Population Ecology of Northern Bobwhite (*Colinus virginianus*) on a Reclaimed Surface Mine

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ABSTRACT

Direct linkages between habitat management and northern bobwhite (*Colinus virginianus*) survival are not well documented; therefore, we implemented an experiment to evaluate those responses. We conducted our experiment on a reclaimed surface mine, a novel landscape where conditions were considered sub-optimal for bobwhite. Nonetheless, these areas have great potential for contributing to bobwhite conservation. Our objectives were to determine if habitat management could improve (1) seasonal and (2) nest survival, how (3) multi-scale habitat contributed to seasonal and nest survival, and (4) conduct life stage simulation analyses (LSA) to determine which vital rates were affecting population growth rate. Research was conducted on Peabody Wildlife Management Area in western Kentucky. Two units of the site (Sinclair and Ken, 1471 and 1853 ha, respectively) served as replicates and were each randomly divided into a treatment (disking, burning, herbicide application) and control. Treatments were applied October 2009 - September 2013. We detected evidence that treatments may have improved summer survival (Part II). However, we found no evidence that treatments had an impact on nest survival (Part III). Among habitat covariates, litter depth (β [beta] = -0.387, CI = -0.5809, -0.1930) was the most influential covariate on survival (Part II). Pooled seasonal survival rates differed between winter ($S = 0.281$, SE = 0.022) and summer ($S = 0.148$, SE = 0.015). Nest survival (0.352 ± 0.037, 23-day period) was low compared to other studies and was not related to habitat (Part III). Instead, nest age ($\beta = 0.641$, CI = 0.372-0.911) and nest initiation date ($\beta = 0.022$, 95% CI = 0.001-0.043) influenced (positive) nest survival. Our LSA revealed that clutch size (r^2 [coefficient of determination] = 0.384), followed by hatching success ($r^2 = 0.207$), and nest survival ($r^2 = 0.141$) explained most variation in λ [lambda] (Part IV). Total fecundity explained 94% of the variation in λ. It appears that summer survival and elements of fecundity may be limiting factors on our site. Additional experiments across a wider range of habitat

conditions may be required to determine management intensity and duration thresholds required to elicit greater changes in survival for bobwhite populations.

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PART I

INTRODUCTION

The northern bobwhite (*Colinus virginianus*; hereinafter, “bobwhite”) is an important game bird that is intensively managed for hunting recreation in the southeastern United States. However, populations have been declining for much of the last 40 years (Brennan 1991). The Breeding Bird Survey (BBS) has documented a population decline, with an annual decrease of 4.0% in the United States from 1966 to 2010, and within Kentucky, an annual decrease of 2.9% during that same period (Sauer et al. 2011). Hypotheses thought to explain this decline include reduction in fledgling survival as a result of predation by imported fire ants (Mueller et al. 1999), loss and fragmentation of habitat (Fleming and Giuliano 2001), extreme weather events, such as drought and flooding (Lusk et al. 2001, Hernandez et al. 2005), and hunting pressure (Madison et al. 2002, Guthery et al. 2004).

Although all of these factors may influence such declines, it has become clear that the major causative factor is a loss of useable space (Guthery 1997, Brady et al. 1998, Veech 2006). Habitat loss can be attributed to clean farming practices combined with silvicultural practices that increase tree density within stands (Brennan 1991, Twedt et al. 2006). Another factor influencing the degradation of bobwhite habitat is the decreased use of prescribed burning (Twedt et al. 2006). Prescribed burning promotes early successional communities and increases plant species richness (Collins 1987), that contribute to bobwhite population viability (Stoddard 1931, Greenfield et al. 2003). Additionally, much of the native grass component in the southeastern United States has been replaced by non-native grasses, such as tall fescue (*Schedonorus phoenix*) and bermudagrass (*Cynodon dactylon*). Along with intensive silvicultural and agricultural practices, an increase in urbanization has led to the loss and substantial fragmentation of early successional plant communities (Terhune et al. 2007) essential for sustaining bobwhite populations at a landscape scale. The course of action required to reverse such declines is creation and maintenance of early successional native plant communities on a landscape scale (Williams et al. 2004).

One potential opportunity for increasing bobwhite habitat in portions of the eastern United States is management of reclaimed surface mine sites. Indeed, a considerable amount of early successional vegetation has been established in the eastern United States under the support of the Surface Mining Control and Reclamation Act of 1977 (SMCRA). This legislation was enacted to minimize the impact of surface mining on wildlife populations, unique vegetation types, and other important environmental elements. Under this act, surface mines may qualify for reclamation plans that provide for environmental improvement following mining. This has led to the reclamation of over 600,000 ha in the eastern United States, of which 269,000 ha are in Kentucky (Table 1.1), since the passage of SMCRA. These reclamation efforts have resulted in early successional vegetation communities that have the potential to support quail populations at a very minimal direct cost to the conservation community.

However, surface mine reclamation success has been assessed only in the short-term (e.g., <5 years) and during cover establishment, plant species diversity has been a lower priority than prevention of soil erosion (Holl 2002). Thus, establishment of dense stands of sericea lespedeza (*Lespedeza cuneata*) and other non-native herbaceous species is common on these reclaimed mine sites to quickly provide cover that minimizes soil erosion. Sericea lespedeza is an aggressive perennial legume that out-competes native grasses and is less palatable and nutritious than native species because of its high tannin and lignin content at maturity (Blocksome 2006). In addition, hard-seeded plant foods, such as sericea lespedeza, are virtually indigestible (Davison 1958). Sericea lespedeza seeds simply overwhelm the seedbank and can remain viable for up to 30 years (Fitzgerald et al. 2004). These characteristics have led to this forb being classified as an exotic plant of management concern by the Southeast Exotic Pest Plant Council (Eddy et al. 2003). On reclaimed mine sites, soil is often poor-quality and heavily compacted. The ability of sericea lespedeza to become established and be competitive in a variety of soil types (Ohlenbusch 2007) has also contributed to its domination of reclaimed mine sites.

Although research regarding bobwhite responses to reclaimed mine sites is lacking, studies have shown such sites provide habitat for other early succession specialists (Allaire 1978, Whitmore and Hall 1978, DeVault 2002, Karo 2009). Although non-native species have been effective in reducing erosion, the resulting plant community can be unfavorable for bobwhite (Eddy 1999). Eddy (1999) concluded that invertebrate and vertebrate species declined by 73 and 55% respectively in sericea lespedeza sites. High stem density, lack of singing perches, fewer canopy openings and a decrease in the seasonal availability of foliage, flowers, seeds, and prey insects were given as factors in the decline. Management practices must be focused on removing this unfavorable vegetation and promote the growth of more desirable native plant species.

Table 1.1. Eastern US coal-mined land area (ha) reclaimed under SMCRA, 1978-2005[a].

State	Phase III Released	Phase I Released (2001-2005)[b]	Total
E KY[c]	243,533	26,094	269,627
MD	5,372	118	5,490
OH	74,167	9,495	83,662
PA	93,670	13,359	107,029
TN	14,962	2,946	17,908
VA	37,076	1,125	38,201
WV	93,685	11,673	105,358
Total	562,465	64,810	627,275

[a] Including the interim SMCRA program. Source US OSMRE “20th Anniversary of the Surface Mining Law” (http://www.osmre.gov/annivrep.htm) and annual reports to Congress.
[b] As reported by states to OSMRE; these figures overestimate total affected areas due to double-counting of areas that were both mined and re-mined under SMCRA.
[c] Estimated from total Kentucky areas, as proportionate to the east-west distribution of surface coal tonnage.

Most studies monitoring bobwhite population dynamics have been concentrated in two regions, the Great Plains and the southeastern Coastal Plain. Studies in the western Great Plains have evaluated macrohabitat influences on bobwhite summer survival (Taylor et al. 1999), chick survival (DeMaso et al. 1997), over-winter habitat use and winter survival (Williams et al. 2000, Williams et al. 2004), nest success (Potter et al. 2011), and population responses to habitat management (Webb and Guthery 1982). Cox et al. (2004) also evaluated survival and mortality of bobwhites within this region. Numerous bobwhite studies have been conducted in the Red Hills region of the Gulf Coastal Plain and have evaluated effects of radio-telemetry research on bobwhite survival (Terhune et al. 2007), over-winter survival in relation to landscape composition (Holt et al. 2009), population dynamics based on banding (Pollock et al. 1989), and assessment of bobwhite population levels based on hunting success (Palmer et al. 2002). Burger et al. (1998) evaluated bobwhite survival and cause-specific mortality within this region in an intensively-managed plantation setting. Also working within this region, Sisson et al. (2009) evaluated bobwhite survival and analyzed causes of mortality. Dixon et al. (1996), though not working in the Red Hills, examined winter bobwhite survival and habitat use in a pine-dominated Coastal Plain system in South Carolina. Within the Sandhills of North Carolina, studies have included survival of bobwhites on hunted vs. non-hunted areas (Robinette and Doerr 1993) as well as documenting seasonal survival and cause-specific mortality (Curtis et al. 1988). Singh et al. (2010*a*) conducted a study in south Florida examining whether nest-site selection influences nesting success.

A smaller number of studies have been conducted in the Midwest and have examined effects of hunting pressure on survival rates (Suchy and Munkel 2000), population dynamics related to weather parameters and hunting pressure (Stanford 1972), and detailed ecology of localized bobwhite populations (Roseberry and Klimstra 1984, Burger et al. 1995). Roseberry and Klimstra (1984) conducted an intensive 26-year banding study in Illinois that assessed survival, cause-specific mortality, fecundity, and hunting effects on bobwhite survival. In Missouri, Burger et al (1995) evaluated bobwhite survival and cause-specific mortality. A few studies have been conducted at the northern edge of the bobwhite range (Janke 2011, Lohr et al. 2011). In southern New Jersey, for example, Lohr et al. (2011) conducted a two-year study on home range, movement, and habitat selection and their effect on survival. Janke (2011) investigated survival and habitat use during the non-breeding season on four private-land study sites in southwestern Ohio.

Despite the large amount of research on bobwhite population ecology, the preponderance of it has been conducted in areas with dramatically different climates, landscape contexts, and management. Few studies have examined northern bobwhites in the Mid-South region. Dimmick

(1971) assessed the influence of controlled burning on nesting patterns. His results suggested controlled burning of nesting cover had a measurable impact upon the distribution of quail nests and the chronology of nesting. Also, bobwhite quail nesting (Dimmick 1968), aspects of winter behavior (Yoho 1970), habitat use by quail during winter (Yoho and Dimmick 1972), and winter survival as a function of landscape composition has been assessed in western Tennessee (Seckinger et al. 2008). Extensive population dynamics studies within the Mid-South region are lacking.

Regardless of location, most studies have focused on survival, mortality, and fecundity of bobwhites, but have not related the effects of habitat to these parameters. Pollock et al. (1989) assessed survival rates of male and female bobwhites at Tall Timbers Research Station in Northwest Florida. They found adult quail had slightly higher survival rates than juveniles and that males had a higher rate of mean annual survival than females. Although this study was conducted over a 15-year period, it was based on band recovery and did not use radio-telemetry. Likewise, a 26-year study by Roseberry and Klimstra (1984) looked in detail at the population ecology of northern bobwhites in Illinois. This long-duration study included analysis of survival, mortality, fecundity, and hunting effects on populations. Roseberry and Klimstra (1984) reported typical survival rates of 53–75% for the first 16 weeks post-hatching. They also suggested that because of the relationship between fecundity and recruitment, juvenile survival might play a secondary role in determining autumn population size. However, the effect of juvenile survival on recruitment and autumn population size is still considerable. Based on the results of this study, bobwhites were thought to be heading toward extirpation in southern Illinois. Although this study provided extensive knowledge about bobwhite population ecology, results were based on band recovery and not radio-telemetry. Burger et al. (1995) used radio-telemetry to estimate both seasonal and annual survival of bobwhites in Missouri. This study had a large sample size of radio-collared birds (n = 1,001) and assessed survival for both sexes, but did not relate survival and mortality rates to habitat parameters. The influence of radio-collars on survival of bobwhites was assessed in Georgia (Terhune et al. 2007) during a long-term study (8 years) with a sample size of more than 6,000 birds banded and more than 2,000 birds collared. They found no evidence that radiotransmitters affected bobwhite survival. Survival and mortality were assessed for both collared birds and banded birds, but these estimates were not related to habitat quality. Sisson et al. (2009) assessed survival and cause-specific mortality of bobwhites on 13 separate areas in Georgia and Alabama, collaring 7,105 birds. Survival was assessed for both winter and summer seasons for males and females, but habitat quality was not related to these estimates. Burger et al. (1998) also assessed survival and cause-specific mortality of males and females among both seasons in Georgia, with a sample size of 813 collared birds. They concluded variation in annual survival among years was heavily dependent on overwinter mortality, but also did not assess habitat parameters pertinent to survival. In Oklahoma, Cox et al. (2004) also assessed survival and mortality on 2,647 collared birds for both seasons and both sexes. Cox et al. (2004) noted low survival for populations during the winter season, but did not relate habitat to survival.

Studies relating bobwhite population dynamics to habitat parameters have been focused at both the local and landscape levels in a number of ecological regions. Williams et al. (2000) assessed survival and habitat use of bobwhites on cropland and rangeland ecosystems in east-central Kansas over a three-year period and found woody escape cover greatly influenced winter movement and survival. Although radio-telemetry was used on 157 birds, survival was only assessed for the hunting season. Habitat use and selection was assessed, along with survival, on a

South Carolina plantation over two years (Dixon et al. 1996). Agricultural fields were selected, but the analysis included only 71 collared birds and did not include summer survival. Singh et al. (2010*b*) assessed habitat use by bobwhites in south Florida over a five-year period and found winter home ranges were generally larger than summer home ranges. They also found food plots (i.e., *Sesbania* spp.) were selected by both sexes and during both seasons. Although radio-telemetry was used, survival was not assessed and related to habitat use. Veech (2006) found declining bobwhite populations were more likely in forested and urban landscapes, whereas growing populations were more likely in agricultural and rangeland landscapes. Holt et al. (2009) analyzed over-winter survival in relation to landscape composition and structure within Mississippi during a two-year study. Landscape parameters were related to survival for 167 birds. Mean patch size and edge density of linear herbaceous composition (field borders and cover strips) were negatively related to survival. There was no strong evidence to suggest that seasonal ranges differed quantitatively in their survival benefits in relation to the composition or structure of the habitat within them (Holt et al. 2009). Likewise, winter survival was assessed and related to landscape composition in Tennessee (Seckinger et al. 2008). Treatments of landscape vegetation included converting 33% of closed canopy forests into early successional vegetation. Survival was assessed on 929 birds over four years. The edge density of closed-canopy forests was the most influential (negative) factor for survival. Potter et al. (2011) found nest survival was strongly associated with increased forb canopy cover. A decrease in daily nest survival was more severe on land dominated by agricultural production than on publically owned land managed for bobwhite. Greater nest survival on publically owned land was coincident with greater availability of presumed nesting cover, such as grassland and roadside. Only percent forb cover on the site dominated by agricultural production had a significant positive influence on daily survival rate. Successful nests had twice the average forb cover (38%) as unsuccessful nests (19%). Macrohabitat effects on bobwhite population dynamics were assessed in eastern Kansas on two study areas; one dominated by rangeland and the other by cropland. This study assessed habitat selection, composition, and distribution and related it to survival of males and females (Taylor et al. 1999). Survival of nesting and brood-rearing birds was also related to macrohabitat effects. Among the two study areas, availability of breeding (i.e., grassland) vegetation had no effect on female survival and was not determined for male survival because of sample size. Although this study was conducted over four years, only 267 collared birds were used in analysis and only summer survival was assessed. In New Jersey, Lohr et al. (2011) used habitat parameters to assess home range, movement, and habitat selection. They found differences in habitat selection occurred between seasons, with lower use of shrub-scrub and forest during the breeding season. Lohr et al. (2011) also found the risk of breeding season mortality was increased by longer daily movement, lower use of grassland, and more use of forest. Risk of mortality during the nonbreeding season increased with shorter daily movement and proximity to occupied buildings and barns. Although this study was conducted year-round during the breeding and nonbreeding seasons, the effects of habitat manipulation were not considered.

There have been few attempts to examine the sensitivity of key vital rates on population growth and decline for bobwhites (Sandercock et al. 2008, Gates et al. 2012, Williams et al. 2012). Sandercock et al. (2008) used life-stage simulation analyses (LSA) to examine sensitivity of population growth rates (λ) to demographic parameters, which included investigating the relationship between fecundity, nesting attempts, and survival. They reported overwinter survival explained the largest amount of variation in annual population growth rates for declining bobwhite populations across their range. Likewise, Gates et al. (2012), who conducted LSA using empirical data collected on 8 private lands sites in southwestern Ohio during 2008-2011, found non-breeding season survival was the most limiting population vital rate. In their study, fecundity rates and chick survival modeled with survival estimates explained 23.5% and 1.5%, respectively, of the variation in λ. Similarly, Williams et al. (2012) conducted LSA on data collected from 3-year study of a population of bobwhite in New Jersey. They compared local demographic parameters of bobwhite to a national compilation of demographic parameters (Sandercock et al. 2008) and reported both summer and winter survival made the greatest contribution to variance in λ. However, their local model showed that annual variation in components of fecundity had a large effect on variation in λ, including the number of young produced per nest that survived 30 days ($r^2 = 0.53$), nest success, ($r^2 = 0.20$), and clutch size ($r^2 = 0.18$) (Williams et al. 2012).

Although there have been studies monitoring populations of passerines (Whitmore and Hall 1978, Wray et al. 1982, DeVault 2002) and game birds (Karo 2009) that use early successional plant communities, few have monitored bobwhite populations with regard to reclaimed strip mines. In southwest Virginia, Stauffer (2011) conducted a pilot study to evaluate the potential of reclaimed mines as habitat for bobwhite quail. He found reclaimed mine lands supported poor to moderate quail numbers; however, reclaiming approaches were focused more on preventing runoff rather than providing bobwhite habitat. Habitat use for bobwhites in both breeding and non-breeding seasons needs to be evaluated on reclaimed sites to determine associated survival and fecundity rates. It is also important to assess bobwhite population responses to large-scale habitat management efforts, as bobwhite home ranges are highly variable and often depend on habitat composition and individual reproductive status (Brennan

1991). This work is especially needed because most habitat management efforts for bobwhite on reclaimed strip mines involve killing the existing cover (nonnative species) and promoting more native species. Studies have focused management efforts at a relatively small scale, and research must assess large-scale habitat management and its effects on bobwhite populations. The temporal scale also can be influential in population responses to habitat management and should be evaluated.

This research is a continuation of a study initiated in 2009 to evaluate bobwhite population dynamics on a reclaimed surface mine site in western Kentucky. My objectives were to (1) investigate the effects of habitat manipulation (treatments) on bobwhite survival and reproductive success; (2) document these influences at three different scales (home range, landscape, and microhabitat); (3) document nest success and nest productivity as a function of habitat parameters at two scales (microhabitat and landscape); and (4) develop a life stage analysis (LSA) for the study site based on data collected on the site to examine sensitivity of population growth/decline to demographic parameters (survival and fecundity vital rates) that might be important for management purposes. In Part II, we assessed survival rates as a function of habitat parameters during the winter and summer seasons at the home range, landscape, and microhabitat scales. In Part III, we assessed nest success as a function of habitat parameters at the microhabitat and landscape scales. In Part IV, we used LSA to conduct a thorough demographic analysis and evaluate the primary vital rates affecting bobwhite population growth rates (λ). Parts II, III, and IV are written as individual, stand-alone manuscripts for future publication.

