 30-m point transect centered on the brood location to quantify plant composition. Plant groups were NWSG, cool-season sod-forming grasses, sericea lespedeza, bobwhite summer food plants (Eubanks and Dimmick1974, Buckner and Landers 1979, Brennan and Hurst 1995), forbs, and shrub species <1.37 m tall. We measured litter depth and visual obstruction (VO1–VO8) at 10-m intervals along the transect. Additionally, we measured the structure of the vegetation at ground level by using a sight-tube (Gruchy and Harper 2014), which provided an index of openness at ground level. We measured woody stem density as described for nests. We collected vegetation parameters for brood sites at 1 paired random location as well.

We used variables from our vegetation sampling for the microhabitat resource selection analysis. Variables for nest sites included composition metrics, the 8 VO readings, litter depth, and the density of woody stems. Variables for brood resources selection included the same variables as the nest sites with the addition of bobwhite food plant data and the sight tube measurements.

We collected macrohabitat covariates from digitized aerial imagery with a cell size of 10 m in ArcGIS 10.0 (ESRI, Redlands, CA). We mapped habitat management activities using global positioning system (GPS) units immediately following application. We created raster layers with the digitized vegetation types and overlaid all management activities. We updated treatments on the raster images monthly to account for any management taking place during the breeding season. Given the importance of edge to bobwhite resource selection (Guthery and Bingham 1992), we also used Fragstats 4.0 (McGarigal et al. 2012) to estimate the amount of woody-to-open edge within 165 m of each brood or nest telemetry location and random point. We defined woody-to-open edge in 2 ways: the density (m/ha) of shrub cover to open cover (open herbaceous and NWSG) and the density (m/ha) of forest cover to open cover. We also estimated the contagion index within 165 m of all locations. The contagion index is a measure of the interspersion and dispersion of the landscape (O'Neill et al. 1988). Smaller contagion values (near 0) indicate a more interspersed area with smaller patches of differing vegetation and larger values (near 1) indicate larger areas of similar vegetation (O'Neill et al. 1988). We estimated bobwhite use of each vegetation type and landscape feature using continuous proximity-to variables (e.g., Euclidean distance to the nearest road) rather than categorical indicator variables (e.g., in road, not in road) to avoid misclassifications resulting from telemetry error and patch size (Conner et al. 2003, Hoffman et al. 2010). Furthermore, proximity-to variables are more robust compared with indicator variables allowing patch size and shape to influence selection (Conner et al. 2003). We also calculated the core area of all 4 major vegetation types (using a 30-m edge effect; Peters et al. 2015) around each nest location.

We used variables related to habitat manipulations to assess the influence of management on nest-site selection and brood resource selection. We measured the proximity of each brood and nest location and random point to the nearest firebreak and disked area in open herbaceous and NWSG to determine the influence of these practices. We also calculated

the total disked area (ha) in a 165-m buffer around telemetry and random locations. We included all disked area, regardless of the time since disking (i.e., 1, 2, 3 years). We used indicator variables (0 or 1) to determine the influence of herbicide applications and prescribed fire (during the prior dormant season) on nest-site and brood resource selection.

Nest-Site and Brood Resource Selection

We used discrete-choice analysis at 2 spatial scales to identify characteristics influencing bobwhite nest-site and brood resource selection based on available resources. We chose discrete-choice analysis because it allowed us to quantify resource selection as availability changed over time and space, and it accommodated continuous and categorical variables (Cooper and Millspaugh 1999, McDonald et al. 2006).

Discrete choice assumes selection is a function of a series of choices made by an individual and the choices are based on the characteristics of available resources at a given time and place. The resources available to an individual are considered the choice-set and availability is defined by the researcher (Cooper and Millspaugh 1999). We defined availability for the microhabitat-scale analysis based on the random vegetation sampling described above. We defined availability for the macrohabitat scale as the resources within 165 m (i.e., greatest daily average movement in our study area; Unger et al. 2015) of each brood telemetry location and 210 m from each nest. We chose a 210-m buffer for nest-site selection based on previous literature (Taylor et al. 1999b, Potter et al. 2011). To determine availability at the macrohabitat scale, we created 5 random (i.e., available) points within 165 m of each brood location (used) and 5 random points within 210 m of each nest.

We performed the analyses using Cox proportional hazards (COXPH) and Cox proportional hazards mixed-effects models (COXME; Therneau 2015) in program R (R package version 3.1.1, www.r-project.org, accessed 1 Dec 2014). We conducted separate analyses for nest site and brood locations and for the different scales of selection. We used a likelihood-ratio test to compare models with and without random effects for each variable to control for variation between individuals (Duchesne et al. 2010). We retained significant random terms in all subsequent models. For each analysis, we used the purposeful model-building strategy outlined in Hosmer et al. (2013) to construct candidate models. We started by analyzing each variable independently in a model. We retained any variable with a $P < 0.25$ and included all retained variables in a global model. We then ran the global model and removed the most insignificant variable ($P > 0.05$) based on the P-value. Finally, we ran the model without the removed variable and repeated the process until only significant variables remained in the model. Variables eliminated in the first step were added back into the reduced model, individually, to determine whether the significance of the variable changed with the inclusion of only significant variables. We used a unit (i.e., control, treatment) interaction with each significant variable to determine whether selection differed between control and treatment.