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PART II

IMPACT OF EXPERIMENTAL HABITAT MANIPULATION ON NORTHERN BOBWHITE SEASONAL SURVIVAL ON A RECLAIMED SURFACE MINE

ABSTRACT In order to be successful, habitat management for northern bobwhite (*Colinus virginianus*) needs to impact vital rates, but direct linkages with survival are not well documented; therefore, we implemented an experiment to evaluate those responses. We conducted our experiment on a reclaimed surface mine, a novel landscape where conditions were considered sub-optimal because of non-native vegetation, such as sericea lespedeza (*Lespedeza cuneata*) and tall fescue (*Schedonorus phoenix*), that has been reported to provide poor habitat for northern bobwhite and may lead to reduced seasonal survival. Nonetheless, these areas have great potential for contributing to bobwhite conservation. Our study site was a 3,330-ha reclaimed surface mine, Peabody WMA, in western Kentucky. Two units of the site (Sinclair and Ken, 1471 and 1853 ha, respectively) served as replicates and were each randomly divided into a treatment (disking, burning, herbicide application) and control. Treatments were applied October 2009 - September 2013. We used radio telemetry to monitor northern bobwhite (n = 1,198) during summer (1 Apr-30 Sep) and winter (1 Oct-31 March), 2009 – 2013. We used the known fate model in Program MARK to determine if treatments had an impact on seasonal survival rates. We also included group, home range, landscape, and microhabitat metrics as covariates to help improve sensitivity and further elucidate experimental impacts. Survival varied annually, ranging from 0.139 (SE = 0.031) to 0.301 (SE = 0.032), and seasonally (summer, 0.148 [SE = 0.015]; winter, 0.281 [SE = 0.022]). Our treatment effect was included in the top model and had a positive effect (β = 0.256, CI = 0.057, 0.456) on survival, but interacted with season (positive in summer and negative in winter compared to control). Among habitat covariates, litter depth (β = -0.387, CI = -0.5809, -0.1930) was the most influential, negatively effecting survival. Additional experiments across a wider range of habitat conditions may be required to determine

management intensity and duration thresholds required to elicit greater changes in survival for northern bobwhite populations.

The northern bobwhite quail (*Colinus virginianus*) (hereinafter, bobwhite) is an important game bird that has been declining for much of the last 40 years (Brennan 1991). Throughout the bobwhite's range there has been a 3.8% decline annually (Sauer et al. 2011). Significant local declines have also been reported. In the Central Hardwood Bird Conservation Region (CHBCR), there has been a 4.0% decline annually from 1966-2010 and a 4.9% decline since 2000 (Sauer et al. 2011). There are many hypotheses thought to explain this decline such as land use change (Brennan 1991), extreme weather (Lusk et al. 2001, Hernandez et al. 2005), predation (Mueller et al. 1999, Palmer et al. 2005, Staller et al. 2005), hunting pressure (Madison et al. 2002, Guthery et al. 2004), and loss and fragmentation of habitat (Fleming and Giuliano 2001). Although all of these factors may influence such declines, it has become clear that the major causative factor is a loss of habitat (Guthery 1997, Brady et al. 1998, Veech 2006).

Numerous studies have assessed the impact of management practices on habitat responses (Dimmick 1971, Carver et al. 2001, Greenfield et al. 2003, Singh et al. 2010, Crosby et al. 2013), but few have assessed the direct impact of such manipulations on bobwhite survival (Seckinger et al. 2008). Dimmick (1971) showed that controlled burning of nest habitat had a measurable impact upon the distribution of quail nests and the chronology of nesting. Carver et al. (2001) compared the vegetative response in seasonal timing of disking and found that fall disking promoted more desirable plant species for bobwhite than spring disking. Greenfield et al. (2003) found that fall disking increased percent bare ground and plant diversity and decreased percent litter cover and litter depth while burning increased plant diversity and improved quality of habitat for bobwhite. Singh et al. (2010) assessed habitat use by bobwhites in south Florida over a five-year period and found food plots (i.e., *Sesbania* sp.) were selected by bobwhite during both winter and summer. Crosby et al. (2013) found that the Quail Habitat Restoration Initiative program (a cost-share incentive program restoring early successional habitat primarily through overstory tree removal in eastern Oklahoma) had no significant treatment effect on bobwhite occupancy. Winter survival was assessed and related to habitat improvements in Tennessee (Seckinger et al. 2008). Treatments of landscape vegetation included converting 33% of closed canopy forests into early succession resulting in 12% higher survival on treatment areas compared to control areas.

Previous research has stressed the importance of directly assessing habitat attributes and their influence on bobwhite survival to develop strategies to increase population densities (Taylor et al 1999, Seckinger et al. 2008, Holt et al. 2009). These studies, however, were not experimental in nature, but rather assessed existing habitat conditions. The need to assess these attributes at different spatial scales has been suggested (Brady et al. 1993, Roseberry 1993), especially in the context of fragmented landscapes. Seckinger et al. (2008) and Janke (2011) showed evidence of multi-scale habitat influence on survival thus underscoring the importance of considering scale when examining impacts of habitat on survival. Furthermore, studies relating multi-scale habitat metrics to survival typically have focused on a single (Taylor et al. 1999, Seckinger et al. 2008, Holt et al. 2009) rather than multiple seasons (Lohr et al. 2011). Because bobwhite survival has been shown to vary by season (Curtis et al. 1988, Burger et al. 1995), both summer and winter should be assessed when relating survival to multi-scale habitat attributes.

Understanding broader scale constraints on habitat may provide insight on how to best allocate resources for local-scale habitat improvement efforts.

Site-specific habitat management has been and will continue to remain the core strategy for bobwhite recovery (Dimmick et al. 2002, McKenzie 2009). However, it has been recognized that the success of a local management program is scale-dependent; that is, a given level of management intensity is more effective when conducted on a larger scale (Guthery 1997, Williams et al. 2004). One opportunity to manage large tracts of land for bobwhite and that lends itself to conducting habitat manipulations in an experimental framework is reclaimed surface mines. More than 600,000 ha have been reclaimed throughout the eastern United States under the Surface Mining Control and Reclamation Act of 1977 (SMCRA). However, the main focus of reclamation has been to prevent erosion and this has led to the establishment of undesirable species, such as sericea lespedeza (*Lespedeza cuneata*; Eddy 1999). Nonetheless, surface mines may substantially contribute to bobwhite conservation because of the scale of land that can be managed and they also could potentially serve as source populations for surrounding properties. However, efforts are needed to understand how best to manage reclaimed mined lands dominated by non-native species considered to provide poor habitat for bobwhite. Surface mines provide a good opportunity to better understand habitat relationships and to explore these relationships in an experimental setting because of the consistent nature of habitat and the scale at which it occurs thus making large-scale manipulations possible and without the limited inferences smaller or fragmented landscapes would impose. Furthermore, few studies have addressed the potential of reclaimed mined lands for bobwhite (Bekerle 2004, Stauffer 2011, Tanner 2012), and no research has been conducted in the context of habitat manipulation on these areas. In southwest Virginia, Stauffer (2011) conducted a pilot study for the potential of reclaimed mines as habitat for bobwhite quail. He found reclaimed mine lands supported poor to moderate quail numbers; however, reclaiming approaches were focused more on preventing runoff rather than providing bobwhite habitat.

Therefore, to assess the effects of habitat manipulation and to further our understanding of how vegetation on reclaimed mine lands affect bobwhite survival, we implemented an experiment on Peabody Wildlife Management Area (PWMA), USA from 2009 to 2013. Our primary objective was to determine if habitat management on treatment sites resulted in an increase in seasonal survival (SSR) versus largely unmanaged controls. Additionally, we sought to determine which vegetation attributes contribute to survival at three scales: home range, landscape, and microhabitat. Finally, we also sought to document overall survival and cause-specific mortality of bobwhite on a reclaimed surface mine. We hypothesized that our treatments would impact habitat in a manner leading to improved survival. Specifically, we predicted summer survival would improve the most because treatments were mainly focused on improving summer habitat conditions for bobwhite. We also expected to find multi-scale habitat effects on bobwhite survival, with those at the microhabitat scale being particularly influential because our management activities had the greatest impact at this scale.

STUDY AREA

The study was conducted on the Sinclair (1471 ha; 37°14'N, 87°15'W) and Ken (1853 ha; 37°17'N, 86°54'W) units of the Peabody Wildlife Management Area (PWMA) in Muhlenberg and Ohio counties, Kentucky, USA. Both units are reclaimed surface mine sites dominated by early-successional vegetation communities. Soils on both units consisted primarily of udorthents, which are characteristic of reclaimed mine sites. Areas on the WMA with trees were established pre-SMCRA (1977), whereas all early successional areas were established post-SMCRA. Sericea lespedeza, established during reclamation, constituted much of the vegetation on both units. From 2000-2004, native warm-season grasses (NWSG), including mixtures of big bluestem (*Andropogon gerardii*), little bluestem (*Schizachyrium scoparium*), indiangrass (*Sorghastrum nutans*), and switchgrass (*Panicum virgatum*), were established on both units. Soil compaction during reclamation led to the slow growth of American sycamore (*Platanus occidentalis*), eastern cottonwood (*Populus deltoides*), oaks (*Quercus* spp.), and maples (*Acer* spp.), which were the dominant tree species planted during reclamation throughout both units. Shrubs occurred across both areas and included patches of volunteer blackberry (*Rubus fruticosus*), smooth sumac (*Rhus glabra*), and coralberry (*Symphoricarpos orbiculatus*). Annual food plots were maintained on both units, and consisted primarily of a mixture of grain sorghum (*Sorghum bicolor*), corn (*Zea mays*), pearl millet (*Pennisetum glaucum*), browntop millet (*Urochloa ramose*), soybeans (*Glycine max*), and annual sunflower (*Helianthus annuus*).

METHODS

Study Design

We used an experimental approach to test our research hypothesis. Because no birds were ever documented to move between our two experimental units, we considered them separate sites. The two study sites were divided into approximately equal halves (Figure A.1), each with similar proportions of vegetation types that occurred on the property (Table A.1). One half of each study site was randomly assigned to receive intensive management treatments (disking, burning, and herbicide application) during 2009-2013, whereas the other half of the site was minimally disturbed and served as a control (two replicates of treatment and control) (Figure A.1). Efforts were focused on improving vegetation composition while trying to limit encroachment of invasive non-native plants that were previously established. The Sinclair control and treatment units were 673 and 798 ha, respectively, whereas the Ken control and treatments units were 1,043 and 810 ha, respectively (Table A.1). Treatments applied varied by year and season because of weather and logistical constraints. A total of 963 ha (Sinclair = 543 ha, Ken = 420 ha) were treated during the four years.

Hunting was allowed on both management units in the past, and up until 2008, small game hunting was governed by prevailing state-wide regulations established by Kentucky Department of Fish and Wildlife Resources (KDFWR). From 2009 to 2013, regulated quota hunts for bobwhite were established on both units to reduce hunting pressure and gather data on harvested birds (i.e., age, sex and crop contents).

Land Cover

Four major vegetation types (OH, SS, FOR, and NWSG), which constituted 91% of the total land cover on our study site (Table A.1), were delineated based on aerial imagery in ArcGIS 9.3 (ESRI, Redlands, CA, USA). To delineate between forest, shrubland, and open vegetation, 1-m resolution aerial imagery (2010) was used from the National Agriculture Inventory Program, US Department of Agriculture, Farm Service Agency. We selected representative woody cover on our study site as a template for reclassifying all 1x1-m cells as either "woody" or "open" with the Image Analyst tool in ArcGIS. We then used the Aggregate Tool to create unique polygons of "woody" or "open" vegetation with a minimum size of 0.2 ha, which was the average size of the smallest habitat management activity (disking) implemented on the site. To delineate between open vegetation, shrubland vegetation, and forest, we used percentage breaks within our individual raster cells based on the percent of woody vegetation present within each 0.2-ha polygon. We classified polygons with <10% woody cover as open vegetation, those with 11-55%

woody cover as shrubland, and those with >56% woody cover as forest. Forest vegetation had a mean basal area (stems >10 cm DBH) of 20.9 m^2/ha (SE = 1.77) and shrubland 9.6 m^2/ha (SE = 1.23); shrubland stems were typically 10–20 cm DBH. We classified NWSG by mapping areas comprised of ≥51% native grass using ArcPad 8.0 (ESRI, Redlands, CA, USA) on handheld Global Position System (GPS) units (Trimble Navigation Limited, Inc., Sunnyvale, CA), and classified areas that had <51% native grass as open herbaceous. All classifications were subjected to ground-truthing to validate GIS-based assignments of vegetation types.

Trapping and Radio Telemetry

We captured bobwhites during all months of the year using funnel traps (Stoddard 1931, Palmer et al. 2002) baited with cracked corn and grain sorghum placed throughout the study area at known covey locations as well as locations that appeared to contain suitable cover for bobwhite. Additionally, we used a second technique involving netting of radiomarked birds at night during the winter season to enhance the number of radiomarked bobwhite in a covey (Truitt and Daily 2000). We defined a biological year as 1 Oct-30 Sep and seasons as winter (1 Oct-31 Mar) and summer (1 Apr-30 Sep), based on Burger et al. (1995). Traps were covered with burlap and surrounding vegetation to reduce stress and predation on captured individuals. Traps were set ≥5 days per week and checked once daily in the evening. Upon meeting a minimum body mass of 90 g, each captured bobwhite was banded with aluminum bands (both legs) and, if body mass was >120 g, fitted with a necklace-style radio transmitter (crystal-controlled, two-stage design, pulsed by a CMOS multivibrator, American Wildlife Enterprise, Monticello, Florida, USA) that weighed ≤6.5 grams. Corteville (1998) reported that radiotransmitters may slightly reduce annual survival, but we assumed the magnitude of bias was consistent between treatment and control sites on our study area (Palmer and Wellendorf 2007, Terhune et al. 2007). We also recorded the sex, age, weight (g), and overall condition. Sex was determined by plumage and age was determined by the presence or absence of buff-tipped primary coverts. Condition was rated on a scale of 1 to 6 with 1 being excellent condition and 6 being extremely lethargic or injured. Birds were not collared if condition was >4. The project goal was to maintain 200 radiomarked birds for the entire study area, with a target of 50 collared birds per treatment unit. We also sought to maintain a sex ratio of radiomarked birds favoring females, if possible, to help adjust for typical male-biased (Roseberry and Klimstra 1984) populations and to increase the sample size of nests. Our trapping and handling methods complied with protocols of University of Tennessee Institutional Animal Care and Use Committee Permit 2042-0911.

Radiomarked birds were located ≥3 times/week using a scanning receiver and a handheld Yagi antenna (Advanced Telemetry Systems, Inc., Isanti, MN). We used the homing method (White and Garrot 1990) to locate bobwhites by walking to ≤50 m of the bird to minimize disturbance. We recorded the distance and azimuth to the bird by assessing the strength and direction of the telemetry signal. We then recorded the Universal Transverse Mercator (UTM) coordinates of the observer on a GPS unit (Garmin GPSMAP 60CSx, Garmin International, Inc. Olathe, KS, USA) and used the distance and azimuth to estimate the actual location of each bird. We recorded locations of birds at different times on subsequent days to capture the variability of diurnal patterns and defined bobwhites as being in the treatment, control, or off the study area according to the study area boundary. We also recorded the vegetation type in which the bird was located based on our four major vegetation types delineated. Upon detection of a mortality signal (12-hr signal), we immediately located the collar and determined the fate of the individuals as predation (mammal, avian), investigator-induced (consequence of research efforts), or unknown, based on evidence at the site of recovery and condition of the recovered transmitter (Curtis et al. 1988).

Microhabitat Vegetation Sampling

Collection of microhabitat vegetation based on bird-centered locations started during year 3 of our study and was added in an effort to gain insight into finer-scale habitat relationships than what was possible at the home-range scale. Therefore, the metrics described below represent two summers (2012 and 2013) and one winter (2012-2013). A subset of 40 birds (20 birds from each site) were selected at the beginning of each season of sampling; additional birds were added to compensate for mortalities throughout the season. We sampled vegetation at known locations for each bird in the subset each week throughout the season to minimize any temporal bias; a minimum of 4 sampling weeks was used to include birds in analysis.

For microhabitat vegetation, we selected, *a priori*, 12 metrics (Table A.5) that we thought might affect bobwhite survival, including litter depth (cm; LitterDepth), woody stem density (stems/ha; Midstory), the structure of vegetation representing overhead herbaceous cover (Nudds board cover for 1.25–1.5 m strata; Canopy3), ground sighting distance (cm; SightTube), distance to edge (m; DtoED), distance to woody cover (m; DtoWC), maximum herbaceous vegetation height (cm; MaxVegHeight), and 5 composition metrics (proportion; NWSG, Forbs, Ragweed, Brambles, and CoolSeason). Two metrics (MaxVegHeight and DtoWC) were collected exclusively during winter (Dec – Mar), seven metrics (SightTube, DtoED, NWSG, Forbs, Ragweed, Shrubs, CoolSeason) were collected exclusively during summer (May – Aug), and three metrics (Canopy3, LitterDepth, and Midstory) were collected during both seasons.

Microhabitat vegetation composition, SightTube, and LitterDepth metrics were collected from a 30-m transect centered on known bird locations. Species composition was collected at every meter along the 30-m transect generating 30 subsamples per transect. Proportion of vegetation cover was calculated as the number of points with a species present belonging in a given composition category divided by the total number of sampling points along the transect. Species composition categories were selected, *a priori*, based on biological importance. Species within the NWSG category included sideoats grama (*Bouteloua curtipendula*), big bluestem, broomsedge (*Andropogon* spp.), indiangrass, switchgrass, and little bluestem. Forb species included musk thistle (*Carduus nutans*), horseweed (*Conyza canadensis*), daisy fleabane (*Erigeron annuus*), Maximillian sunflower (*Helianthus maximiliani*), prickly lettuce (*Lactuca serriola*), sumpweed, sweet clover (*Melilotus officinalis*), Canada goldenrod (*Solidago canadensis*), old-field aster (*Symphyotrichum pilosum*), Carolina geranium (*Geranium carolinianum*), bird's foot trefoil (*Lotus corniculatus*), and yellow wood sorrel (*Oxalis stricta*). The single species category, Ragweed, included common ragweed (*Ambrosia artemisiifolia*) and was chosen because it was a dominant plant on disked areas and was representative of an uncommon cover type on our sites' annual plant communities. Brambles was chosen as a category because they are known to provide escape cover for bobwhite; species included pasture rose (*Rosa carolina*), multiflora rose (*Rosa multiflora*), prairie rose (*Rosa arkansana*), southern blackberry (*Rubus argutus*), southern dewberry (*Rubus trivialis*), black raspberry (*Rubus occidentalis*), coralberry, and Japanese honeysuckle. Cool-season grasses (CoolSeason) included tall fescue (*Festuca arundinacea*), Kentucky bluegrass (*Poa pratensis*), and orchardgrass (*Dactylis glomerata*). Cool-season grass species usually occur as dense stands of sod and provide structure not typically beneficial to bobwhite. Litter was defined as dead vegetative material on the soil surface (McCoy et al. 2001). Litter depth measurements were taken at the 0-, 6-, 12-, 18-, 24-, and 30-m mark on the transect by placing a ruler perpendicular to the ground and measured to the nearest 0.5 cm. The 6 measurements at each location were averaged to provide each bird with one litter depth value. Ground sighting distance (SightTube), which was considered an index of the openness at ground level (Gruchy 2007), was measured at the 0-, 10-, 20-, and 30-m mark on the transect by an observer kneeling down perpendicular to the transect and looking through a PVC tube 3.2 cm in diameter and 15.2 cm in length, mounted horizontally on a metal stake 20.3 cm above ground (Gruchy 2007). The distance (cm) at which vegetation obscured the visibility of a ruler viewed through the tube was recorded. The 4 sighting distances were averaged at each location to give each bird one sight-tube value. We estimated Midstory based on stems <11.4 cm dbh within a 5-m radius plot during summer and a 10-m radius plot during winter centered, in both cases, at each known bird location. Vegetation structure was assessed using a Nudds board (Nudds 1977), which was 2 m tall and consisted of 8, 0.25-m stratum. Nudds board (visual obstruction) readings were taken at 0, 10, 20, and 30 m along the transect during summer, where an observer determined the amount of vegetation covering each stratum (0 = no vegetation, 1 = 0-20%, 2 = 21-40%, 3 = 41-60%, 4 = 61-80%, and 5 = 81-100%) 5 m from the board from a kneeling position. During winter, the Nudds board was read from a distance of 10 m in each cardinal direction. Based on the 4 visual obstruction readings at each location, an average cover for each stratum was estimated for each sampled bird. We calculated Canopy3 as the average vegetative cover of stratum 3 (1.25-1.5 m). The average height of the tallest herbaceous vegetation (MaxVegHeight) was measured 10 m from plot center in each cardinal direction during winter. We estimated DtoED and DtoWC using a rangefinder. We considered edge to be where 2 different delineated vegetation types met and distance to woody cover to be the nearest woody cover offering acceptable escape from predators.

Home Range and Landscape Estimation

We used home range tools (HRT; Rodgers et al. 2007) extension in ArcGIS 9.3 (Environmental Systems Research Institute, Inc., Redlands, California, USA) and Geospatial Modelling Environment (GME; Spatial Ecology LLC, USA) to calculate 95% fixed-kernel home ranges (Worton 1989, Seaman et al. 1999) for each individual within each season. We only calculated home ranges for individual birds with $\geq$20 locations (DeVos and Mueller 1993, Taylor et al. 1999). During winter, home ranges were estimated for individuals rather than coveys because our subsequent analyses were focused on survival estimates derived for individual birds.

We used ArcGIS 9.3 to calculate home range and landscape-scale metrics. Home range covariates included proportion of each vegetation type and seasonal home range size for each bird (Table A.4). At the landscape scale, we calculated metrics associated with a buffer placed around each home range. Buffers were created using a radius equal to double the average daily movement observed during our study within each season (summer, 127 m; winter, 133 m). Average daily movement was calculated as the mean distance between consecutive daily locations for an individual, averaged across all individuals. Similar studies have used a buffer equal to the mean daily movement observed during the study (Holt et al. 2009). We doubled the average daily movement to help ensure we captured the vegetation where any bird could have theoretically traveled based on their actual locations. We selected, *a priori,* 7 patch-level metrics based on previous research that identified bobwhite habitat needs and population responses to habitat at different spatial scales: Forest/open vegetation (both NWSG and Open Herbaceous) edge density, shrubland/open vegetation edge density, core area of all four major vegetation types (using a 30-m edge effect), and a contagion index (Table A.4). The contagion index is a measure of patch-type interspersion and overall patch dispersion (O’Neill et al. 1988), and has been shown to influence bobwhite presence on an area (Roseberry and Sudkamp 1998). We calculated these metrics for each buffered home range using FRAGSTATS (McGarigal and Marks 1994) using a 150-m moving window. Based on values obtained from the moving window, we averaged metrics within each buffered home range.

Survival Analysis

We estimated SSR using the known fate model with a logit link function in Program MARK (White and Burnham 1999). We censored the first 7 days after a bird was released in our analysis to control for a potential short-term effect of capturing and radio-marking (Guthery and Lusk 2004). We used a staggered-entry method to analyze survival with the known fate model (Pollock et al. 1989). This method left-censors individual’s encounter histories until they are captured and enter the monitored population. We right-censored individuals because of emigration from the study area, radio failure or loss, or unknown fate. We assumed birds were randomly sampled, survival times for individuals were independent, and censoring mechanisms were independent of animal fate. Within each of our sites, some birds moved between treatment and control units throughout the season. Thus we determined whether a bird was a treatment or control bird by the number of locations (>70%) on each unit. Each survival period (summer and winter) consisted of 183 days. Encounter histories were coded as weekly survival periods, whereby each survival period had 26 encounter history periods. If individual birds survived from one season to the next, we considered them to be new individuals at the start of the next season.

Our survival analysis consisted of four different suites of covariates that were sequentially combined into a single analysis to assess the value of all covariates on survival (Doherty et al. 2012). These four suites of covariates represented group metrics (Table A.3) and vegetation metrics measured at three different spatial scales: home range, landscape (Table A.4), and microhabitat (Table A.5). In order to minimize inclusion of extraneous models and keep model sets to a reasonable size, we used a 3-stage modeling process whereby we found the best biological, experimental, and habitat models (in that order) from our four suites of covariates. Further, we assessed additional terms in the final model set to explore key relationships associated with our experimental framework and that may have been masked earlier in our modeling process (Doherty et al. 2012). Vegetation metrics were only calculated for birds with estimated home ranges (i.e., >20 locations) to establish a reliable, explicit spatial context for these individuals that could then be geo-referenced to our vegetation layers. For birds without estimated home ranges or associated microhabitat data, we used null (average) values for missing covariates (Cooch and White 2008). Using this approach, we were able to include all birds in our analysis resulting in less bias by having to exclude short-lived birds for which home range estimates were unavailable.

For survival analysis, the selection of covariates were based on published studies and our assessment of relative biological importance. The first stage of analysis (biological) included group metrics: sex, age, weight, condition index, season, linear time, and weekly time effects (Table A.3). Linear time was variation from week 1 to week 26 during a season and weekly time was variation among weeks during a season. The top model from the first stage was then used as the baseline model in the second stage of analysis (experiment). Experiment covariates (Site, Year, and T/C; Table A.3) were then combined with the top biological model as additive and interaction models. We also developed additive and interaction models among treatment covariates exclusively as well as in the final model set to test our hypothesis concerning treatment effects on survival. For the third stage of analysis, we added habitat covariates (home range, landscape, and microhabitat; Table A.4, A.5). All covariates from the third stage of analysis were incorporated as additive models containing the top biological and experimental model to incorporate effects among multiple habitat covariates. After analyzing all three model stages and assessing other combinations and interactions, we arrived at the top model based on the ΔAIC_c score and used this model for estimating SSR. We used a ΔAIC_c value of <2 (Burnham and Anderson 2002) to determine validity of a model for explaining variance in survival. We also computed model-averaged parameter estimates for SSR using the survival estimates from each model.

RESULTS

We captured and double banded 1,794 bobwhites (866 males, 652 females, and 276 birds for which we were not able to confirm gender) from 1 October 2009 – 30 September 2013 (Table A.2). We captured more juveniles (*n* = 1,443) than adults (*n* = 351) during the course of our study. Of the 1,794 captured birds, we radiomarked 1,198 (Table A.2), but were only able to use 1,131 in our survival analyses because of censoring. We obtained $\geq$20 locations on 635 birds for which we were able to estimate home ranges and associated habitat metrics. Our trap success between sites was similar (t = 2.66, *P* = 0.11) between winter (Ken = 2.8%, Sinclair = 2.7%) and summer (Ken = 2.0%, Sinclair = 2.4%; Table A.2). Unknown causes (n = 363, 37.1%) accounted for the majority of mortality events; however, mammalian predation (n = 303, 31.0%) was the most common cause of known mortality followed by avian predation (n = 187, 19.1%; Table A.7). Mammalian predation also accounted for the majority of known predation during both summer (n = 121, 12.4%) and winter (n = 182, 18.6%).

The best biological model from our first stage of analysis was {Season} and was used as the baseline model in our second (experiment models) stage of analysis (Table A.8). After incorporating models that included experiment covariates with additive and interaction terms, {Season × Year} became the top model and was used as the baseline for the third stage of analysis (habitat models). The model {Season × Year + LitterDepth + OH_CA + HRS} became the top model once we incorporated additive models with habitat covariates. After running other combinations and interactions with this model, {Season × Year + LitterDepth + OH_CA + HRS + T/C × Season} became the final overall model based on ΔAIC_c and AIC_c weight (Table A.8). Based on 95% confidence intervals of beta estimates, all variables within the top model differed from zero except OH_CA and HRS (Table A.9). The model receiving the second most support did not include the {T/C × Season} interaction and was 15.47 times less likely than the top model ($\Delta AIC_c = 11.466$, AIC weight = 0.003; Table A.8), suggesting the effect of this interaction was considerable.

The interaction {Season×Year} term showed a significant negative relationship among seasons and years (Figure A.1), signifying that the relationship of how survival varied from winter to summer differed among years. Our treatment effect (T/C) was included in the top model {Season × Year + LitterDepth + OH_CA + HRS + T/C × Season} and had a positive effect ($\beta = 0.256$, CI = 0.057, 0.456; Table A.9) on survival, suggesting SSR was influenced by our habitat treatments. Although survival declined from winter to summer for both control and treatment, the interaction {T/C × Season} term indicated survival was higher for treatment than control during the summer (Figure A.3). Among habitat covariates, LitterDepth ($\beta = -0.387$, CI = -0.5809, -0.1930) was the most influential covariate, having a negative relationship with survival (Table A.9). Both OH_CA ($\beta = -0.039$, 95% CI = -0.0823–0.0037) and HRS ($\beta = 0.001$, 95% CI = -0.0001–0.0021) were included in the top model, however, their beta estimates did not differ from 0 (Table A.9), suggesting they had a weak influence on SSR. Other habitat covariates, as well as single variable, additive, and interaction models, had no support ($\Delta AIC_c > 4$).

Based on the top model {Season × Year + LitterDepth + OH_CA + HRS + T/C × Season}, summer survival was 0.148 ± 0.015 and winter survival was 0.281 ± 0.022 pooled across years (Table A.6). Pooled SSR was 0.206 ± 0.024 and 0.232 ± 0.027 for treatment and control, respectively. Model-averaged SSR were 0.148 (SE = 0.015) during summer and 0.282 (SE = 0.022) during winter. Winter survival was quite variable (2010 = 0.217, 2011 = 0.128, 2012 = 0.230, 2013 = 0.484), whereas summer survival was rather consistent (2010 = 0.168, 2011 = 0.150, 2012 = 0.177, 2013 = 0.117) among years (Table A.6). Pooled survival rates during the summer were higher on treatment (0.179) than control (0.109); however, survival was higher on control (0.355) than treatment (0.233) during the winter (Table A.6; Figure A.3).

Additionally, we evaluated a number of models seeking to identify relationships that could explain the differences in survival associated with the {T/C × Season} interaction. First we assessed models for winter data only using the top model {Season × Year + LitterDepth + OH_CA + HRS + T/C × Season} plus 6 habitat covariates (SS, SS_CA, Brambles, FOR, FOR_CA, and Midstory). Of these covariates, SS (β = 0.010, CI = 0.001–0.018) had the strongest impact having a positive relationship with survival. Similarly, we assessed a set of models for summer only using 6 covariates (LitterDepth, Canopy3, Brambles, Forbs, Ragweed, and OH_CA). Among these, LitterDepth (β = -0.514, 95% CI = -0.713, -0.316) and Canopy3 (β = -0.935, 95% CI = -1.433–0.436) had the strongest (negative) impact on survival.

DISCUSSION

In our experimental approach, we altered the landscape structure on a reclaimed surface mine by applying treatments (prescribed fire, disking, and herbicide application) to approximately 60% of treatment areas (78% of non-forested portions of treatment areas).

Management efforts were focused on improving vegetation composition for bobwhite and reducing dominance of invasive species, such as sericea lespedeza. Our results suggest that applied treatments (T/C) had an effect on increasing survival. Specifically, higher survival was associated with treatment during the summer. We predicted our treatments would have the greatest impact on summer survival because they mostly focused on improving summer habitat conditions for bobwhite. However, the interaction among our treatment effect (T/C) and seasons showed that survival was lower on treatment during the winter compared to control. This finding suggests that our management activities may have negatively affected winter habitat conditions for bobwhite. Increasing the availability of woody cover has been suggested as a means for increasing winter survival (Yoho and Dimmick 1972, Roseberry and Klimstra 1984, Williams et al. 2000). Janke (2011) observed higher winter survival related to increases in availability of woody cover in Ohio. Flock (2006) suggested that lower bobwhite survival on CRP fields was the result of a lack of woody cover during winter in southeastern Kansas. Likewise, Tanner (2012) found a positive relationship between survival and the amount of forest vegetation within bobwhite home ranges. Our post-hoc models help further explain the {T/C × Season} interaction and demonstrate that winter survival increased as the amount of shrub vegetation increased within bobwhite home ranges, suggesting shrub cover may have been limiting on treatment areas. Through burning, we may have reduced the quality of shrub cover on treatment areas. Therefore, control areas may have served as a refuge for bobwhite during the winter due to low disturbance and more available thermal/woody escape cover. Regarding our post-hoc summer models, both litter depth and canopy cover at 1.25-1.5 m had a negative influence on survival, suggesting an increase in either of these metrics reduced survival. This finding directly relates to our management activities focused on reducing ground litter and removing sericea lespedeza.

The seasonal timing and duration of treatments may have profound differences in their effects on the vegetative community. Given the progression of treatments applied, we suspected that SSR would increase from year 1 (0.193) to year 4 (0.301). The top model from our survival analysis included a significant interaction among seasons and years, which was mainly a result of the variation in winter survival over the 4 years of our experiment (0.217, 0.128, 0.230, and 0.484). In contrast, summer survival rates were relatively constant (0.168, 0.150, 0.177, and 0.117) during this same period. We expected annual variation from year to year as well as from summer to winter as a result of changes in weather patterns and predator abundance (Palmer and Wellendorf 2007). Similarly, Holt et al. (2009) reported large annual variation in winter survival rates in Mississippi, 0.060 and 0.465 during the two years of their study.

Survival was greater during winter (0.281) than summer (0.148), which is in contrast to most reported studies on bobwhite. In northern Missouri, Burger et al. (1995) estimated winter survival at 0.159 and summer survival at 0.332, whereas Lohr et al. (2011) reported winter survival rates of 0.183 and summer survival rates of 0.343. Sisson et al. (2009) reported summer survival estimates averaged 0.352 during a 13-year study in southern Georgia and eastern Alabama, and Curtis et al. (1988) reported winter survival rates of 0.185 in North Carolina. However, in concurrence with our estimates, Williams et al. (2012) reported greater winter survival (0.308) than summer survival (0.267) in New Jersey.

Although past studies have documented reproductive (Burger et al. 1994), incubation (Klimstra and Roseberry 1975, Roseberry and Klimstra 1984), and brood rearing (DeVos and Mueller 1993, Burger et al. 1995) costs during summer, we do not believe these factors would be necessarily disproportionate on our site. In our case, differing seasonal habitat use by bobwhite may be an important explanation. Bobwhite frequented forested areas (Tanner 2012) as opposed to more open herbaceous areas associated with nesting activities during the summer months. This contrasts the findings by Seckinger et al. (2008) who reported an increase in winter survival after removal of closed-canopy forest vegetation. However, forest vegetation on PWMA was established during reclamation and was uncharacteristic of typical forested areas within the Mid-South. Forest vegetation (22% of study area) rarely had canopy closure and as a result, was more similar to open-canopy woodlands. The understory was comprised primarily of blackberry and honeysuckle, which provided good escape cover. In Virginia, Tonkovich and Stauffer (1993) reported quail tended to use sites with more Japanese honeysuckle (*Lonicera japonica*) than was generally available. Likewise, Roseberry and Klimstra (1984) noted Japanese honeysuckle was frequently an understory component of woody headquarters of coveys in Illinois. Yoho and Dimmick (1972) noted the consistency with which covey activity centers in Tennessee were characterized by honeysuckle, reporting that 63 of 107 roosts were located in honeysuckle.

Our estimates of summer survival were exceptionally low compared to past research (Burger et al. 1995, Sisson et al. 2009, Lohr et al. 2011). This may have been because our study was conducted on a reclaimed surface mine composed primarily of sericea lespedeza. Sericea lespedeza seeds were commonly found in crops of bobwhite on our study area, but are virtually indigestible and provide no nutritional benefit (Davison 1958). In addition, sericea lespedeza litter accumulation has been associated with reduced forb establishment and species richness (Foster and Gross 1998) and may actively interfere with germination of other species through allelopathic compounds generated in leaf and stem tissue (Adams et al. 1973, Wade 1989). Bugg and Dutcher (1989) reported invertebrate use of sericea lespedeza was lowest of all plant species studied. Eddy (1999) concluded invertebrate and vertebrate species declined by 73 and 55% respectively in sericea lespedeza sites. High stem density, lack of singing perches, fewer canopy openings, and a decrease in the seasonal availability of foliage, flowers, seeds, and prey insects were given as factors in the decline. During the nesting and chick-rearing season, hens require a large proportion of invertebrate protein in their diet (Wiens and Rotenberry 1979). Likewise, bobwhite chicks require a high protein diet during the first two weeks of life, with invertebrate species comprising 94.1% of all foods eaten (Eubanks and Dimmick 1974). With this is mind, more beneficial plant species may have been lacking on our site; thus, reducing invertebrate density and seed production. In regards to low survival, these factors may have contributed to lower adult body condition as well as sub-optimal foraging conditions for newly hatched chicks during the summer.

We did not detect a site effect; however, we anticipated survival would be consistent between sites because treatments were generally being applied in equal proportions at both sites. Furthermore, variation among sites was not especially great as both were surface mines reclaimed using similar techniques at a similar time. Because we considered our two sites blocks, failing to detect differences supported our approach and allowed greater sensitivity in evaluating the experimental treatments themselves.

Among covariates tested, we hypothesized those at the microhabitat scale would be most influenced by our treatments. Our two major management activities (disking and burning) were aimed at reducing sericea lespedeza and litter cover while increasing annual, seed-bearing forbs. Litter depth had a negative relationship with survival and was contained in the top model, suggesting disking and burning may have been beneficial. The importance of bare ground for foraging, brooding, and roosting has been both qualitatively and quantitatively described (Stoddard 1931, Klimstra and Zicarrdi 1963, Ellis et al. 1969, Brown and Samuel 1978). Although non-significant, the top model also included an additive effect for the amount of open herbaceous core area (negative) and home range size (positive), suggesting poor proximity of escape cover in large blocks of open vegetation was a problem.