We tested the influence of variables directly related to management at the macrohabitat scale by using only locations

from the treatment units. We used the best model from the above analysis as a base model to which we added management-related variables, again using purposeful model-building strategies.

We used an information-theoretic approach to evaluate the models created by purposeful model-building strategies. We used Akaike's Information Criterion adjusted for small samples sizes (AIC_c) to rank the models. We considered all models with a $\Delta AIC_c < 2$ to be competing models, explaining some of the variation in resource selection (Burnham and Anderson 2002). We averaged beta estimates across all the competing models if the top model was not the most parsimonious model in the candidate set (Burnham and Anderson 2002). We considered beta estimates with 95% confidence intervals overlapping 0 to have minimal impact on selection.

Nest Survival Analysis

We used the nest survival model in Program MARK (White and Burnham 1999) to estimate daily survival rate (DSR) of nests across the 4 years of our study. We defined nest survival as the probability of a nest surviving the 23-day incubation period (Rosene 1969). Our survival analysis consisted of 3 suites of covariates that we sequentially combined into a single analysis to assess treatment effects on nest survival. The first stage of analysis included nest age, nest initiation date, and time effects. Nest age was a dynamic variable and was coded as described by Dinsmore and Dinsmore (2007). Time effects were variation in survival that could have changed linearly from day 1 to day 136 of the nesting season. Next, we evaluated the treatment effect, using unit and year variables as terms additive to and in interaction with treatment. We considered year an experimental covariate because habitat manipulations were progressive and cumulative. We used results from the resource selection analysis to help guide model building activities related to the influence of habitat features and management activities. Again, management activities were evaluated only for nests in the treatment area. As a general guide for reducing the complexity of our model selection procedure, we selected the top model from each suite of covariates and used that model as a baseline for the subsequent suite based on AIC_c. Later, we re-assessed covariates that had some support from previous models to explore key relationships associated with our experimental framework that may have been masked earlier in our modeling process (Doherty et al. 2012). We used model averaging for all models with ΔAIC_c scores <2 to estimate daily survival rates and overall nest survival (Burnham and Anderson 2002). Nest age and nest initiation were dynamic variables, thus only choosing 1 value from these variables for model averaging would bias daily survival rate estimates. To account for this, we included daily survival rates from models where nest initiation was held constant (median date of 68) but nest age was dynamic, which resulted in 23 additional daily survival rate estimates. We also held nest age constant (median age of 12) but used nest initiation date as a dynamic variable, which resulted in 136 additional daily survival estimates. We considered effects to be strongly

supported if the 95% confidence interval of the beta estimate excluded 0. We used the delta method (Powell 2007) to expand our survival estimates to a temporal scale that encompassed the 23-day incubation period.

RESULTS

Nest Ecology and Nest-Site Selection

We captured and banded 1,002 bobwhite (445 M, 284 F, and 273 unknown sex) of which 655 were radio-marked and monitored during the 4 breeding seasons. We located 129 nests, 53 on control and 76 on treatment units. Females incubated 108 nests (84%) and males incubated 21 nests (16%). Of these 129 nests, 60 hatched (47%), 44 were destroyed (34%), 19 were abandoned before completion (15%), and 6 were unsuccessful because of adult mortality (5%). Of the 19 nests that were abandoned, 6 were abandoned presumably because of investigator presence and were excluded from any nest survival analyses. We collected vegetation data at 72 nest sites during the 2012 and 2013 breeding seasons combined (Table 1). Fifty-nine percent of nests were in open herbaceous, 16% in NWSG, 23% in shrub cover, and 2% in forest cover. Nests were constructed with a variety of material: 60% with cool-season grasses (predominantly field brome), 53% of nests included sericea lespedeza, and 29% included NWSG.

Clutch size on our study area averaged 12.5 ± 3.2 (SD) eggs and ranged from 2 to 19 eggs. The mean percent of eggs successfully hatching from a nest (hatching rate) was $84.3 \pm 2.9\%$. We did not observe males initiating a second nest, but 13.8% of females did initiate a second nest after a failed attempt (re-nesting). Double-clutching, or initiating a second nest after a successful nesting, occurred in 4.5% of females (1 F successfully hatched 2 nests). We found nests as early as 3 May and as late as 13 September with a median date of 22 June. Nesting activity peaked around 19 June with an average of 5.75 ± 1.43 (SE) active nests/year (Fig. 2).

We fit 15 models to estimate bobwhite macrohabitat nest-site selection. There were 3 models within $2 \Delta AIC_c$ of the top model (Table 2). We averaged beta estimates for variables included in the top 4 models: shrub-open edge density ($\beta = 0.0054$, 95% CI = -0.0002 to 0.0110), distance to NWSG ($\beta = -0.0027$, 95% CI = -0.0058 to 0.0004), distance to road ($\beta = -0.0020$, 95% CI = -0.0045 to 0.0005), and distance to shrub cover ($\beta = -0.0023$, 95% CI = -0.0076 to 0.0030). The confidence intervals for all variables overlapped 0, indicating these variables did not significantly contribute to nest-site selection. The top model for treatment areas only included the herbicide variable ($\beta = 1.312$, 95% CI = 0.071–2.554; Table 2) and the beta estimate did not include 0, indicating bobwhite selected areas treated with herbicide more than areas untreated with herbicide for nest sites. The top model for predicting microhabitat nest-site selection contained litter depth ($\beta = 0.489$, 95% CI = 0.009–0.968), woody stem density ($\beta = 0.0006$, 95% CI = -0.0003 to 0.0015), percent coverage of NWSG ($\beta = 2.34$, 95% CI = -0.15 to 4.83), and percent coverage of sericea lespedeza ($\beta = -1.24$, 95% CI = -2.69 to 0.21; Table 3), but only litter depth was informative indicating bobwhite selected areas with more litter than was available within the vicinity of the nest.

Nest Survival

We used 118 nests ($n = 63$ on Sinclair; $n = 55$ on Ken) in the nest survival analysis of which 73 (62%) were assigned to treatment and 45 (38%) to control depending on nest location. We found 5 nests that were not associated with a radio-marked individual and we censored these nests from the survival analysis because we did not check them daily.

The best model from our first stage of analysis included an interaction between nest age and nest initiation and we used this model as the baseline model (Table 4). Beta estimates and confidence intervals indicated that nest age ($\beta = 0.19$, 95% CI = 0.04–0.33) and the interaction term ($\beta = -0.003$, 95% CI = -0.006 to -0.0009) were supported, but nest initiation date ($\beta = 0.006$, 95% CI = -0.01 to 0.02) was not. The negative beta estimate for the interaction term suggested older nests had a higher probability of survival early in the breeding season, though they had a lower probability of survival as the breeding season progressed toward the end of the season (Fig. 3). The decrease in daily survival rates occurred during the last half of the breeding season (Fig. 3). No treatment or habitat covariates were supported based on beta estimates, and the models with low ΔAIC_c values were nested and likely had artificially inflated AIC values (Arnold 2010).

The model-averaged overall daily survival rate was 0.93 ± 0.02 (SE) and period survival was 0.19 ± 0.10 for the 23-day incubation period. However, daily survival rates were dynamic during our study because the top model contained an interaction between nest age and nest initiation date. Thus, daily survival rates ranged from $0.99 \pm < 0.01$ (older nests occurring early in the breeding season) to 0.08 ± 0.13 (older nests occurring at the end of the breeding season; Fig. 3).

Brood Ecology and Resource Selection

We tracked 89 adults with broods, including 59 individuals with broods from an identified hatched nest and 30 individuals that were captured with a brood. We recorded 1,277 brood locations for a mean of 14 locations/brooding adult. We collected vegetation data at 57 brood locations during the 2012 and 2013 breeding seasons, combined, representing 19 brooding adults (Table 1). We tracked broods as early as 23 May and as late as 30 September with a median date of 14 July. Brood-rearing activity peaked around 10 July with a mean of 8.3 ± 1.3 (SE) active broods per year (Fig. 2).

We fit 13 models to identify important variables for bobwhite brood macrohabitat resource selection, 4 of which accounted for 98% of the model weight (w_i; Table 5). The top model contained only distance to shrub cover, but the confidence interval for the beta estimate overlapped 0 (-0.0027, 95% CI = -0.006 to 0.001). We fit 11 additional models to assess the influence of management on selection. The top model contained 94% of the weight and included distance to a disked NWSG area, distance to a firebreak, and the herbicide variable (Table 5). The beta estimates indicated brooding adults were closer to disked NWSG areas

Table 1. Macrohabitat and microhabitat characteristics of northern bobwhite nest sites and brood locations compared to random locations, Peabody Wildlife Management Area, Kentucky, USA, 2010–2013.