Although the beta confidence intervals for all other habitat covariates contained zero and were included in models with a $\Delta AIC_c > 4$, we expected some of these would impact survival. For instance, we suspected that an increase in ragweed would positively affect survival. Ragweed, a valued annual plant for bobwhite providing food and the proper structure for overhead cover, was promoted by disking, an intended outcome of the practice. Additionally, we hypothesized that metrics such as contagion index, forest and scrub shrub edge density, and distance to edge and woody cover would have an impact on survival. All of these metrics are related to the interspersion of vegetation types, which is important for bobwhite survival (Williams et al. 2000, Holt et al. 2009, Janke and Gates 2013).

Bobwhite population dynamics are driven by many ecological factors operating at multiple spatial scales and different amounts and types of treatments are likely to have varying effects on bobwhite survival. The fact that our study was conducted on a reclaimed surface mine may have weakened results relating to survival. Soil is often of poor quality and heavily compacted, which allows sericea lespedeza to be have a competitive advantage. When KDFWR acquired PWMA in 1995, sericea lespedeza had already become the dominant vegetative cover (Fitzgerald et al. 2005). Native seedbanks are severely impacted during surface mining activities. Once established, sericea lespedeza usually becomes dominant and its seed can remain viable in the seedbank for 30 years (Fitzgerald et al. 2005). Seeding native forbs and grasses could help alleviate the problem and yield much more desirable results (Washburn et al. 2000). Although sericea has limited forage value for wildlife, it can provide some acceptable cover for bobwhite.

A shorter return interval of applied treatments may have been needed to sufficiently hinder growth of non-native species, such as sericea lespedeza. Greenfield et al. (2002) reported that fall disking improved quality of tall fescue and orchardgrass CRP fields for bobwhite in Missouri. However, the relative effectiveness of disking differed between tall fescue and orchardgrass fields, as improvements in tall fescue fields were minimal and short-lived (i.e. confined to one growing season). Greenfield et al. (2003) also noted similar trends in tall fescue fields in Mississippi where the relative effectiveness of disking and burning diminished greatly following the first growing season. Regrowth of sericea lespedeza after initial disking or prescribed burning may have occurred at a fast rate, diminishing the positive effects of applied treatments. Our data supports this hypothesis, as the percent cover of sericea lespedeza on disk blocks after 3 years ($\bar{x}$ = 70.7) was very similar to untreated areas ($\bar{x}$ = 75.8) on our site (University of Tennessee, unpublished data). These results suggest management (disking and herbicide application) may need to be implemented on a shorter rotation (e.g., 2 years). Furthermore, the range of conditions we examined were not extreme. We did not have high-quality cover as occurs on managed lands within the Red Hills region of Florida and Georgia, nor did we have extensive low-quality habitat exemplified by mature hardwood forest and fescue pastures as is typical of large portions of the CHBCR.

Management of reclaimed surface mines may substantially contribute to bobwhite conservation because of the scale of land that can be managed. However, a lack of plants that can provide appropriate cover and/or nutrition on severely disturbed, reclaimed habitats may offer new challenges to managers. Seasonal food availability may impact survival on reclaimed habitats which are dominated by perennial plant species (Jones et al. 1994). We found evidence that our treatments had an effect on improving summer survival. However, it appears that our management activities may have negatively affected winter survival, which may have been attributable to the seasonal requirement of woody cover during winter. Additionally, our estimates of summer survival were exceptionally low compared to past research (Parent et al. 2012), possibly as a result of nutritional deficiency during this time period. Regardless of the mechanism, extensive stands of perennial herbaceous vegetation, overwhelmed by sericea lespedeza, led to exceptionally low summer survival.

MANAGEMENT IMPLICATIONS

Management on existing reclaimed surface mines should focus on reducing extensive stands of non-native vegetation through methods that promote bare soil (i.e., disking) and encourage increased plant diversity at a finer scale. Management activities such as prescribed burning that may reduce extent and density of woody cover must be used judiciously to ensure adequate winter cover is maintained. Establishment of new woody cover through shrub plantings, taking care to increase interspersion of this cover within extensive stands of open herbaceous cover should also be pursued. A shorter return interval (e.g., 2 years) for disturbances may be the most effective way to promote the persistence of native plant species and reduce the cover of non-native species, especially those that are aggressive such as sericea lespedeza. On new reclamation sites, use of an appropriate mix of native herbaceous vegetation (i.e, forbs, grasses, and legumes) that is well-interspersed with woody cover should lead to far more successful bobwhite management in the long run. However, given the importance of loss and fragmentation of habitat, bobwhite management aimed at increasing survival may only be effective on large landscapes where population persistence is likely. Additional experiments across a range of habitat conditions over a longer period of time may be required to determine management intensity and duration thresholds required to elicit greater changes in survival for northern bobwhite populations.

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APPENDIX A

Table A.1. Delineated vegetation types and total cover (ha) on Peabody WMA, Ohio and Muhlenberg Counties, Kentucky, USA, 1 Sep 2009-30 Sep 2013.

Site	Unit	Vegetation	Abbreviation	Hectares	Total by unit (ha)	Total (ha)
Ken	Treatment	Annual grain	AG	21.0	810.2	1853.1
		Forest deciduous	FOR	216.6		
		Native warm-season grass	NWSG	93.7		
		Open herbaceous	OH	244.7		
		Scrub shrub	SS	175.4		
		Water	Water	52.6		
		Wetland emergent	WE	6.2		
	Control	Annual grain	AG	4.8	1042.8	
		Forest deciduous	FOR	189.0		
		Native warm-season grass	NWSG	112.0		
		Open herbaceous	OH	279.7		
		Scrub shrub	SS	344.0		
		Water	Water	112.7		
		Wetland emergent	WE	0.6		
Sinclair	Treatment	Forest deciduous	FOR	162.0	798.2	1470.6
		Native warm-season grass	NWSG	24.1		
		Open herbaceous	OH	367.8		
		Scrub shrub	SS	194.9		
		Water	Water	44.1		
		Wetland emergent	WE	5.3		
	Control	Annual grain	AG	6.5	672.5	
		Forest deciduous	FOR	165.3		
		Native warm-season grass	NWSG	33.8		
		Open herbaceous	OH	303.8		
		Scrub shrub	SS	126.8		
		Water	Water	25.5		
		Wetland emergent	WE	10.7		

Table A.2. Trapping success by site and season and age and sex of captured and radiomarked northern bobwhite on Peabody WMA, Ohio and Muhlenberg Counties, Kentucky, USA, 1 Sep 2009-30 Sep 2013.

Captured								
			Sex			Age		
Site	Season	Trapping success (%)	Males	Females	Unknown	Adult	Juvenile	Total
Sinclair	Winter	2.7	187	155	1	40	303	343
	Summer	2.4	241	138	128	91	416	507
Ken	Winter	2.8	194	190	1	102	283	385
	Summer	2.0	244	169	146	118	441	559
	Total		866	652	276	351	1443	1794
Radiomarked								
Sinclair	Winter		135	117	0	34	218	252
	Summer		184	119	60	85	278	363
Ken	Winter		161	160	0	89	232	321
	Summer		123	97	42	42	220	262
	Total		603	493	102	250	948	1198

Table A.3. List and description of group and treatment metrics used to assess effects on survival of radiomarked northern bobwhite on Peabody WMA, Ohio and Muhlenberg Counties, Kentucky, USA, 1 Sep 2009-30 Sep 2013.

Metric	Description
T[a]	Linear time
W[b]	Weekly time
Year	Year, 2009-2013
Sex	Sex, either male or female
Age	Age, either juvenile or adult
Cond[c]	Condition index
Weight	Weight of bird (g)
Site	Site, either Ken or Sinclair
Season	Season, either summer or winter
T/C	Treatment or control

[a] Variation from week 1 to week 26 during a season.

[b] Variation among weeks during a season.

[c] Measure of body condition on a scale of 1 to 6 with 1 being excellent condition and 6 being extremely lethargic or injured; birds were not collared if Cond >4.

Table A.4. List and description of home range and landscape metrics used to assess effects on survival of radiomarked northern bobwhite on Peabody WMA, Ohio and Muhlenberg Counties, Kentucky, USA, 1 Sep 2009-30 Sep 2013.

Metric	Description	Scale
FOR	FOR vegetation within a home range (%)	Home range
SS	SS vegetation within a home range (%)	Home range
NWSG	NWSG vegetation within a home range (%)	Home range
OH	OH vegetation within a home range (%)	Home range
HRS	Home range size (ha)	Home range
ED_FOR	FOR to open vegetation edge density (m/ha)	Landscape
ED_SS	SS to open vegetation edge density (m/ha)	Landscape
FOR_CA	Core area of FOR vegetation (ha)	Landscape
SS_CA	Core area of SS vegetation (ha)	Landscape
NWSG_CA	Core area of NWSG vegetation (ha)	Landscape
OH_CA	Core area of OH vegetation (ha)	Landscape
CI	Contagion index (%)	Landscape

Table A.5. List and description of microhabitat metrics used to assess the effects on survival of radiomarked northern bobwhite on Peabody WMA, Ohio and Muhlenberg Counties, Kentucky, USA, 1 Sep 2009-30 Sep 2013.

Metric	Description
Canopy3[a]	Nudds board coverage 1.25-1.5 (m)
LitterDepth	Litter depth (cm)
Midstory[b]	Stem density (stems/ha)
SightTube[c]	Sight tube (cm)
DtoED	Distance to edge (m)
DtoWC	Distance to woody cover (m)
MaxVegHeight	Maximum herbaceous vegetation height (cm)
NWSG[d]	NWSG species
Forbs[d]	Forb species
Ragweed[d]	Ragweed
Brambles[d]	Bramble species
CoolSeason[d]	Cool-season grasses

[a] Average vegetation cover on Nudds board (1.25-1.5 m) centered on bird locations identified through radio-telemetry.

[b] Average stem density of woody species <11.4 cm dbh within 5-m radius plot (summer) and 10-m radius plot (winter) centered on bird locations identified through radio-telemetry.

[c] Average ground-sighting distance 20.3 cm above ground level at 0, 10, 20, and 30 m along 30-m transect on bird-centered locations identified through radio-telemetry.

[d] Average proportion of cover based on 30-m transect centered on bird locations identified through radio-telemetry.

Table A.6. Winter (1 Oct-31Mar) and summer (1 Apr-30 Sep) seasonal survival (*S*) estimates from Program MARK of radiomarked northern bobwhite by treatment and control on Peabody WMA, Ohio and Muhlenberg Counties, Kentucky, USA, 1 Sep 2009-30 Sep 2013. Estimates were derived from top model in survival analysis.

			Pooled			Treatment			Control		
Year	Season	Site	*n*	S	SE	*n*	S	SE	*n*	S	SE
2010	Winter	Sinclair	43	0.203	0.043	15	0.160	0.041	28	0.271	0.053
		Ken	108	0.222	0.039	57	0.177	0.039	51	0.291	0.048
		Pooled	151	0.217	0.038	72	0.172	0.038	79	0.286	0.047
	Summer	Sinclair	57	0.158	0.041	48	0.190	0.045	9	0.117	0.039
		Ken	70	0.176	0.041	42	0.209	0.046	28	0.132	0.040
		Pooled	127	0.168	0.039	90	0.201	0.043	37	0.126	0.038
2011	Winter	Sinclair	110	0.121	0.024	82	0.089	0.021	28	0.177	0.037
		Ken	119	0.137	0.027	69	0.101	0.025	50	0.195	0.039
		Pooled	229	0.128	0.022	151	0.094	0.021	78	0.186	0.035
	Summer	Sinclair	40	0.140	0.041	19	0.171	0.046	21	0.102	0.037
		Ken	68	0.156	0.041	47	0.188	0.046	21	0.116	0.038
		Pooled	108	0.150	0.039	66	0.181	0.044	42	0.110	0.036
2012	Winter	Sinclair	72	0.223	0.050	33	0.178	0.048	39	0.292	0.058
		Ken	30	0.243	0.054	20	0.196	0.053	10	0.313	0.061
		Pooled	102	0.230	0.049	53	0.184	0.048	49	0.300	0.057
	Summer	Sinclair	145	0.168	0.029	87	0.201	0.035	58	0.126	0.029
		Ken	129	0.186	0.031	64	0.220	0.037	65	0.142	0.030
		Pooled	274	0.177	0.027	151	0.211	0.033	123	0.134	0.027
2013	Winter	Sinclair	94	0.475	0.045	73	0.424	0.047	21	0.543	0.051
		Ken	120	0.495	0.045	79	0.446	0.048	41	0.562	0.050
		Pooled	214	0.484	0.042	152	0.434	0.044	62	0.553	0.048
	Summer	Sinclair	124	0.108	0.025	55	0.136	0.031	69	0.076	0.023
		Ken	150	0.123	0.025	80	0.152	0.031	70	0.087	0.023
		Pooled	274	0.117	0.022	135	0.145	0.029	139	0.083	0.021
Pooled 4 years	Winter	Sinclair	319	0.272	0.027	203	0.224	0.028	116	0.344	0.039
		Ken	377	0.293	0.028	225	0.243	0.031	152	0.365	0.039
		Pooled	696	0.282	0.022	428	0.233	0.025	268	0.355	0.035
	Summer	Sinclair	366	0.139	0.020	209	0.170	0.026	157	0.101	0.021
		Ken	417	0.155	0.019	233	0.187	0.025	184	0.115	0.021
		Pooled	783	0.148	0.015	442	0.179	0.022	341	0.109	0.019

Table A.7. Winter (1 Oct-31Mar) and summer (1 Apr-30 Sept) causes of mortality for 978 radiomarked northern bobwhite by site on Peabody WMA, Ohio and Muhlenberg Counties, Kentucky, USA, 1 Sep 2009-30 Sep 2013.

			Causes of mortality						
Site	Season		Avian	Mammal	Harvest	Investigation	Other	Unknown	Total
Sinclair	Winter		68	95	0	13	6	37	219
	Summer		42	59	0	28	17	100	246
Ken	Winter		35	87	1	8	2	108	241
	Summer		42	62	0	33	17	118	272

Table A.8. Highest ranking models (above Null model) from 59 models based on ΔAIC_c values and AIC_c weights used to assess the influence of group, home range, landscape, and microhabitat metrics on northern bobwhite survival on Peabody WMA, Ohio and Muhlenberg Counties, Kentucky, USA, 1 Sep 2009-30 Sep 2013[a].

Model	AICc	ΔAICc	AICc weights	Model likelihood	*k*	Deviance
Season × Year + LitterDepth + OH_CA + HRS + T/C × Season	5533.74	0.0000	0.98781	1	13	5507.71
Season × Year + LitterDepth + OH_CA + HRS	5545.21	11.4662	0.00320	0.0032	11	5523.18
Season × Year + LitterDepth + OH_CA + HRS + T/C × Year	5546.33	12.5865	0.00183	0.0019	15	5516.29
Season × Year + LitterDepth + OH_CA	5546.54	12.7950	0.00165	0.0017	10	5526.52
Season × Year + LitterDepth + HRS	5547.06	13.3226	0.00126	0.0013	10	5527.04
Season × Year + LitterDepth + OH_CA + HRS + T/C	5547.17	13.4349	0.00119	0.0012	12	5523.15
Season × Year + LitterDepth + OH_CA + Canopy3 + Brambles + DtoED	5547.22	13.4815	0.00117	0.0012	13	5521.19
Season × Year + LitterDepth + OH_CA + T/C	5548.52	14.7807	0.00061	0.0006	11	5526.50
Season × Year + LitterDepth + Brambles	5548.59	14.8465	0.00059	0.0006	10	5528.57
Season × Year + LitterDepth	5550.17	16.4335	0.00027	0.0003	9	5532.16
Season × Year + LitterDepth + Canopy3	5551.30	17.5575	0.00015	0.0002	10	5531.28
Season × Year + LitterDepth + DtoED	5551.51	17.7672	0.00014	0.0001	10	5531.49
Season × Year + LitterDepth + T/C	5552.18	18.4362	0.00010	0.0001	10	5532.16
Season × Year + HRS	5556.91	23.1661	0.00001	0	9	5538.89
Season × Year + OH_CA	5557.44	23.6990	0.00001	0	9	5539.42
Season × Year + Canopy3	5559.29	25.5508	0.00000	0	9	5541.28
Season × Year + Brambles	5559.69	25.9532	0.00000	0	9	5541.68
Season × Year + DtoED	5559.93	26.1911	0.00000	0	9	5541.92
Season × Year	5559.97	26.2329	0.00000	0	8	5543.96
Season × Year + SightTube	5560.36	26.6247	0.00000	0	9	5542.35

Continued

Table A.8. Continued

Model	AICc	ΔAICc	AICc weights	Model likelihood	*k*	Deviance
Season × Year + SS_CA	5560.74	27.0007	0.00000	0	9	5542.73
Season × Year + NWSG	5560.74	27.0007	0.00000	0	9	5542.73
Season × Year + NWSG_CA	5560.98	27.2387	0.00000	0	9	5542.96
Season × Year + FOR_CA	5561.17	27.4340	0.00000	0	9	5543.16
Season × Year + SS	5561.28	27.5381	0.00000	0	9	5543.26
Season × Year + CoolSeason	5561.39	27.6544	0.00000	0	9	5543.38
Season × Year + NWSG	5561.54	27.7950	0.00000	0	9	5543.52
Season × Year + Ragweed	5561.55	27.8063	0.00000	0	9	5543.53
Season × Year + CI	5561.57	27.8321	0.00000	0	9	5543.56
Season × Year + DtoWC	5561.58	27.8362	0.00000	0	9	5543.56
Season × Year + FOR	5561.79	28.0525	0.00000	0	9	5543.78
Season × Year + Forbs	5561.80	28.0648	0.00000	0	9	5543.79
Season × Year + Midstory	5561.88	28.1448	0.00000	0	9	5543.87
Season × Year + MaxVegHeight	5561.91	28.1706	0.00000	0	9	5543.90
Season × Year + T/C	5561.95	28.2103	0.00000	0	9	5543.94
Season × Year + ED_SS	5561.95	28.2106	0.00000	0	9	5543.94
Season × Year + OH	5561.96	28.2214	0.00000	0	9	5543.95
Season × Year + ED_FOR	5561.98	28.2354	0.00000	0	9	5543.96
Season × T/C + Year	5566.95	33.2076	0.00000	0	7	5552.94
Season + Year	5577.03	43.2905	0.00000	0	5	5567.03

Continued

Table A.8. Continued

Model	AICc	ΔAICc	AICc weights	Model likelihood	*k*	Deviance
Season × T/C	5587.51	53.7709	0.00000	0	4	5579.51
Season + W	5588.38	54.6361	0.00000	0	27	5534.25
Year × T/C	5590.28	56.5375	0.00000	0	8	5574.27
Year × Site	5590.35	56.6072	0.00000	0	8	5574.34
Year	5593.30	59.5623	0.00000	0	4	5585.30
Year + Site + T/C	5594.87	61.1264	0.00000	0	6	5582.86
Year + T/C	5595.17	61.4348	0.00000	0	5	5585.17
Season	5595.42	61.6843	0.00000	0	2	5591.42
Season + T/C	5596.92	63.1848	0.00000	0	3	5590.92
Weight	5601.09	67.3455	0.00000	0	2	5597.08
W	5605.33	71.5932	0.00000	0	26	5553.22
Sex	5607.97	74.2268	0.00000	0	2	5603.97
Site	5608.22	74.4813	0.00000	0	2	5604.22
Null	5608.63	74.8945	0.00000	0	1	5606.63

[a] Notation generally follows that of Lebreton et al. (1992).