| | Nest | | | | Brood | | | |
| | Location | | Random | | Location | | Random | |
Variable	$\bar{x}$	SE	$\bar{x}$	SE	$\bar{x}$	SE	$\bar{x}$	SE
Macrohabitat								
Shrub-open edge density (m/ha)	96.21	5.36	89.00	2.34	105.70	1.90	103.99	0.83
Forest-open edge density (m/ha)	4.02	1.24	4.73	0.62	2.62	0.25	2.66	0.12
Contagion (%)	53.54	1.10	53.96	0.54	54.51	0.32	54.25	0.14
Distance to forest (m)	290.80	15.23	289.87	7.05	308.72	4.89	306.22	2.18
Distance to shrub (m)	51.02	5.34	58.28	2.53	40.40	1.38	41.82	0.61
Distance to road (m)	191.12	19.82	205.60	8.54	217.27	5.34	219.65	2.40
Distance to open herbaceous (m)	21.50	3.75	26.41	1.75	36.34	1.31	35.47	0.60
Distance to native warm-season grass (m)	166.66	20.53	177.64	9.24	204.65	6.98	205.40	3.15
Distance to firebreak[a]	114.40	9.87	118.08	4.57	77.54	2.27	85.81	1.04
Distance to disked open herbaceous (m)[a]	144.46	20.21	148.09	8.77	77.47	3.56	81.24	1.58
Distance to disked native warm-season grass (m)[a]	244.29	33.44	249.98	14.85	258.39	10.14	263.59	4.57
Total disked area (ha)[a]	0.30	0.07	0.28	0.03	0.25	0.02	0.25	0.01
Microhabitat								
Litter depth (cm)	2.28	0.15	2.08	0.15	1.36	0.12	1.56	0.15
Sight tube (cm)					101.35	20.17	106.58	21.01
VO1[b]	0.43	0.09	0.27	0.08	0.32	0.07	0.38	0.08
VO2	0.53	0.10	0.29	0.08	0.48	0.09	0.48	0.09
VO3	0.67	0.12	0.38	0.08	0.70	0.11	0.74	0.12
VO4	0.99	0.14	0.71	0.09	1.21	0.16	1.22	0.15
VO5	1.57	0.14	1.58	0.14	1.82	0.19	1.89	0.17
VO6	2.90	0.17	3.04	0.18	2.89	0.21	2.92	0.21
VO7	3.94	0.46	4.10	0.48	3.89	0.19	3.87	0.20
VO8	4.861	0.04	4.86	0.05	4.62	0.13	4.63	0.12
Woody stem density (stems/ha)	398.1	97.75	215.9	82.87	520.73	197.78	487.21	201.95
Native grass (%)	15.19	3.27	7.205	2.29	4.68	1.90	4.33	1.69
Food plants (%)					25.20	3.65	27.60	3.63
Forb (%)	19.27	3.02	16.93	3.04	39.94	4.58	42.28	4.69
Low shrub (%)	3.559	1.87	2.431	1.42	3.98	1.21	3.74	1.16
Cool-season grass (%)	15.89	3.47	17.27	4.00	11.29	3.05	9.71	2.97
Total grass cover (%)	36.46	4.22	29.43	4.64	19.88	3.43	14.21	3.11
Sericea lespedeza (%)	63.98	4.02	75.35	4.21	59.36	5.04	53.92	5.12

[a] Averages for metrics related to management only include locations from treatment units.

[b] VO1 = visual obstruction from 1.75 m to 2 m aboveground, VO2 = visual obstruction from 1.5 m to 1.75 m aboveground, VO3 = 1.5–1.25 m, VO4 = 1.25–1 m, VO5 = 1–0.75 m, VO6 = 0.75–0.5 m, VO7 = 0.5–0.25 m, VO8 = 0.5–0 m.
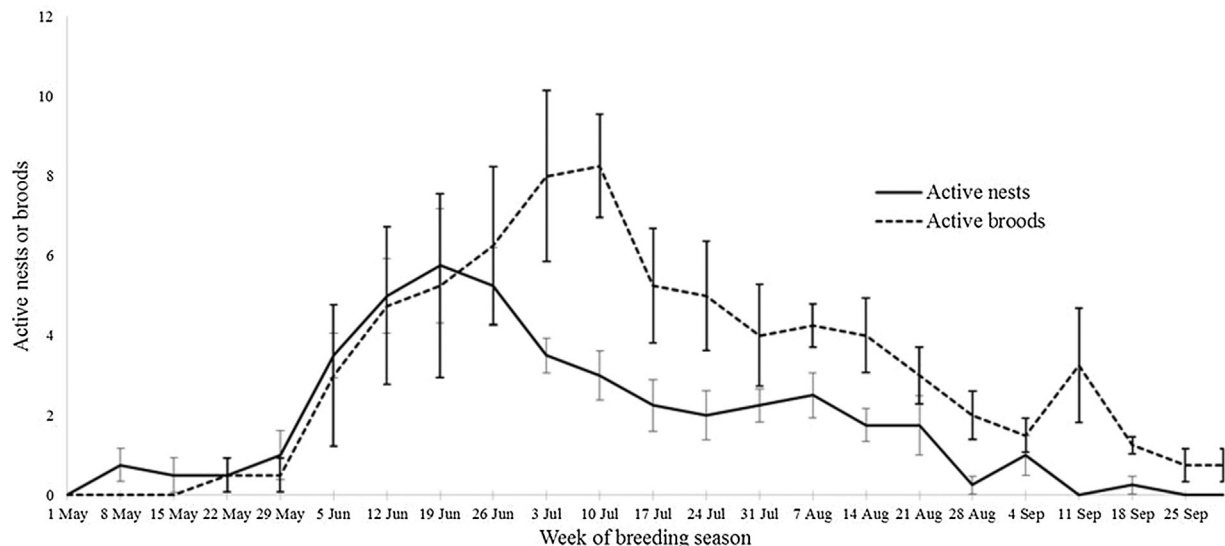