Table A.9. List of group, landscape, and micro-habitat metrics with associated beta values contained in models within a ΔAIC_c <2 for survival of radiomarked northern bobwhite on Peabody WMA, Ohio and Muhlenberg Counties, Kentucky, USA, 1 Sep 2009-30 Sep 2013.

Metric	Beta value	LCI	UCI
Season	1.4416	1.0822	1.8010
Year 1	0.1862	-0.1265	0.4990
Year 2	0.1249	-0.1943	0.4440
Year 3	0.2173	-0.0337	0.4683
Year 1 × Season	-0.9342	-1.3902	-0.4782
Year 2 × Season	-1.1710	-1.6032	-0.7388
Year 3 × Season	-0.9264	-1.3831	-0.4697
LitterDepth	-0.3870	-0.5809	-0.1930
OH_CA	-0.0393	-0.0823	0.0037
HRS	0.0010	-0.0001	0.0021
T/C	0.2564	0.0569	0.4558
T/C × Season	-0.5984	-0.8984	-0.2985

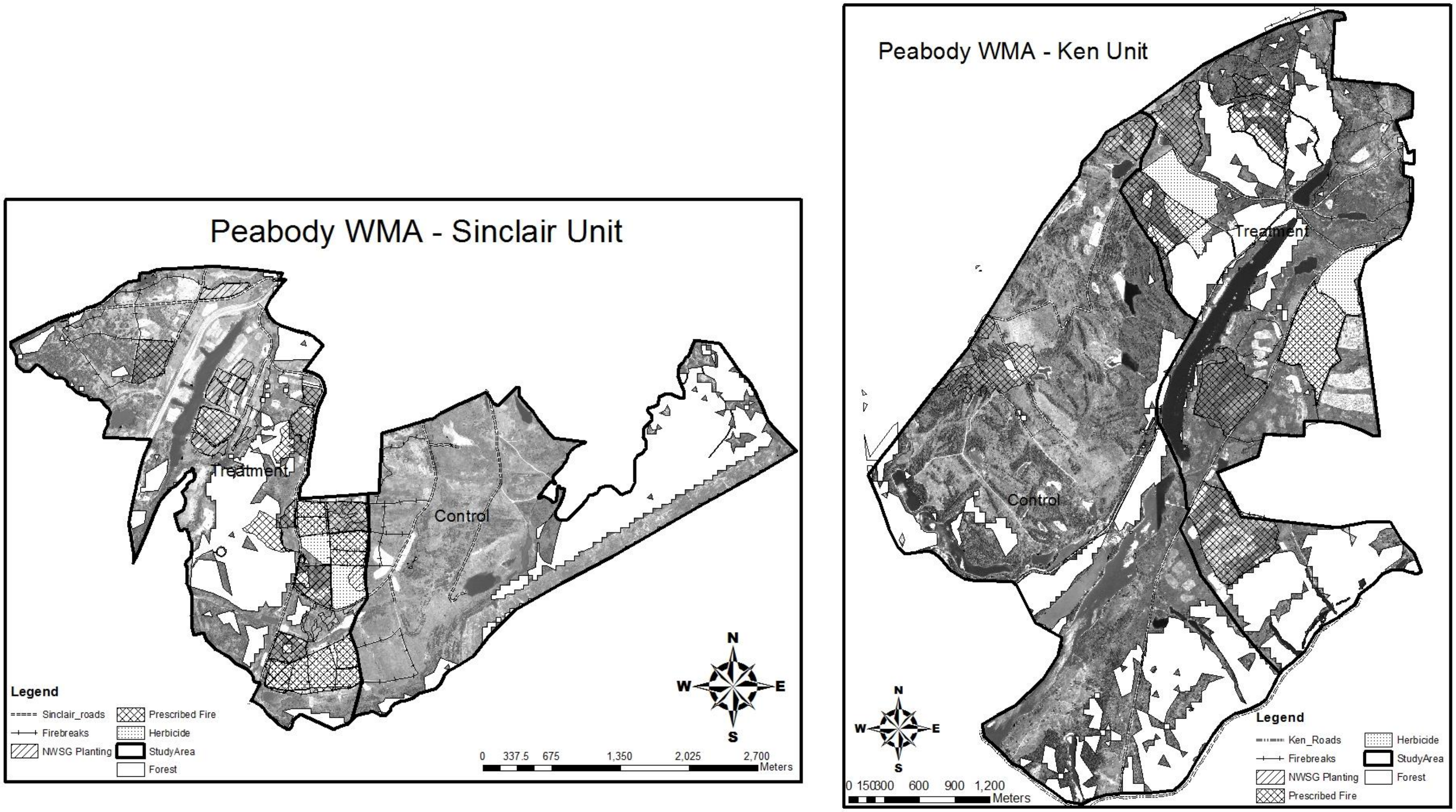

Figure A.1. Map of total treatments conducted on the Sinclair and Ken units on Peabody WMA, Ohio and Muhlenberg Counties, Kentucky, USA, 1 Oct 2009-30 Sep 2013.

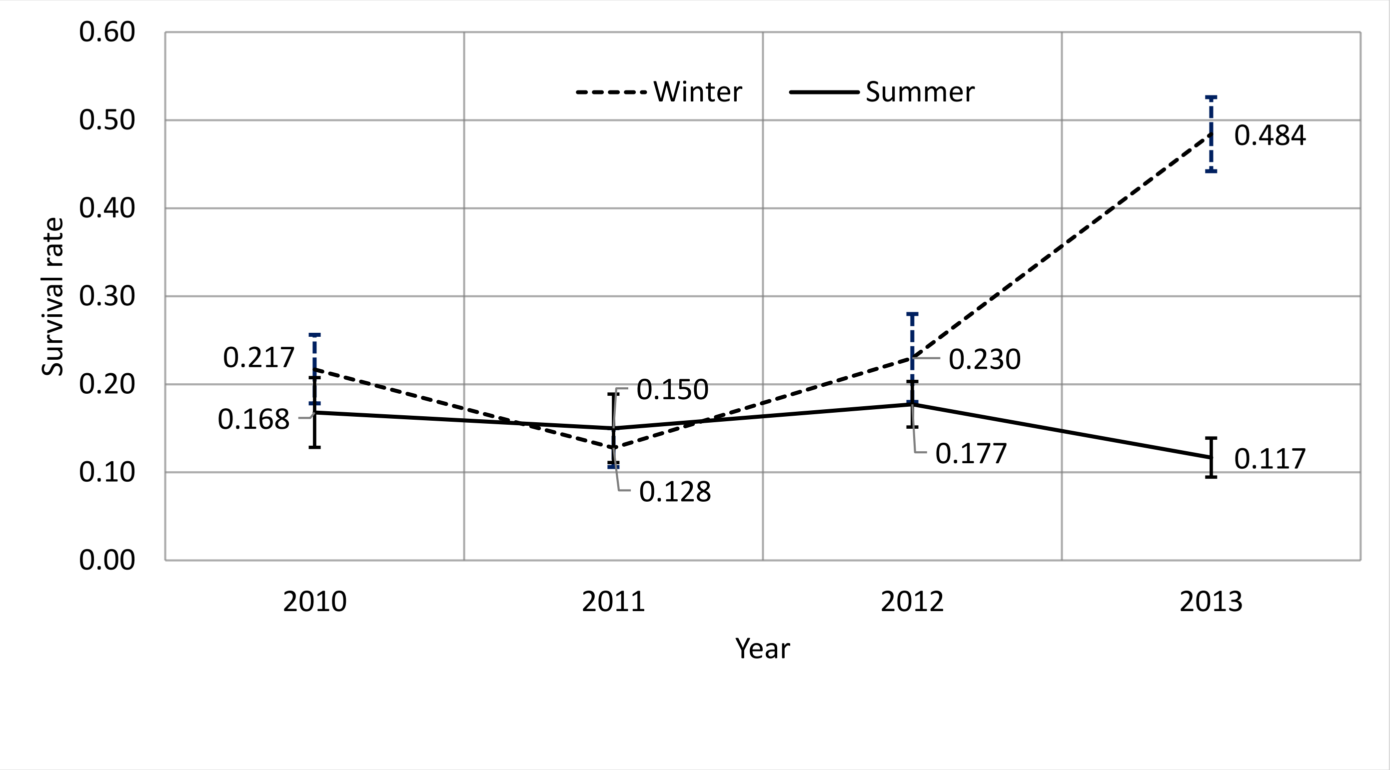

Figure A.2. Seasonal survival rates and confidence intervals of radiomarked northern bobwhite as a function of year on Peabody WMA, Ohio and Muhlenberg Counties, Kentucky, USA, 1 Sep 2009-30 Sep 2013.

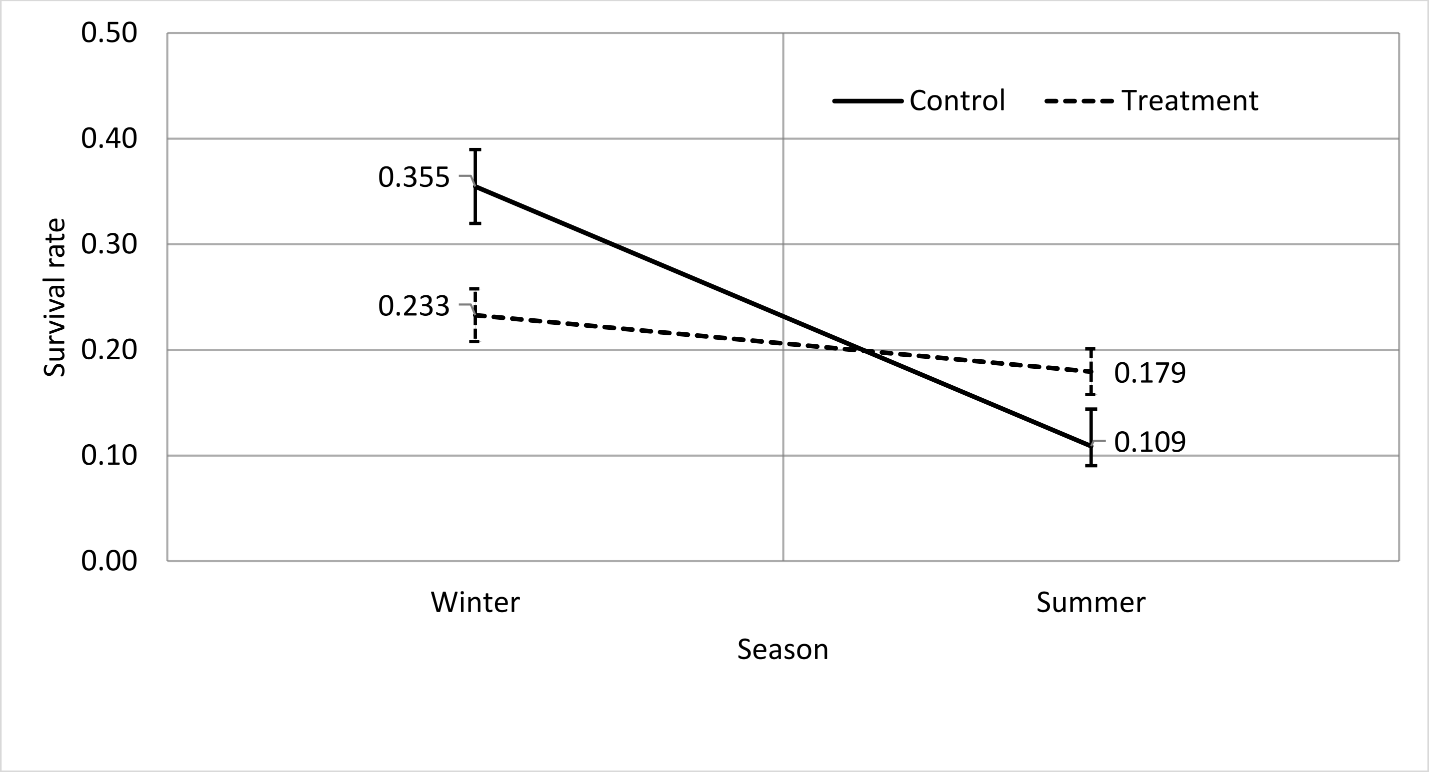

Figure A.3. Treatment and control survival rates and confidence intervals of radiomarked northern bobwhite as a function of season on Peabody WMA, Ohio and Muhlenberg Counties, Kentucky, USA, 1 Sep 2009-30 Sep 2013.

Table A.10. Summary of home range and landscape metrics by site and season used to evaluate survival of northern bobwhite on Peabody WMA, Ohio and Muhlenberg Counties, Kentucky, USA, 1 Sep 2009-30 Sep 2013.

		Sinclair				Ken			
		Winter		Summer		Winter		Summer	
Metric	Scale	$\bar{x}$	SE	$\bar{x}$	SE	$\bar{x}$	SE	$\bar{x}$	SE
FOR (%)	Home range	12.9	1.4	3.4	0.7	11.2	1.2	7.9	0.7
SS (%)	Home range	28.8	1.3	19.9	1.3	34.6	1.5	30.6	1.3
NWSG (%)	Home range	4.2	0.5	5.6	0.7	14.5	1.2	16.6	1.1
OH (%)	Home range	49.9	1.7	64.0	1.6	32.2	1.6	36.9	1.6
Home range size (ha)	Home range	24.7	1.5	30.1	3.2	33.9	2.0	46.5	3.8
FOR core area (ha)	Landscape	1.4	0.1	0.7	0.1	1.7	0.1	1.4	0.1
SS core area (ha)	Landscape	2.3	0.1	2.1	0.1	3.5	0.1	3.2	0.1
NWSG core area (ha)	Landscape	0.5	0.0	0.5	0.1	1.6	0.1	1.8	0.1
OH core area (ha)	Landscape	5.8	0.1	6.8	0.1	3.3	0.1	3.6	0.1
Contagion index (%)	Landscape	52.3	0.8	55.1	0.9	48.9	0.7	48.5	0.7
FOR and open edge density (m/ha)	Landscape	2.2	0.2	1.4	0.2	1.1	0.1	1.3	0.1
SS and open edge density (m/ha)	Landscape	27.0	0.8	27.5	0.8	30.2	1.0	30.8	1.0

Table A.11. Summary of microhabitat metrics used to evaluate survival of northern bobwhite on Peabody WMA, Ohio and Muhlenberg Counties, Kentucky, USA, 1 Sep 2009-30 Sep 2013.

	Sinclair				Ken			
	Winter		Summer		Winter		Summer	
Metric	$\bar{x}$	SE	$\bar{x}$	SE	$\bar{x}$	SE	$\bar{x}$	SE
Canopy3 (m)[a]	1.93	0.13	1.37	0.06	1.88	0.12	1.38	0.07
Litter depth (cm)	2.68	0.23	2.05	0.12	3.00	0.22	2.38	0.21
Midstory (stems/ha)[b]	805.25	125.31	852.09	172.57	836.94	124.07	369.41	109.92
Sight tube (cm)[c]	-	-	88.62	7.81	-	-	76.63	4.58
Distance to edge (m)	-	-	42.91	6.25	-	-	45.76	5.82
NWSG species[d]	-	-	0.12	0.02	-	-	0.18	0.03
Forb species[d]	-	-	0.15	0.01	-	-	0.23	0.02
Ragweed[d]	-	-	0.07	0.02	-	-	0.06	0.01
Shrub species[d]	-	-	0.10	0.02	-	-	0.06	0.01
Cool-season grasses[d]	-	-	0.06	0.01	-	-	0.03	0.01
Maximum herbaceous vegetation height (cm)	47.46	3.67	-	-	65.84	4.54	-	-
Distance to woody cover (m)	5.78	1.30	-	-	6.28	1.16	-	-

[a] Average coverage of vegetation on Nudds board (1.25-1.5 m) centered on bird locations identified through radio-telemetry.

[b] Average stem density of species < 11.4 cm dbh within 5 m radius plot (summer) and 10 m radius plot (winter) centered on bird locations identified through radio-telemetry.

[c] Average ground sighting distance viewed 20.3 cm above ground level at 0, 10, 20, and 30 m along 30 m transect on bird centered locations identified through radio-telemetry.

[d] Average proportion of cover based on 30 m transect centered on bird locations identified through radio-telemetry.

PART III

IMPACT OF EXPERIMENTAL HABITAT MANIPULATION ON NORTHERN BOBWHITE NEST SURVIVAL ON A RECLAIMED SURFACE MINE

ABSTRACT Reclaimed surface mines represent an opportunity to provide large tracts of early successional habitat that could potentially make important contributions to northern bobwhite (*Colinus virginianus*) conservation. However, information regarding bobwhite nesting ecology and how management impacts nesting success on such sites is lacking. Furthermore, reclaimed mines are often dominated by non-native vegetation, such as sericea lespedeza (*Lespedeza cuneata*) and tall fescue (*Schedonorus phoenix*), that has been reported to provide poor structure for northern bobwhite nesting and may limit reproductive success. Therefore, we conducted an experiment to determine if habitat management practices could improve nesting success on a 3,330-ha reclaimed surface mine, Peabody WMA, in western Kentucky. Two units of the site (Sinclair and Ken, 1471 and 1853 ha, respectively) were each divided into a treatment (disking, burning, herbicide application) and control. Treatments were applied October 2009 - September 2013. We used radio telemetry to monitor northern bobwhite (n = 655) during the breeding season (1 Apr-30 Sep) from 2010 to 2013. We used the nest survival model in Program MARK to estimate daily nest (n = 127) survival rates to determine if treatments had an impact. We also included group, landscape (210-m radius), and microhabitat metrics as covariates to help improve sensitivity and to further elucidate experimental impacts. The influence of factors related to the experiment (site, year, and treatment) and three habitat covariates (percent cover of shrub vegetation surrounding a nest, litter depth, and herbaceous canopy cover) were retained in models with $\Delta AIC_c < 2.0$, but beta estimates did not differ from zero for any of these factors. Nest age ($\beta = 0.641$, CI = 0.372-0.911) and nest initiation date ($\beta = 0.022$, 95% CI = 0.001-0.043) were the most influential covariates for nest survival. Nest success (0.352 ± 0.062, 23-day period) on a reclaimed surface mine was lower than reported in other studies and was not influenced by our habitat treatments.

The northern bobwhite quail (*Colinus virginianus*) (hereinafter, bobwhite) is an important game bird that has been declining for much of the last 40 years (Brennan 1991). Throughout the bobwhite's range there has been a 3.8% decline annually (Sauer et al. 2011). Significant local declines have also been reported. In the Central Hardwood Bird Conservation Region (CHBCR), there has been a 4.0% decline annually from 1966-2010 and a 4.9% decline since 2000 (Sauer et al. 2011). There are many hypotheses thought to explain this decline such as land use change (Brennan 1991), extreme weather (Hernandez et al. 2005, Lusk et al. 2001), predation (Mueller et al. 1999, Palmer et al. 2005, Staller et al. 2005), hunting pressure (Guthery et al. 2004, Madison et al. 2002), and loss and fragmentation of habitat (Fleming and Giuliano 2001). Poor reproductive success also may be a factor in the long-term, range-wide decline of bobwhites.