Figure 2. Northern bobwhite nesting and brood-rearing chronology based on average number of active nests or broods (±SE) for a given week during the breeding season, Peabody Wildlife Management Area, 2010–2013, Kentucky, USA.

Table 2. Competing models explaining the variation in macrohabitat nest-site selection for northern bobwhite on a reclaimed surface mine, Peabody Wildlife Management Area, Kentucky, USA, 2010–2013. Support for each model is indicated by the difference in corrected Akaike's Information Criterion (AIC$_c$) values (ΔAIC$_c$) and by Akaike model weights (w_i).

Model[a]	K[b]	log(L)[c]	AIC$_c$	ΔAIC$_c$	w_i
Control and treatment					
Shrub edge + nwsg + road	3	−225.99	458.01	0.00	0.20
Shrub edge + nwsg	2	−227.20	458.42	0.41	0.16
Shrub edge + nwsg + road + shrub	4	−225.61	459.28	1.27	0.10
Shrub edge	1	−228.99	459.99	1.98	0.07
Null	0	−231.14	462.27	4.26	0.02
Treatment					
Shrub edge + nwsg + road + herbicide	4	−122.18	252.45	0.00	0.56
Shrub edge + nwsg + road	3	−124.53	255.11	2.66	0.15

[a] Shrub edge = shrub-open edge density (m/ha), nwsg = distance to native-warm season grass (m), road = distance to road (m), shrub = distance to shrub cover (m), herbicide = unsprayed or sprayed with herbicide to control sericea lespedeza (0 or 1).
[b] K = number of model parameters.
[c] log(L) = negative log likelihood of each model.

(−0.0024, 95% CI = −0.004 to −0.001) and firebreaks (−0.0068, 95% CI = −0.010 to −0.004) than would be expected at random and were found in areas sprayed with herbicide more than areas untreated with herbicide (0.65, 95% CI = 0.338–0.962). Furthermore, the beta estimate for distance to firebreak was 2.8 times greater than the beta estimate for disked NWSG, indicating firebreaks were 2.8 times as important to brood resource selection. Probability of selection decreased 6.8% with every 10-m increase in the distance from a firebreak compared to a 2.4% decrease for disked NWSG. Microhabitat variables did not influence brood resource selection. The top model (AIC$_c$ = 76.66, w_i = 0.17) contained the litter depth variable (β = −0.59, 95% CI = −1.32 to 0.15), but the confidence intervals of the beta estimate overlapped 0. The null model was the next best model and was <2 ΔAIC$_c$ from the top model (ΔAIC$_c$ = 0.98, w_i = 0.10).

DISCUSSION

Nest-site selection and nest survival of northern bobwhite at Peabody were not influenced by the configuration of vegetation types at the scales we examined. Litter depth was the only supported variable for resource selection, and no microhabitat or macrohabitat variables influenced nest survival. Selection and survival may have been influenced by variables beyond what we measured. Our nest-site selection results indicated bobwhite were more likely to nest in areas

treated with herbicide to reduce sericea lespedeza compared to areas untreated with herbicide, but nest survival was not improved by habitat management. Bobwhite exploited a wide range of nesting substrates, indicating a high level of elasticity (i.e., slack; Guthery 1999) for nesting requirements. However, low nest survival and overall fecundity (i.e., clutch size, hatching rate, re-nest rate, double clutching) suggest there are problems associated with reclaimed mine land vegetation. These issues may be beyond concealment provided by vegetation and may be indicative of a nutritional limitation. In contrast, brood resource selection was strongly influenced by management activities, suggesting broods were attracted to areas altered by management, likely as a result of the change in vegetation composition and structure.

Nest-site selection has been studied extensively for bobwhite in multiple landscapes across their range (Klimstra and Roseberry 1975, Taylor et al. 1999a, White et al. 2005, Singh et al. 2010, Tanner et al. 2015). Klimstra and Roseberry (1975) reported 41% of nests occurred in idle fields, dominated by perennial grasses and forbs; our results were similar; 75% of nests were in open herbaceous or NWSG, which consisted of perennial grasses and forbs in an idle state. We likely did not see a difference between vegetation composition and structure between nest sites and available vegetation because the height and the structure of vegetation within a particular vegetation type were uniform and sufficient to facilitate bobwhite nesting. Our nest-site

Table 3. Competing models explaining the variation in microhabitat nest-site selection for northern bobwhite on a reclaimed surface mine, Peabody Wildlife Management Area, Kentucky, USA, 2010–2013. Support for each model is indicated by the difference in corrected Akaike's Information Criterion (AIC$_c$) values (ΔAIC$_c$) and by Akaike model weights (w_i).