The reproductive ecology of bobwhites is the least understood part of their life history and studies have shown that poor quality nesting and brood-rearing habitats are major limiting factors for bobwhite (Taylor et al. 1999*b*, Dimmick et al. 2002). Bobwhites depend on their prolific reproduction (i.e. renesting and double clutching) to compensate for high annual mortality (Roseberry and Klimstra 1984). Without adequate nesting habitat, populations may not be sustainable (Dimmick et al. 2002). To manage nesting cover properly, it is important to understand what factors of nesting habitat contribute to improved nest survival and requirements specific to local bobwhite populations.

Although, scale-dependent habitat metrics have been linked to bobwhite survival (Seckinger et al. 2008, Holt et al. 2009, Janke 2011), few have linked scale-dependent vegetation metrics to nest survival (Taylor et al. 1999*a*, Potter et al. 2011, Tanner 2012). In Kansas, Taylor et al. (1999*a*) compared nest success to macrohabitat surrounding nest sites and found no differences resulting from habitat composition or diversity at a landscape scale. At the microhabitat scale, successful nests had taller vegetation and less shrub cover and litter than unsuccessful nests (Taylor et al. 1999*b*). Furthermore, few studies have assessed the influence of direct, active habitat manipulation on nest success (Potter et al. 2011). In southeastern Iowa, Potter et al. (2011) evaluated nest success on managed and unmanaged landscapes via radiocollaring 158 adults and monitoring 67 nests over a 2-year period. Daily nest survival was lower on land dominated by row-crops than on land managed for bobwhite. Greater nest survival on the managed site coincided with greater availability of presumed nesting cover, such as grassland and roadside. They observed no evidence of multi-scale habitat influences on nest success. Only percent forb cover on the site dominated by row-crops had a significant positive influence on daily survival rate. Successful nests had twice the average forb cover (38%) as unsuccessful nests (19%); however, this influence was minimal based on its beta value.

Site-specific habitat management has been and will continue to remain the core strategy for bobwhite recovery (Dimmick et al. 2002, McKenzie 2009). However, management success is scale-dependent such that a given level of management intensity is more effective when conducted on a larger scale (Guthery 1997, Williams et al. 2004). One opportunity to manage large tracts of land for bobwhite and that lends itself to conducting habitat manipulations in an experimental framework is reclaimed surface mines. Over 600,000 ha have been reclaimed throughout the eastern United States under the Surface Mining Control and Reclamation Act of 1977 (SMCRA). However, the main focus has been to prevent erosion and this has led to the establishment of undesirable species, such as sericea lespedeza (*Lespedeza cuneata*; Eddy 1999). Nonetheless, surface mines may substantially contribute to bobwhite conservation because of the scale of land that can be managed.

However, efforts are needed to understand how best to manage reclaimed mined lands that are dominated by non-native species that have been considered to provide poor habitat for bobwhite. Surface mines provide a good opportunity to better understand habitat relationships and to explore these relationships in an experimental setting because of the consistent nature of habitat and the scale at which it occurs thus making large-scale manipulations possible and without the limited inferences smaller or fragmented landscapes would impose. Furthermore, few studies have addressed the potential of reclaimed mined lands for bobwhite (Bekerle 2004, Stauffer 2011, Tanner 2012), and no research has been conducted in the context of habitat manipulation on these areas. In southwest Virginia, Stauffer (2011) conducted a pilot study to evaluate the potential of reclaimed mines as habitat for bobwhite quail and found they supported poor to moderate quail numbers; however, reclamation on his site had been focused on preventing runoff rather than providing bobwhite habitat. Tanner (2012), assessed scale-dependent habitat metrics and their influence on bobwhite survival on Peabody Wildlife Management Area (PWMA), a reclaimed surface mine in Kentucky, USA from 2010-2011. He found no evidence that nest survival was linked to microhabitat or landscape metrics, but his research was not focused on habitat management.

Therefore, we conducted research on reclaimed mine lands to assess the effects of habitat manipulation and to further our understanding of how vegetation affects bobwhite nest survival. Our primary objective was to determine the effects of habitat management on nest success. Additionally, we sought to determine which vegetation attributes contribute to nest success and if these relationships were scale-dependent. To assess the importance of scale on nest success in relation to vegetation attributes, analysis was focused at the microhabitat and landscape scales. We hypothesized that treatments would improve nest survival because large, monotypic blocks of marginal cover associated with sericea lespedeza would be broken up allowing for establishment of annual and perennial grasses that could provide improved nesting cover. Although past workers have found limited impacts on survival associated with various measures of habitat (Taylor et al. 1999*a*, Townsend et al. 2001, Potter et al. 2011), the generally uniform and novel environment of reclaimed mine land may demonstrate different responses leading us to hypothesize that multi-scale vegetation effects on nest survival would be evident as a function of our treatments.

STUDY AREA

The study was conducted on the Sinclair (1471 ha; 37°14'N, 87°15'W) and Ken (1853 ha; 37°17'N, 86°54'W) units of the Peabody Wildlife Management Area (PWMA) in Muhlenberg and Ohio counties, Kentucky, USA. Both units are reclaimed surface mine sites dominated by early-successional vegetation communities. Soils on both units consisted primarily of udorthents, which are characteristic of reclaimed mine sites. Areas on the WMA with trees were established pre-SMCRA (1977), whereas all early successional areas were established post-SMCRA. Sericea lespedeza, established during reclamation, constituted much of the vegetation on both units. From 2000-2004, native warm-season grasses (NWSG), including mixtures of big bluestem (*Andropogon gerardii*), little bluestem (*Schizachyrium scoparium*), indiangrass (*Sorghastrum nutans*), and switchgrass (*Panicum virgatum*), were established on both units. Soil compaction during reclamation led to slow growth of American sycamore (*Platanus occidentalis*), eastern cottonwood (*Populus deltoides*), oaks (*Quercus* spp.), and maples (*Acer* spp.), which were the dominant tree species planted during reclamation throughout both units. Shrubs occurred across both areas and included patches of volunteer blackberry (*Rubus fruticosus*), smooth sumac (*Rhus glabra*), and coralberry (*Symphoricarpos orbiculatus*). Annual food plots were maintained on both units, and consisted primarily of a mixture of grain sorghum (*Sorghum bicolor*), corn (*Zea mays*), pearl millet (*Pennisetum glaucum*), browntop millet (*Urochloa ramose*), soybeans (*Glycine max*), and annual sunflower (*Helianthus annuus*).

METHODS

Study Design

We used an experimental approach to test our research hypothesis. Since no birds were ever documented to move between our two experimental units, we considered them separate sites. The two study sites were divided into approximately equal halves (Figure B.1), each with similar proportions of vegetation types that occur on the property (Table B.1). One half of each study site was randomly assigned to receive intensive management treatments (disking, burning, and herbicide application) during 2009-2013, whereas the other half of the site was minimally disturbed and served as a control (two replicates of treatment and control) (Figure B.1). Efforts were focused on maintaining early succession while trying to limit encroachment of invasive non-native plants that were previously established. The Sinclair control and treatment units were 673 and 798 ha, respectively, whereas the Ken control and treatments units were 1,043 and 810 ha, respectively (Table B.1). Treatments applied varied by year and season because of weather and logistical constraints. A total of 963 ha (Sinclair = 543 ha, Ken = 420 ha) were treated during the four years.

Land Cover

Four major vegetation types (OH, SS, FOR, and NWSG), which constituted 91% of the total land cover on our study site (Table A.1), were delineated based on aerial imagery in ArcGIS 9.3 (ESRI, Redlands, CA, USA). To delineate between forest, shrubland, and open vegetation, 1-

m resolution aerial imagery (2010) was used from the National Agriculture Inventory Program, US Department of Agriculture, Farm Service Agency. We selected representative woody cover on our study site as a template for reclassifying all 1x1-m cells as either "woody" or "open" with the Image Analyst tool in ArcGIS. We then used the Aggregate Tool to create unique polygons of "woody" or "open" vegetation with a minimum size of 0.2 ha, which was the average size of the smallest habitat management activity (disking) implemented on the site. To delineate between open vegetation, shrubland vegetation, and forest, we used percentage breaks within our individual raster cells based on the percent of woody vegetation present within each 0.2-ha polygon. We classified polygons with <10% woody cover as open vegetation, those with 11-55% woody cover as shrubland, and those with >56% woody cover as forest. Forest vegetation had a mean basal area (stems >10 cm DBH) of 20.9 m^2/ha (SE = 1.77) and shrubland 9.6 m^2/ha (SE = 1.23); shrubland stems were typically 10–20 cm DBH. We classified NWSG by mapping areas comprised of $\geq$51% native grass using ArcPad 8.0 (ESRI, Redlands, CA, USA) on handheld Global Position System (GPS) units (Trimble Navigation Limited, Inc., Sunnyvale, CA), and classified areas that had <51% native grass as open herbaceous. All classifications were subjected to ground-truthing to validate GIS-based assignments of vegetation types.

Trapping and Radio Telemetry

We captured bobwhites during all months of the year in support of a larger telemetry project using funnel traps (Stoddard 1931, Palmer et al. 2002) baited with cracked corn and grain sorghum placed throughout the study area at known covey locations as well as locations that appeared to contain suitable cover for bobwhite. The breeding season was defined as 1 April-30 Sep (Burger et al. 1995*a*). We covered traps with burlap and surrounding vegetation to reduce stress and predation on captured individuals. Traps were set $\geq$5 days per week and checked once daily in the evening. Upon meeting a minimum body mass of 90 g, each captured bobwhite was banded with aluminum bands (both legs) and, if body mass was >120 g, fitted with a necklace-style radio transmitter (crystal-controlled, two-stage design, pulsed by a CMOS multivibrator, American Wildlife Enterprise, Monticello, Florida, USA) that weighed ≤6.5 grams. Corteville (1998) reported that radiotransmitters may slightly reduce annual survival, but we assumed the magnitude of bias was consistent between treatment and control sites on our study area (Palmer and Wellendorf 2007, Terhune et al. 2007). We also recorded the sex, age, weight (g), and overall condition. Sex was determined by plumage and age was determined by the presence or absence of buff-tipped primary coverts. Condition was rated on a scale of 1 to 6 with 1 being excellent condition and 6 being extremely lethargic or injured. Birds were not collared if condition was >4. We also sought to maintain a sex ratio of radiomarked birds favoring females, if possible, to help adjust for typical male-biased (Roseberry and Klimstra 1984) populations and to increase the sample size of nests. Our trapping and handling methods complied with protocols of University of Tennessee Institutional Animal Care and Use Committee Permit 2042-0911.

Radiomarked birds were located ≥3 times/week using a scanning receiver and a handheld Yagi antenna (Advanced Telemetry Systems, Inc., Isanti, MN). However, nesting birds were located daily to monitor nest success. We used the homing method (White and Garrot 1990) by walking to ≤50 m of the bird to minimize disturbance. We recorded the distance and azimuth to the bird by assessing the strength and direction of the telemetry signal. We then recorded the Universal Transverse Mercator (UTM) coordinates of the observer on a GPS unit (Garmin GPSMAP 60CSx, Garmin International, Inc. Olathe, KS, USA) and used the distance and azimuth to estimate the actual location of each bird. Each bird was located at different times throughout the week to capture the variability in daily movements and habitat use. We assumed birds located 2-3 times at the same site to be potential nesters and placed flagging >10 m away from the presumed nest location. Once the radiomarked bird was away from the nest, we located the nest, and recorded the number of eggs and UTM coordinates. Incubating birds were monitored once daily until hatch, nest failure, or adult mortality occurred (DeVos and Mueller 1993). We monitored nests daily and visually assessed nests at least once per week when incubating adults were away from the nest on feeding excursions (Taylor et al. 1999*a*). After the radiomarked bird and brood had vacated the nest site, nests were examined to determine hatching success (DeMaso et al. 1997). The number of chicks hatched per nest was determined from egg shell remains at the nest site (DeVos and Mueller 1993).

Vegetation Sampling

Within 7 days of nest termination, we collected microhabitat vegetation metrics at both successful and unsuccessful nests. These included litter depth (cm; LitterDepth), woody stem density (stems/ha; WoodyStem), grass cover (%; Grass), the structure of vegetation at ground level (Nudds board cover 0-0.25 m; GroundLayer), the structure representing herbaceous vegetation height (Nudds board cover 0.75-1.25 m; HerbLayer), distance to bare ground (m; DtoBG), distance to edge (m; DtoED), and vegetation type (Vegtype) in which the nest was located (Table B.4). We collected DtoBG, DtoED, and Vegtype over all 4 years. In an effort to identify smaller scale influences on nest success, the five additional microhabitat metrics were added the last 2 years.

Microhabitat vegetation composition and litter depth metrics were collected from a 1x1 m plot centered on nest locations. Species composition was collected at 0.25, 0.5, 0.75, and 1 m in each cardinal direction from the nest location generating 16 subsamples per nest plot. Grass was calculated as the number of subsamples with grass species present divided by the total number of subsamples. Litter depth measurements were taken at 0.5 m from the nest location in each cardinal direction and measured to the nearest 0.5 cm; these 4 measurements were averaged to provide each nest with a single litter depth value. We estimated WoodyStem based on stems <11.4 cm dbh within a 5-m radius plot centered on each nest location. Vegetation structure was assessed using a Nudds board (Nudds 1977), which was 2 m tall and consisted of 8, 0.25-m strata. The Nudds Board was centered on the nest location and an observer determined the amount of vegetation covering each stratum (0 = no vegetation, 1 = 0-20%, 2 = 21-40%, 3 = 41-60%, 4 = 61-80%, and 5 = 81-100%) from a distance of 5 m from a kneeling position in each cardinal direction. Based on the 4 visual obstruction readings from each cardinal direction, an average cover for each strata was estimated for each nest. We calculated GroundLayer as the mean cover of stratum 1 (0-0.25 m) and HerbLayer was calculated as the mean cover of strata 4 and 5 (0.75-1.25 m). We estimated DtoBG and DtoED using a rangefinder. We considered bare ground to be exposed soil with no vegetative cover and edge to be where two different delineated vegetation types met. Vegetation type was recorded as either OH, NWSG, SS, or DF based on nest location.

We used ArcGIS 9.3 to create a buffer with a 210-m radius (Taylor et al. 1999*a*, Potter et al. 2011) around each nest to calculate landscape-scale habitat metrics. Within each buffer, we calculated landscape metrics using FRAGSTATS (McGarigal and Marks 1994) using a 150-m moving window that allowed us to capture variation from adjacent vegetation cells. The 7 landscape metrics we calculated were: forest/open vegetation edge density, shrub/open vegetation edge density, core area of all four major vegetation types (using a 30-m edge effect), and a contagion index (Table B.3). These were selected *a priori* based on their influence on bobwhite habitat. We also calculated the percent of each of our four vegetation types within the buffer for vegetation composition covariates at the landscape level (Table B.3).

Nest Survival Analysis

We used the nest survival model within Program MARK (White and Burnham 1999) to estimate daily survival rate (DSR) of nests across the 4 years of our study. Nest survival was defined as the probability of a nest surviving the 23-day incubation period (Rosene 1969, Potter et al. 2011). A single analysis with a 3-stage modeling process consisting of biological, experiment, and habitat covariates was conducted to assess the value of all covariates (Doherty et al. 2012) on nest survival. Habitat covariates included vegetation composition, landscape, and microhabitat metrics. We selected models from each stage based on values from Akaike's Information Criterion (ΔAIC_{c}) and AIC_c weights. A ΔAIC_c value of <2 (Burnham and Anderson 2002) was used to determine the usefulness of a model for explaining variance in survival.

The first stage of analysis (biological) included group covariates: linear time (variation from day 1 to day 136 of nesting season), nest age, nest initiation date, and constant time effects (Table B.2). We examined additive, interaction, and quadratic models within this first stage. The top model from the first stage was then used as the baseline model in the second stage of analysis (experiment). Experimental covariates (Site, Year, and T/C; Table B.2) were then combined with the top biological model as additive models. We also ran additive and interaction models among treatment covariates exclusively to test our hypothesis concerning treatment effects on nest survival. For the third stage of analysis, we added habitat covariates (landscape, microhabitat, and vegetation composition; Table B.3, B.4). All covariates from the second and third stage of analysis were run in additive models containing the top biological model to incorporate effects among multiple covariates. After analyzing all 3 model stages, we arrived at the top model based on the ΔAIC_c score and used this model for estimating DSR and overall nest survival. We also computed the model-averaged parameter estimates for DSR using the survival estimate from each model. The delta method (Powell 2007) was used to expand our survival estimates to a temporal scale that encompassed the 23-day incubation period.

RESULTS

We captured 1,002 bobwhite (445 males, 284 females, and 273 unknowns) of which 655 were fitted with radiocollars and monitored during the 4 breeding seasons. A total of 127 nests were found, but we were only able to use 124 ($n = 68$ on Sinclair; $n = 56$ on Ken) for analysis. To avoid disturbance and probability of nest abandonment, the remaining 3 nests that were located without a radiomarked adult associated with the nest were not monitored.

The best biological model from our first stage of analysis was $\{NestA^2+NestA\times NestI\}$ and was used as the baseline model in our second (experiment models) and third (habitat models) stage of analysis (Table B.6). After incorporating experiment and habitat covariates, the model $\{NestA^2+NestA\times NestI\}$ was still the top model based on the ΔAIC_c value and AIC_c weight (Table B.6). Based on beta estimates and confidence intervals, all variables within the top model were significant (Table B.7). The most important covariates were nest age and nest initiation date, both having a positive relationship with nest survival. However, the effect of nest initiation date ($\beta = 0.022$, 95% CI = 0.001-0.043) was of less total magnitude in explaining variation in DSR than nest age ($\beta = 0.641$, CI = 0.372-0.911). The quadratic $\{NestA^2\}$ and interaction $\{NestA\times NestI\}$ terms both showed a significant negative relationship with nest survival. The model receiving the second most support included SS (percent of shrub cover within 210-m radius around nest) and was virtually indistinguishable from the top model $\{NestA^2+NestA\times NestI\}$, having Akaike weights of 0.114 and 0.113 (Table B.6). However, the beta value for SS was not different from 0 ($\beta = 0.014$, 95% CI = -0.006-0.033; Table B.7) suggesting that the effect of this covariate was minimal.

Overall DSR based on the top model {$\text{NestA}^2+\text{NestA}\times\text{NestI}$} was 0.956 (SE = 0.009) and 0.352 (SE = 0.062) when expanded to the 23-day incubation period. The model-averaged estimates for DSR across all models had a range of 0.941-0.957. The best model we identified indicated that DSR increased from day 1 to day 23 of incubation (NestA) (Figure B.1) and from day 1 to day 136 of the nesting season (NestI). There were a total of eight models with a ΔAIC_c <2, with 1 landscape, 2 microhabitat, and 3 experiment covariates (Table B.6). However, the effect of covariates other than nest age and nest initiation within these models were marginal based on the fact that their beta values were not different from 0 (Table B.7). Models with landscape and vegetation composition metrics measured within nest buffers were not predictive of nest success as only one covariate (SS) was contained in a model with a ΔAIC_c <4. Single variable covariate models had no support (ΔAIC_c >4).

DISCUSSION

We found no evidence that our treatments had an effect on nest survival. Although, all 3 of our experimental variables (Site, Year, and T/C) were included in models with a ΔAIC_c <2, their beta values were not different from 0 suggesting they had a weak influence on nest survival. We expected treatments to improve nest survival by reducing marginal cover associated with sericea lespedeza allowing for establishment of annual and perennial grasses that could provide improved nesting cover. Data from PWMA support this hypothesis with disking leading to a decrease in mean percent cover of sericea lespedeza (disked = 70.7, untreated = 75.8) and an increase in native warm-season grasses (disked = 10.5, untreated = 6.5; University of Tennessee, unpublished data). However, these increases were short-lived and in any case, did not impact nest success at a level we could detect, suggesting management (disking and herbicide application) may need to be implemented on a shorter rotation (e.g., 2 years) or more extensively to improve results.

Our nest survival estimate was higher than those observed on an unmanaged site in southeast Iowa (0.277; Potter et al. 2011) and north-central Kentucky during one of two years (0.242; West et al. 2012). However, nest survival was lower than published estimates from studies conducted in northern Missouri (0.437; Burger et al. 1995*b*), western Oklahoma (0.48; Cox et al. 2005), east-central Mississippi (0.40; Taylor and Burger 1997), southern New Jersey (0.454; Collins et al. 2009), southern Texas (0.384; Rader et al. 2007), Florida (0.476; Brinkley 2011), and on a managed site in southeastern Iowa (0.495; Potter et al. 2011).

Though predation was a likely proximate cause of low nest survival, another plausible explanation was a lack of nutrition during the breeding season. There is indication of nutritional deficiency as evidenced by our average clutch size (12.5 ± 3.2), which was somewhat lower than the 14.4 and 13.7 reported by Stoddard (1931) and Roseberry and Klimstra (1984), respectively. Similarly, our hatching success (85.5 ± 2.9%) was lower than reported by most studies (85-95%; Sandercock et al. 2008). Furthermore, our estimates of renesting and double-clutching were exceptionally low as no males and only 13.8% of females (n = 12) were known to renest after a failed nesting attempt. There were only two instances of double-clutching that occurred during our study.

PWMA was composed primarily of large blocks of 1-m tall sericea lespedeza. Indeed, this structural arrangement certainly provided nesting cover for bobwhite, albeit cover that resulted in only marginal nesting success. However, hard-seeded plant foods, such as sericea lespedeza, are virtually indigestible and provide no nutritional benefit (Davison 1958). In addition, sericea lespedeza litter accumulation has been associated with reduced forb establishment and species richness (Foster and Gross 1998) and may actively interfere with germination of other species through allelopathic compounds generated in leaf and stem tissue (Adams et al. 1973, Wade 1989). Furthermore, studies have reported invertebrate use of sericea lespedeza is low compared to other plant species (Menhinick 1967, Bugg and Dutcher 1989, Eddy 1999). During the nesting and chick-rearing season, both hens and chicks require a large proportion of invertebrate protein in their diet (Eubanks and Dimmick 1974, Wiens and Rotenberry 1979). With this is mind, forbs as well as other beneficial native plant species may have been lacking on our site because of the dominance of sericea lespedeza; thus, reducing invertebrate populations and leading to lower hen condition as well as sub-optimal foraging conditions for newly hatched chicks.

We found that the age of a nest during incubation and time a nest was initiated during the nesting season had the greatest effect on nest survival. The probability of a successful nest increased throughout the 23-day incubation period as well as the nesting season. Nest age and nest initiation were also influential for nest success of mountain plovers (*Charadrius montanus*) in Montana (Dinsmore et al. 2002). Dinsmore et al. (2002) reported daily survival rate of nests increased as nest age increased as well as throughout the nesting period. In addition, Klett and Johnson (1982) detected a decline in daily mortality rates as the age of mallard (*Anas platyrhynchos*) and blue-winged teal (*Anas discors*) nests increased and the decline was sharpest during the first 10 days.

There are reasons for suspecting that daily nest survival rates might vary throughout the nesting season. As the nesting season progresses, the vegetation in which nests are placed increases in height and density, providing more concealment from predators. Also, as the season progresses, alternative food for predators (buffer prey) may become increasingly available and help reduce mortality of bobwhite (Stoddard 1931, Miller 2011). Furthermore, differences in vulnerability of nest sites to predators (Ricklefs 1969) may mean nests in high-risk sites are quickly found and destroyed by predators, whereas well-placed nests have less of a chance of being located by a predator. There is some evidence that the vulnerability of nests on our study area was related to high-risk areas. For example, several depredated or abandoned nests were located near roads or firebreaks that were subject to disturbance by the public or served as travel lanes for predators.

Another plausible explanation for increasing survival over time is the presence of the incubating parent at the nest site. The cryptic coloration of bobwhite may serve as a camouflage to an exposed nest of eggs. The amount of time spent on nests has been shown to have an inverse relationship to daily mortality rates, which suggests that the presence of the incubating bobwhite is a deterrent to some egg predators (Klett and Johnson 1982). Incubating bobwhite spend increasingly more time at the nest site as the incubation period progresses and, except for occasional daytime feeding excursions, are present day and night during incubation. The increased “protection” provided to the nest may positively affect the success of the nest. The distracting behavior of incubating birds if they get flushed by predators might also serve to reduce egg predation.

In southeastern Iowa, Potter et al. (2011) evaluated nest success on managed and unmanaged landscapes via radiocollaring 158 adults and monitoring 67 nests. In contrast to our results, they found DSR decreased as nest age increased. They attributed their findings to increased scent and sign around nest locations as a result of daily feeding excursions. However, the decrease was more severe on the unmanaged site that was dominated by agricultural production and therefore, could have been subject to more human and predator disturbance compared to the site managed for bobwhite.

We did not find any associations between nest success and the covariates we tested at the landscape scale as no metrics had a beta value that differed from zero. Although SS was included in our second best model, which was only 2.0 times less likely than the top model {NestA^2+NestA×NestI}, its influence on nest survival was negligible. Past studies support our results regarding landscape metrics having minimal impact on nest success (Taylor et al. 1999*a*, Staller et al. 2002, Potter et al. 2011). Taylor et al. (1999*a*) found no association of landscape composition on clutch success in the Flint Hills region of Kansas. Staller et al. (2002) observed only minor differences in macrohabitat composition between successful and unsuccessful nests. Likewise, no support for landscape or vegetation composition metrics on influencing nest success were found in southeastern Iowa (Potter et al. 2011).

In addition, nest survival was not significantly related to any of our microhabitat metrics (Table B.7). Although, we hypothesized that treatments would improve nesting cover by allowing for the establishment of annual and perennial grasses, no compositional metrics related to grass cover were supported in competing models. The rapid recovery of sericea lespedeza following disturbance (3-year disk block = 70.7, untreated areas = 75.8; University of Tennessee, unpublished data) may have contributed to this lack of support. Furthermore, the extent of this treatment (21% of open herbaceous cover annually) may have been inadequate. Townsend et al. (2001), Lusk et al. (2006), and Collins et al. (2009) also found no significant relationship between microhabitat structure and composition on nest survival. Although not significant, Collins et al. (2009) reported bobwhites selected nest sites with greater visual obstruction and percent litter cover. Townsend et al. (2001) reported bobwhites selected nest sites with more woody cover and percent grass cover, and a lower percent of bare ground on prairies in Oklahoma. Likewise, Lusk et al. (2006) reported nest site selection in Texas was positively related to vegetation height and percent shrub cover, and negatively related to percent bare ground.

However, landscape (SS) and microhabitat (LitterDepth, HerbLayer) metrics related to nest concealment, although non-significant, were better predictors of nest success than other metrics tested. The beta estimates for both LitterDepth and SS were positive indicating that as the cover of these metrics increased the probability of a nest being successful increased. Lusk et al. (2006) and Collins et al. (2009) suggested that vegetation characteristics that improve nest concealment were important to nest site selection of bobwhite. Also, increased canopy cover that promotes nest concealment has been associated with providing increased visual and olfactory protection from predators (Mankin and Warner 1992, Patterson and Best 1996).

On the other hand, the beta estimate for HerbLayer was negative suggesting that an increase in herbaceous cover decreased nest survival. Because this metric served as a surrogate for cover of sericea lespedeza (0.75-1.25 m), this result may have suggested that breaking up these monotypic canopies may have had some benefit with respect to nest survival. Even if the metric did not directly relate to sericea dominance, reduced cover in this strata was certainly consistent with the disturbance induced by our treatments.

We did not detect evidence that our treatments or multi-scale vegetation metrics had an effect on nest survival. Reduced native seedbanks resulting from surface mining and subsequent reclamation activities combined with establishment of sericea lespedeza, resulted in dominance of our sites by monotypic stands of this species. Herbaceous communities dominated by this species and densely planted stands of native grasses clearly provided marginal nesting habitat as evidenced by our low nest survival rates. Although predation and/or nest abandonment is likely the proximate cause of low success rate, our estimates of nest survival and elements of fecundity were lower than past research, suggesting that adequate nutrition may have been limiting on our site.

MANAGEMENT IMPLICATIONS

Optimally, initial reclamation efforts on mine sites should incorporate seeding an appropriate mix of native herbaceous vegetation (i.e, forbs, grasses, and legumes) over non-native species. However, current management on reclaimed surface mines should focus on removing non-native monocultures through disking, burning, or herbicide application, all conducted on a short return interval (e.g., 2 years). In addition, seeding native forbs and grasses (at low densities) on disturbed sites could provide more competition for sericea, increased plant diversity, improved insect substrate, improved seed yield, and more, well-distributed grass cover. Increased shrub cover, and likely the interspersion of that cover, may also contribute to improved nest success on reclaimed surface mines. Further experiments across a range of habitat conditions will be required to determine management intensity and duration thresholds required to increase nest survival for bobwhite populations.

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APPENDIX B

Table B.1. Delineated vegetation types and total cover (ha) on Peabody WMA, Ohio and Muhlenberg Counties, Kentucky, USA, 2009–2013.

Site	Unit	Vegetation	Abbreviation	Hectares	Total by unit (ha)	Total (ha)
Ken	Treatment	Annual grain	AG	21.0	810.2	1853.1
		Forest deciduous	FOR	216.6		
		Native warm-season grass	NWSG	93.7		
		Open herbaceous	OH	244.7		
		Scrub shrub	SS	175.4		
		Water	Water	52.6		
		Wetland emergent	WE	6.2		
	Control	Annual grain	AG	4.8	1042.8	
		Forest deciduous	FOR	189.0		
		Native warm-season grass	NWSG	112.0		
		Open herbaceous	OH	279.7		
		Scrub shrub	SS	344.0		
		Water	Water	112.7		
		Wetland emergent	WE	0.6		
Sinclair	Treatment	Forest deciduous	FOR	162.0	798.2	1470.6
		Native warm-season grass	NWSG	24.1		
		Open herbaceous	OH	367.8		
		Scrub shrub	SS	194.9		
		Water	Water	44.1		
		Wetland emergent	WE	5.3		
	Control	Annual grain	AG	6.5	672.5	
		Forest deciduous	FOR	165.3		
		Native warm-season grass	NWSG	33.8		
		Open herbaceous	OH	303.8		
		Scrub shrub	SS	126.8		
		Water	Water	25.5		
		Wetland emergent	WE	10.7		

Table B.2. List and description of group and experiment metrics used to assess effects on nest survival of radiomarked northern bobwhite on Peabody WMA, Ohio and Muhlenberg Counties, Kentucky, USA, 1 Apr 2010-30 Sep 2013.

Metric	Description
T[a]	Linear time
NestA	Nest age (days)
NestI	Nest initiation date
Year	Year (2010-2013)
T/C	Treatment or control
Site	Site, either Ken or Sinclair
Null	Constant time

[a] Variation from day 1 to day 136 of nesting season.

Table B.3. List and description of vegetation composition and landscape metrics (measured within 210-m radius of nest) used to assess effects on nest survival of radiomarked northern bobwhite on Peabody WMA, Ohio and Muhlenberg Counties, Kentucky, USA, 1 Apr 2010-30 Sep 2013.

Metric	Description
FOR	FOR vegetation (%)
SS	SS vegetation (%)
NWSG	NWSG vegetation (%)
OH	OH vegetation (%)
ED_FOR	FOR to open vegetation edge density (m/ha)
ED_SS	SS to open vegetation edge density (m/ha)
FOR_CA	Core area of FOR vegetation (ha)
SS_CA	Core area of SS vegetation (ha)
NWSG_CA	Core area of NWSG vegetation (ha)
OH_CA	Core area of OH vegetation (ha)
CI	Contagion index (%)

Table B.4. List and description of metrics used to assess effects of microhabitat on nest survival of radiomarked northern bobwhite on Peabody WMA, Ohio and Muhlenberg Counties, Kentucky, USA, 1 Apr 2010-30 Sep 2013.

Metric	Description
DtoBG	Distance to bare ground (m)
DtoED	Distance to edge (m)
Vegtype	Vegetation type (FOR, SS, OH, NWSG)
LitterDepth	Litter depth (cm)
WoodyStem[a]	Stem density (stems/ha)
Grass[b]	Grass species (%)
GroundLayer[c]	Nudds board coverage 0-0.25 (m)
HerbLayer[d]	Nudds board coverage 0.75-1.25 (m)

[a] Average stem density of species < 11.4 cm DBH within 5 m radius centered on nest location.

[b] Average percent of grass species coverage within 1x1 m plot centered on nest location.

[c] Average coverage of vegetation on Nudds board (0-0.25 m) centered on nest location.

[d] Average coverage of vegetation on Nudds board (0.75-1.25 m) centered on nest location.

Table B.5. Number of nests and nesting period of radiomarked male and female northern bobwhite on Peabody WMA, Ohio and Muhlenberg Counties, Kentucky, USA, 1 Apr 2010-30 Sep 2013[a].

		Nesting period				
Year	*n*	Start date	Day of period	End date	Day of period	Days
2010	30	5/7/2010	1	9/6/2010	121	121
2011	27	6/1/2011	26	9/7/2011	122	97
2012	32	5/7/2012	1	8/18/2012	103	103
2013	38	5/23/2013	16	9/20/2013	136	120

[a] May 7th was the earliest date a nest was found and was therefore considered the first day of the nesting season across all four years.

Table B.6. Highest ranking models (above Null model) from 65 models based on ΔAIC$_c$ values and AIC$_c$ weights used to assess the influence of group, microhabitat, and landscape metrics on northern bobwhite nest survival on Peabody WMA, Ohio and Muhlenberg Counties, Kentucky, USA, 1 Apr 2010-30 Sep 2013[a].

Model	AICc	ΔAICc	AICc weights	Model likelihood	*k*	Deviance
NestA^2+NestA×NestI	308.5712	0	0.11419	1	5	298.5183
NestA^2+NestA×NestI+SS	308.5865	0.0153	0.11332	0.9924	6	296.5123
NestA^2+NestA×NestI+HerbLayer	309.4668	0.8956	0.07297	0.639	6	297.3927
NestA^2+NestA×NestI+LitterDepth	309.7893	1.2181	0.06211	0.5439	6	297.7152
NestA^2+NestA×NestI+Year	310.309	1.7378	0.04789	0.4194	6	298.2348
NestA^2+NestA×NestI+Year+SS	310.421	1.8498	0.04529	0.3966	7	296.3221
NestA^2+NestA×NestI+Site	310.4745	1.9033	0.04409	0.3861	6	298.4004
NestA^2+NestA×NestI+T/C	310.5458	1.9746	0.04255	0.3726	6	298.4717
NestA^2+NestA×NestI+WoodyStem	310.5735	2.0023	0.04196	0.3674	6	298.4994
NestA^2+NestA×NestI+Site+SS	310.6076	2.0364	0.04125	0.3612	7	296.5086
NestA^2+NestA×NestI+T/C+SS	310.6081	2.0369	0.04124	0.3611	7	296.5092
NestA^2+NestA×NestI+Site+HerbLayer	311.1825	2.6113	0.03095	0.271	7	297.0836
NestA^2+NestA×NestI+Year+HerbLayer	311.3115	2.7403	0.02901	0.254	7	297.2126
NestA^2+NestA×NestI+T/C+HerbLayer	311.3786	2.8074	0.02806	0.2457	7	297.2797
NestA^2+NestA×NestI+Site+LitterDepth	311.6539	3.0827	0.02445	0.2141	7	297.555
NestA^2+NestA×NestI+Year+LitterDepth	311.7647	3.1935	0.02313	0.2026	7	297.6657
NestA^2+NestA×NestI+T/C+LitterDepth	311.8084	3.2372	0.02263	0.1982	7	297.7095
NestA×NestI	312.0451	3.4739	0.0201	0.176	4	304.0099
NestA^2+NestA×NestI+Year+Site	312.1945	3.6233	0.01866	0.1634	7	298.0956
NestA^2+NestA×NestI+T/C+Year	312.2914	3.7202	0.01777	0.1556	7	298.1924
NestA^2+NestA×NestI+Year+WoodyStem	312.33	3.7588	0.01744	0.1527	7	298.2311
NestA^2+NestA×NestI+Site+WoodyStem	312.4699	3.8987	0.01626	0.1424	7	298.3709
NestA^2	312.5539	3.9827	0.01559	0.1365	3	306.5328
NestA^2+NestA×NestI+T/C+WoodyStem	312.5612	3.99	0.01553	0.136	7	298.4623
NestA	312.8548	4.2836	0.01341	0.1174	2	308.8442
NestA+T	313.9051	5.3339	0.00793	0.0694	3	307.884
NestA^2+Year	313.962	5.3908	0.00771	0.0675	4	305.9268
NestA^2+NestA+NestI	313.9683	5.3971	0.00769	0.0673	4	305.933

Continued

Table B.6. Continued

Model	AICc	ΔAICc	AICc weights	Model likelihood	*k*	Deviance
NestA+NestI	314.4687	5.8975	0.00598	0.0524	3	308.4476
$NestA^2$+Site	314.543	5.9718	0.00577	0.0505	4	306.5078
NestA+T/C	314.7982	6.227	0.00508	0.0445	3	308.777
NestI	387.741	79.1698	0	0	2	383.7305
$NestI^2$	389.7309	81.1597	0	0	3	383.7098
LitterDepth	389.9072	81.336	0	0	2	385.8967
WoodyStem	392.0038	83.4326	0	0	2	387.9933
HerbLayer	394.7589	86.1877	0	0	2	390.7484
Null	394.7849	86.2137	0	0	1	392.7814

[a] Notation generally follows that of Lebreton et al. (1992).

Table B.7. List of group, landscape, and micro-habitat metrics with associated beta values contained in models within a ΔAIC_c <2 for nest survival of radiomarked northern bobwhite on Peabody WMA, Ohio and Muhlenberg Counties, Kentucky, USA, 1 Apr 2010-30 Sep 2013.