Model[a]	K[b]	log(L)[c]	AIC$_c$	ΔAIC$_c$	w_i
Litter + stems + nwsg + sericea	4	−41.57	91.43	0.00	0.26
Litter + stems + nwsg + sericea + VO3	5	−40.78	92.00	0.57	0.19
Litter + nwsg + sericea	3	−42.94	92.06	0.63	0.19
Null	0	−49.91	99.81	8.39	0.00

[a] Litter = depth of litter (cm), stems = woody stem density (stems/ha), nwsg = % cover of native grasses, sericea = % cover of sericea lespedeza, VO3 = visual obstruction from 1.25 m to 1.5 m aboveground.
[b] K = number of model parameters.
[c] log(L) = negative log likelihood of each model.

Table 4. Highest rankings models (and the null model) based on differences in corrected Akaike's Information Criterion values (ΔAIC_c <2) and AIC_c weights (w_i) used to assess the influence of biological, microhabitat, and landscape metrics on northern bobwhite nest survival on Peabody Wildlife Management Area, Kentucky, USA, 1 April 2010–30 September 2013.

Model[a]	AIC_c	ΔAIC_c	w_i	Model likelihood	K[b]	Deviance[c]
Nest age × nest initiation	357.84	0.00	0.20	1.00	4	349.80
Nest age × nest initiation + litter depth	358.10	0.26	0.18	0.88	5	348.05
Nest age × nest initiation + stems	358.20	0.36	0.17	0.84	5	348.14
Nest age × nest initiation + unit	359.65	1.81	0.08	0.41	5	349.59
Nest age × nest initiation + year	359.66	1.83	0.08	0.40	5	349.61
Null[d]	368.80	10.96	0.00	0.00	1	366.80

[a] Woody stem density (stems/ha) covariate is represented as stems, unit = treatment or control unit.
[b] Number of parameters in each model.
[c] Deviance is the difference in −2ln(Likelihood) of the current model and −2ln(Likelihood) of the saturated model.
[d] Survival rate is constant.

selection results support the idea of plasticity in bobwhite nesting conditions (Guthery 1999). A study in a more arid environment than our study area reported bobwhite selected sites with increased grass cover (Townsend et al. 2001). Our results, along with others, suggest bobwhite are opportunistic nesters and will construct nests in a wide variety of plant groups, especially in temperate environments where herbaceous nesting material is not lacking (Klimstra and Roseberry 1975).

Our nest survival estimate (0.19, SE = 0.10) was lower than that of published estimates in southeast Iowa (0.28 and 0.50; Potter et al. 2011), northern Missouri (0.44; Burger et al. 1995), western Oklahoma (0.48; Cox et al. 2005), east-central Mississippi (0.40; Taylor and Burger 1997), southern New Jersey (0.45; Collins et al. 2009), southern Texas (0.38; Rader et al. 2007), and Florida (0.41; Rolland et al. 2010). Mean clutch size (12.5 ± 3.2) was slightly lower than the 14.4 and 13.7 reported by Stoddard (1931) and Roseberry and Klimstra (1984), respectively. Similarly, mean hatching success (85.5 ± 2.9%) was slightly lower than the population average reported by Sandercock et al. (85–95%; 2008), and our estimates of re-nesting and double-clutching were exceptionally low; no males and only 13.8% of females (n = 12) were known to re-nest after a failed nesting attempt.
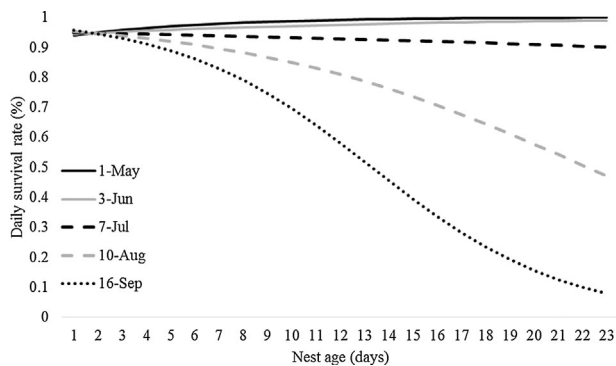

Figure 3. Daily nest survival rates for northern bobwhite from the top model of nest survival analysis using Program MARK (nest age × nest initiation) plotted with nest initiation starting from 0% (black line), 25% (grey line), 50% (black dashed line), 75% (grey dashed line), and 100% (black dotted line) through the breeding season. Data are from Peabody Wildlife Management Area, 2010–2013, Kentucky, USA.

Our re-nesting rate was less than half of what was reported in a study from Florida (0.28; Rolland et al. 2011) and below the range reported by Sandercock et al. (25–69%; 2008).

Vegetation characteristics consistent with improved nest concealment are commonly selected by bobwhite; however, these features rarely have been associated with improved nest survival. For example, Taylor et al. (1999b), Townsend et al. (2001), Lusk et al. (2005), and Collins et al. (2009) reported vegetation characteristics consistent with improved nest concealment (i.e., increased visual obstruction, increased shrub cover, taller vegetation) were important to bobwhite nest-site selection, but only Lusk et al. (2005) linked the improved nest concealment (i.e., taller vegetation, increased shrub cover) with nest survival. Although woody stem density was included in a competing model for nest survival, neither nest-site selection nor survival at Peabody was influenced by characteristics associated with nest concealment, suggesting predation may be the proximate, but not ultimate, cause of reduced nest survival.

One of many possible explanations for low nest survival and fecundity at Peabody could be a nutritional limitation for adult bobwhite. Nutritional limitations resulting in reduced fecundity have been reported in multiple species of birds (Nagy and Holmes 2004, Zanette et al. 2006, Haley and Rosenberg 2013) and Giuliano et al. (1996) reported nutritional limitations may lead to reproductive failures in pen-raised bobwhite. Bobwhite actively defend their nest from predators (Ellis-Felege et al. 2013) and increasing the amount of time spent away from the nest foraging may increase vulnerability of nests. Furthermore, a nutritional limitation may reduce the ability of birds to re-nest after a failed attempt or produce a second clutch (Newton 1998, Nagy and Holmes 2004). This has important implications when considered in the context of the interaction term of nest age and nest initiation on nest survival (Fig. 3). The opportunity for confounding effects exist, in which re-nesting inherently occurs later in the breeding season, when nests have lower survival rates (Fig. 3). When this trend is coupled with relatively fewer re-nesting attempts because of nutritional limitations, bobwhite recruitment could be severely limited. Finally, reduced fecundity during our study may have been related to a low proportion of females attempting to initiate a first nest. The proportion of females

Table 5. Competing models explaining the variation in brood macrohabitat resource selection for northern bobwhite on a reclaimed surface mine, Peabody Wildlife Management Area, Kentucky, USA, 2010–2013. Support for each model is indicated by the difference in corrected Akaike's Information Criterion (AIC$_c$) values (ΔAIC$_c$) and by Akaike model weights (w_i).

Model[a]	K[b]	log(L)[c]	AIC$_c$	ΔAIC$_c$	w_i
Treatment and control					
Shrub	35	−2,246.17	4,363.43	0.00	0.32
Shrub + forest	36	−2,245.80	4,563.68	0.25	0.28
Shrub + forest + road	37	−2,245.37	4,564.24	0.81	0.21
Shrub × unit	36	−2,246.08	4,564.74	1.31	0.17
Null	0	−2,288.08	4,576.15	12.72	0.00
Treatment					
Shrub + disked nwsg + firebreak + herbicide	50	−1,426.91	2,953.90	0.00	0.94
Shrub + disked nwsg + firebreak + herbicide + disked area + disked oh	70	−1,408.54	2,960.10	6.15	0.04
Shrub	27	−1,478.06	3,009.20	55.27	0.00
Null	0	−1,508.66	3,017.30	63.39	0.00

[a] Shrub = distance to shrub cover (m), forest = distance to forest cover (m), road = distance to road (m), unit = treatment or control unit, disked nwsg = distance to disked native warm season grass cover (m), firebreak = distance to firebreak (m), herbicide = unsprayed or sprayed with herbicide to control sericea lespedeza (0 or 1), disked area = total disked area (ha), disked oh = distance to disked open herbaceous cover (m).
[b] K = number of model parameters.
[c] log(L) = negative log likelihood of each model.

initiating ≥1 nest during our study ranged from 54% to 72.7% across years (Tanner 2012, Peters 2014). The proportion of females that attempt to lay ≥1 clutch can influence productivity in bobwhite populations (Guthery 2000, Hernández and Peterson 2007). If it is assumed that all females in a population may attempt to initiate ≥1 nest (Guthery 2000), the estimated initiation rates observed during our study could account for the reduced fecundity in our population. However, the relationship between nutritional limitations and nest initiation rates is unknown for bobwhite and offers an opportunity for future research.

Although food is rarely considered a limitation to wild bobwhites (Guthery 1997), reclaimed mine lands present a unique situation because of the extensive coverage of sericea lespedeza. Extensive stands of sericea lespedeza may limit the abundance of invertebrates compared with other forbs (Bugg and Dutcher 1989) and limit the establishment of native herbaceous vegetation (Adams et al. 1973, Wade 1989, Foster and Gross 1998). Invertebrates and seeds are important components of adult bobwhites' diets during the breeding season, and invertebrates are especially important for female bobwhite (Brennan and Hurst 1995). Crop content data from bobwhite harvested during the hunting season in our study area have indicated the diversity of seeds in the crops has increased since initiating habitat management for our study (E. S. Williams, Kentucky Department of Fish and Wildlife Resources, unpublished data).

The selection of nest sites in areas treated with herbicide to reduce sericea lespedeza may be evidence that these treatments improved conditions by improving food availability. Although experimental variables did not directly influence nest survival, disking and herbicide application are effective in reducing sericea lespedeza and increasing the coverage of plants important as bobwhite summer food (Brooke et al. 2015). Additionally, these areas were selected by adult bobwhite (Brooke et al. 2015) and brood-rearing bobwhite during the breeding season and breeding season survival was greater on treatment units compared to control units (Peters et al. 2015). This may explain why >60% of nests were on treatment units compared to control units, despite control units being 108 ha larger than treatment units. The overall positive relationship between bobwhite use and survival with management aimed at reducing sericea lespedeza supports the continuation of these practices, regardless of its insignificant effect on nest survival.

We expected broods to select areas with increased openness at ground level (Doxon and Carroll 2010) and an increased amount of forbs or bobwhite food plants (Martin et al. 2009) because these conditions improve chick mobility and foraging (Taylor et al. 1999a, Collins et al. 2009, Martin et al. 2015). However, our brood microhabitat analysis did not detect differences between these variables within the same vegetation type. Our inability to detect differences in brood microhabitat selection was likely a result of the homogeneity of vegetation within the vegetation types where broods were located. However, openness at ground level at brood locations (Table 1) overlapped openness at ground level within disked areas and areas treated with herbicide (91.2 ± 7.4 cm and 82.2 ± 7.1 cm, respectively; Brooke et al. 2015), indicating disking and herbicide application were effective at producing the structure used by broods.

The estimated slope for distance to firebreaks was >2 times greater than the estimate for distance to disked NWSG, suggesting broods selected firebreaks more than blocks of disked vegetation. Broods likely were attracted to firebreaks more than disked areas because of the proximity to shrub cover and the structure in the firebreaks. Firebreaks were disked annually and were dominated by annual plants (e.g., common ragweed) with sufficient bare ground and overhead cover to facilitate foraging (Taylor et al. 1999a, Collins et al. 2009). Moreover, desiccated wheat stems with seedheads remained on the firebreaks, providing an additional food source for broods and adults. Disked areas, on the other hand, were treated on a 3-year interval and were dominated by sericea lespedeza by the third growing season (Brooke et al. 2015).

We hypothesized prescribed fire would have been an effective management tool to improve the cover for broods and adult bobwhite, given its importance in setting back succession and maintaining brood-rearing cover in other landscapes (Brennan et al. 2000, Jones and Chamberlain 2004). However, prescribed fire did not influence brood resource selection at Peabody. Moreover, burned areas were avoided by non-nesting and non-brood rearing bobwhite during the breeding season (Brooke et al. 2015). Prescribed fire is an effective tool to set back the successional trajectory in multiple landscapes and promote annual and perennial forbs creating an open structure important to bobwhite (Gruchy and Harper 2014). However, vegetation composition on reclaimed mine lands may limit the effectiveness of prescribed fire as a management option. Fire stimulated growth and germination of sericea lespedeza, which led to similar cover of sericea lespedeza in burned areas compared to untreated areas (Brooke et al. 2015). As a result, prescribed fire did not promote the open structure at ground level that would facilitate movement and foraging for chicks (Collins et al. 2009) and may have reduced shrub cover within open areas that is important to adult bobwhite throughout the year.

MANAGEMENT IMPLICATIONS

We suggest disking and herbicide application to reduce sericea lespedeza and to enhance the quality of open areas for bobwhite on reclaimed mine land. Herbicide applications were effective in reducing sericea lespedeza and improving nesting and brood-rearing cover, but disking also reduced sericea lespedeza and densely planted NWSG and promoted more desirable plant species. Both practices only temporarily reduced sericea lespedeza and treatments must be conducted on a short-return interval (i.e., <3 yr). Linear habitat features that are disturbed regularly, in our case firebreaks, also can make a valuable contribution to improved cover for broods. Disking should be conducted closer (<50 m) to woody cover to improve its value for bobwhite. We suggest discontinuing prescribed fire as a stand-alone practice in open areas dominated by sericea lespedeza. We suggest future research to evaluate the nutritional status of bobwhite on reclaimed surface mines. Additionally, future reclamation projects should use native species at appropriate rates rather than non-native species. Native species can fulfill mine reclamation requirements for erosion control without degrading habitat quality for bobwhite and other wildlife species (Yeiser et al. 2016). Although reclaimed mine lands provide marginal habitat for bobwhite, conditions can be improved and reclaimed mine lands can make a valuable contribution to bobwhite conservation.

ACKNOWLEDGMENTS

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