Metric	Beta value	LCI	UCI
NestA	0.641	0.372	0.911
NestI	0.022	0.001	0.043
$NestA^2$	-0.011	-0.020	-0.002
NestA×NestI	-0.003	-0.005	-0.001
SS	0.014	-0.006	0.033
HerbLayer	-0.196	-0.550	0.158
LitterDepth	0.162	-0.199	0.522
Year	-0.065	-0.303	0.174
Site	0.097	-0.456	0.649
T/C	-0.066	-0.664	0.533

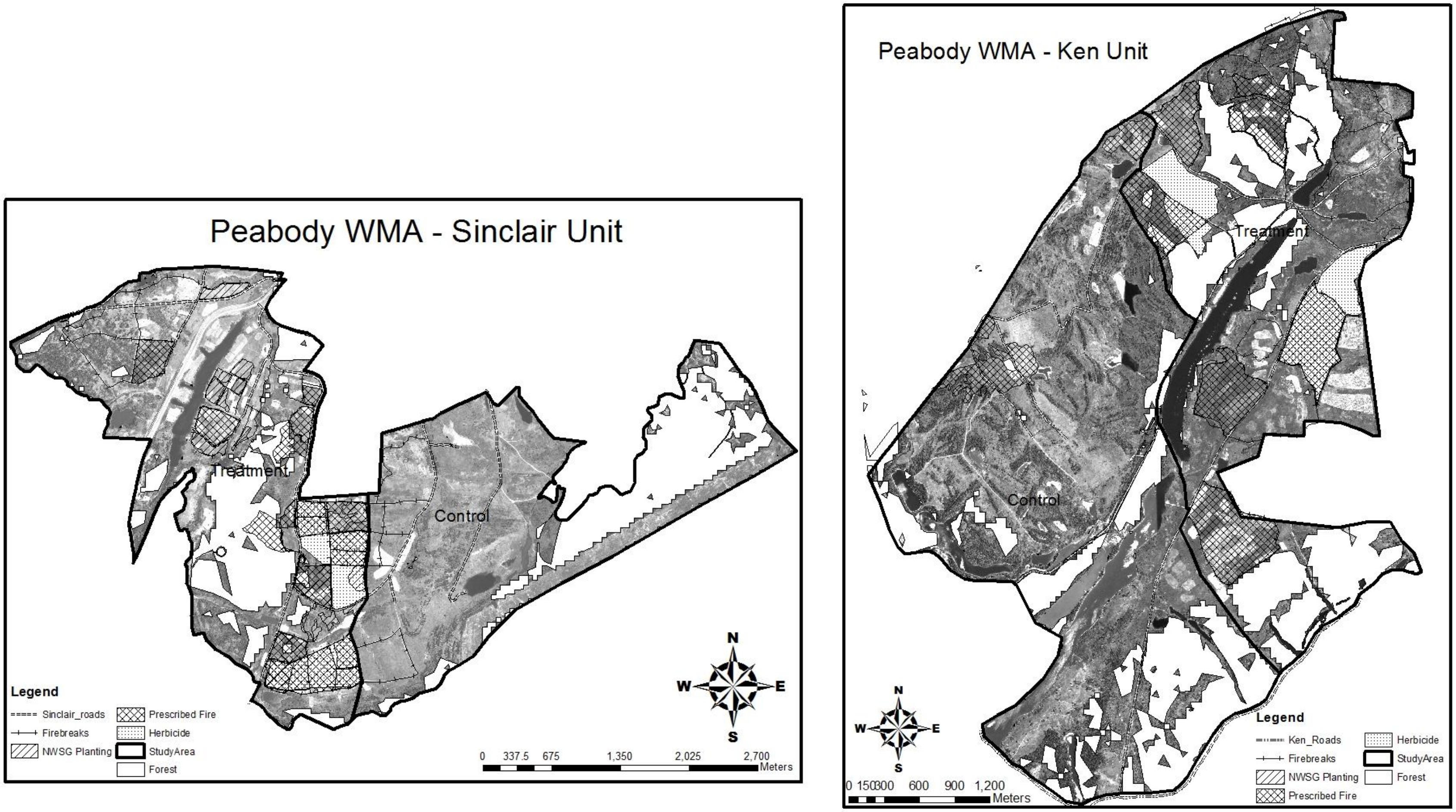

Figure B.1. Map of total treatments conducted on the Sinclair and Ken units on Peabody WMA, Ohio and Muhlenberg Counties, Kentucky, USA, 1 Oct 2009-30 Sep 2013.

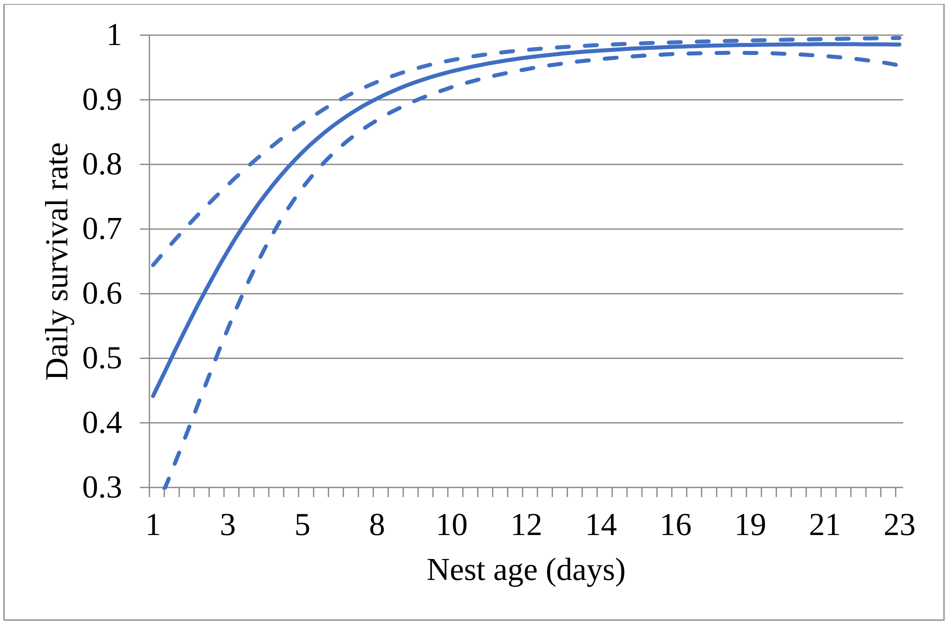

Figure B.2. Daily survival rates and confidence intervals (dotted lines) of northern bobwhite nests as a function of nest age on Peabody WMA, Ohio and Muhlenberg Counties, Kentucky, USA, 1 Apr 2010-30 Sep 2013.

Table B.8. Reproductive efforts of radiomarked male and female northern bobwhite surviving past 7 May on Peabody WMA, Ohio and Muhlenberg Counties, Kentucky, USA, 1 Apr 2010-30 Sep 2013.

	n		Nesting rate (%)[a]		Success rate (%)[b]		Renest rate (%)[c]	
Site	Male	Female	Male	Female	Male	Female	Male	Female
Sinclair	86	55	5.8	72.7	2.3	36.4	0.0	20.0
Ken	96	87	7.3	54.0	2.1	27.6	0.0	8.5
Pooled	182	142	6.6	61.3	2.2	31.0	0.0	13.8

[a] No. and % of radiomarked birds surviving past 7 May that attempted to incubate ≥1 nest.

[b] No. and % of radiomarked birds surviving past 7 May that successfully hatched ≥1 nest.

[c] No. and % of birds that failed on their initial attempt and initiated a second nest.

Table B.9. Mean clutch size for female incubated first nests, female incubated renests, and male incubated first nests of radiomarked northern bobwhite on Peabody WMA, Ohio and Muhlenberg Counties, Kentucky, USA, 1 Apr 2010-30 Sep 2013.

	Nest type								
	Female incubated first nests			Female incubated renests			Male incubated first nests		
Site	*n*	$\bar{x}$	SE	*n*	$\bar{x}$	SE	*n*	$\bar{x}$	SE
Sinclair	47	12.9	0.5	10	9.7	1.3	12	11.9	0.8
Ken	45	13.4	0.3	4	12.0	1.2	9	10.7	0.9
Pooled	92	13.1	0.3	14	10.4	1.0	21	11.4	0.6

Table B.10. Nest fates of radiomarked northern bobwhite on Peabody WMA, Ohio and Muhlenberg Counties, Kentucky, USA, 1 Apr 2010-30 Sep 2013.

	Site					
	Sinclair		Ken		Total	
Fate	*n*	%	*n*	%	*n*	%
Successful	32	46.4	29	50.0	61	48.0
Unsuccessful	37	53.6	29	50.0	66	52.0
Abandoned	11	29.7	8	27.6	19	28.8
Nest depredation	21	56.8	19	65.5	40	60.6
Mammalian	10		5		15	37.5
Snake	3		2		5	12.5
Unknown	8		12		20	50.0
Adult mortality	6	16.2	4	13.8	10	15.2
Mammalian	2		1		3	30.0
Avian	2		0		2	20.0
Trap related	0		1		1	10.0
Unknown	2		2		4	40.0
Total	69	100.0	58	100.0	127	100.0

Table B.11. Number and percentages of incubated and successful nests of northern bobwhite from first female nests, female renests, and first male nests on Peabody WMA, Ohio and Muhlenberg Counties, Kentucky, USA, 1 Apr 2010-30 Sep 2013.

		Nest Type					
		F-incubated first nest		F-incubated renest		M-incubated nest	
	Site	*n*	%	*n*	%	*n*	%
Incubated nests	Sinclair	47	68.1	10	14.5	12	17.4
	Ken	45	77.6	4	6.9	9	15.5
	Pooled	92	72.4	14	11.0	21	16.5
Successful nests	Sinclair	25	78.1	4	12.5	4	12.5
	Ken	23	79.3	2	6.9	3	10.3
	Pooled	48	78.7	6	9.8	7	11.5

Table B.12. Summary of landscape and microhabitat metrics by site used to evaluate nest survival of northern bobwhite on Peabody WMA, Ohio and Muhlenberg Counties, Kentucky, USA, 1 Apr 2010-30 Sep 2013.

		Sinclair		Ken	
Metric	Scale	$\bar{x}$	SE	$\bar{x}$	SE
FOR (%)	Landscape	1.9	0.6	7.4	2.2
SS (%)	Landscape	14.5	1.5	26.0	2.6
NWSG (%)	Landscape	11.3	1.6	23.3	2.9
OH (%)	Landscape	64.8	2.1	36.9	3.1
FOR core area (ha)	Landscape	0.5	0.2	1.5	0.4
SS core area (ha)	Landscape	2.2	0.2	3.5	0.3
NWSG core area (ha)	Landscape	1.5	0.2	3.1	0.4
OH core area (ha)	Landscape	9.0	0.4	4.7	0.4
Contagion index (%)	Landscape	51.1	2.4	44.5	2.2
FOR and open edge density (m/ha)	Landscape	0.9	0.3	1.1	0.3
SS and open edge density (m/ha)	Landscape	28.4	2.0	36.1	2.5
Distance to bare ground (m)	Microhabitat	56.5	9.7	39.2	5.2
Distance to edge (m)	Microhabitat	59.3	8.5	39.5	5.2
Litter depth (cm)	Microhabitat	2.4	0.2	2.0	0.2
Stem density (stems/ha)[a]	Microhabitat	8.8	2.8	4.7	1.5
Grass (%)[b]	Microhabitat	36.1	5.8	35.0	6.4
Herb layer[c]	Microhabitat	1.7	0.1	1.9	0.2
Ground layer[d]	Microhabitat	4.8	0.0	4.9	0.1

[a] Average stem density of species < 11.4 cm DBH within 5 m radius centered on nest location.

[b] Average % of grass species coverage within 1x1 m plot centered on nest location.

[c] Average coverage of vegetation on Nudds board (0.75-1.25 m) centered on nest location.

[d] Average coverage of vegetation on Nudds board (0-0.25 m) centered on nest location.

PART IV

DEMOGRAPHIC ANALYSIS OF NORTHERN BOBWHITE ON A RECLAIMED SURFACE MINE IN WESTERN KENTUCKY

ABSTRACT The northern bobwhite (*Colinus virginianus*) is an important game bird that is intensively managed for hunting recreation in the southeastern United States. Despite management efforts, northern bobwhite populations have declined across their range for much of the last 40 years. Many hypotheses exist to explain this decline; however, the most critical limiting factors affecting population growth rates (λ) remain unclear. To date, no study has assessed bobwhite vital rates on reclaimed surface mined land. To identify which demographic parameter(s) managers should focus on, we used life-stage simulation analyses (LSA) to examine sensitivity of λ to simulated variation in 9 demographic parameters for bobwhite on Peabody WMA, a 3,330-ha reclaimed surface mine in western Kentucky. We based model parameters on data collected from a 4-year, field study using a radiomarked population of wild bobwhites; in total, we radiomarked 1,131 bobwhite and located 127 nests. The mean value of λ after 1,000 simulations was 0.266 (inter-quartile range = 0.213-0.336). Total fecundity explained 94% of variation in λ when modelled across all vital rates. Of the 9 demographic parameters, clutch size explained most of the variation in λ (r^2 =0.384), followed by hatching success (r^2 = 0.207), and nest survival (r^2 = 0.141). Chick survival (r^2 = 0.030), renesting rate (r^2 = 0.070), double clutching rate (r^2 = 0.003), and male nesting rate (r^2 = 0.024) combined to explain 12.6% of the variation in λ. Both winter and summer survival explained <5% of variation in λ. Reproductive parameters appear to be the most responsible for ensuring successful population growth. No single vital rate was capable of achieving population stability (λ = 1) when solving univariate equations. Rather a combination of vital rates was needed to stabilize population growth. Management of northern bobwhite on reclaimed surface mines will require improved productivity.

The northern bobwhite (*Colinus virginianus*) (hereinafter, bobwhite) is an important game bird that has been declining for much of the last 40 years (Brennan 1991). In the Central Hardwoods region there has been a 4.0% decline annually and a 2.9% decline within Kentucky alone (Sauer et al. 2011). Much of the annual population variation shown by bobwhite can be explained by weather conditions during critical periods (i.e., nesting, brood-rearing, and winter; Guthery 1997, Guthery et al. 2000, Folk et al. 2007). Long-term population declines, however, tend to be tied more directly to habitat quality, quantity, and distribution (Brady et al. 1998). Given regional differences in life-history traits and environmental conditions within the bobwhite range, a single set of management recommendations cannot be expected to stabilize or increase bobwhite populations across the range.

Many studies have focused on survival, mortality, and fecundity of bobwhites, but few have examined the sensitivity of key vital rates on population growth and decline (Sandercock et al. 2008, Gates et al. 2012, Williams et al. 2012). Sandercock et al. (2008) used life-stage simulation analyses (LSA) to examine sensitivity of population growth rates (λ) to demographic parameters, which included investigating the relationship between fecundity, nesting attempts, and survival. They reported that overwinter survival explained the largest amount of variation in annual population growth rates for declining bobwhite populations across their range. Likewise, Gates et al. (2012), who conducted LSA using empirical data collected on 8 private lands sites in southwestern Ohio during 2008-2011, reported non-breeding season survival was the most limiting population vital rate. In their study, fecundity rates and chick survival explained 23.5% and 1.5%, respectively, of the variation in λ. Similarly, Williams et al. (2012) conducted LSA on data collected from a 3-year study of a population of bobwhite in New Jersey. They compared local demographic parameters of bobwhite to a national compilation of demographic parameters

(Sandercock et al. 2008) and found that based on the national model, both winter and summer survival made the greatest contribution to variance in λ. However, their local model suggested annual variation in components of fecundity had a large effect on variation in λ, including the number of young produced per nest that survived 30 days ($r^2 = 0.53$), nest success, ($r^2 = 0.20$), and clutch size ($r^2 = 0.18$) (Williams et al. 2012).

Previous studies have demonstrated that bobwhite populations in northern portions of the species' range are limited more by survival than reproductive rates compared to southern populations (Guthery et al. 2000, Folk et al. 2007). Guthery et al. (2000) found survival, production, and demographic capacity were the most management-sensitive variables in their prototype bobwhite population. Demographic capacity (saturation density) in autumn was considered to be the key variable for persistence and was dependent on seasonal weather conditions. In the southern part of the bobwhite range, high temperatures (Guthery et al. 2001, Forrester et al. 1998) and reduced precipitation (Hernández et al. 2005) have been shown to negatively affect productivity, whereas northern populations generally experience harsher winter conditions and associated mortality (Gates et al. 2012). Folk et al. (2007) reported variation in nonbreeding season survival in the subadult (HY) age class contributed the most to changes in population growth rate in northern populations, whereas changes in fertility in the subadult age class contributed the most to population growth rate in southern populations. These studies revealed important patterns in the temporal differences of bobwhite life history at the northern and southern extent of its geographic range.

A common theme among current and past research is that management of large tracts of contiguous habitat is needed to have an impact on reversing the bobwhite decline (Dimmick et al. 2002, Williams et al. 2004). Reclaimed surface mines offer a unique opportunity to create extensive habitat for bobwhite with over 627,000 ha reclaimed throughout the eastern United States and over 269,000 ha in Kentucky alone (Table C.1). Under the auspices of the Surface Mining Control and Reclamation Act of 1977 (SMCRA), early successional habitat has been established on reclaimed mine sites. However, the main focus has been to prevent erosion and this has led to the establishment of species such as sericea lespedeza (*Lespedeza cuneata*) and many other non-native species which provide marginal habitat for bobwhite (Jones et al. 1994, Barnes et al. 1995, Eddy 1999). Although these areas can support bobwhite populations (Bekerle 2004, Tanner 2012), it is unclear what vital rates managers need to focus on to try to increase bobwhite populations on reclaimed mine lands.

To gain a better understanding of the limiting vital rates of bobwhites on reclaimed surface mines, we conducted a LSA to examine sensitivity of population growth/decline to key demographic parameters. Our objectives were to: (1) estimate important vital rates, and (2) conduct LSA to determine the influence of vital rates on population growth rates. We did not pose a specific research hypothesis at the outset of this study. Rather, we sought to identify which vital rates were most important for bobwhite production on reclaimed mine lands.

STUDY AREA

We conducted our study on a reclaimed surface mine, Peabody Wildlife Management Area (PWMA; 3,324 ha), in Muhlenberg (37°14'N, 87°15'W) and Ohio (37°17'N, 86°54'W) counties in western Kentucky, USA. Land cover included four major vegetation types (open herbaceous, scrub shrub, forest, and native warm-season grasses) constituting 91% of the total land cover on our site. Shrub vegetation (25% of our site) was characterized by an abundance of black locust (*Robinia pseudoacacia*), winged sumac (*Rhus copallinum*), wild plum (*Prunus* *americana*), and blackberry (*Rubus spp.*). Open herbaceous vegetation (36%) was dominated by sericea lespedeza (*Lespedeza cuneata*) and annual forbs, such as common ragweed (*Ambrosia artemisiifolia*), sumpweed (*Iva annua*), and goldenrod (*Solidago spp.*). Deciduous forests (22%) primarily consisted of eastern cottonwood (*Populus deltoides*), green ash (*Fraxinus pennsylvaticum*) and American sycamore (*Platanus occidentalis*), and typically had a thick understory consisting of blackberry (*Rubus spp.*) and honeysuckle (*Lonicera japonica* and *Lonicera maakii*). From 2000 to 2004, native warm-season grasses (NWSG), including mixtures of big bluestem (*Andropogon gerardii*), little bluestem (*Schizachyrium scoparium*), indiangrass (*Sorghastrum nutans*), and switchgrass (*Panicum virgatum*), were established (8%). Small lakes, wetlands, and annual food plots comprised the remainder (9%) of our study area.

METHODS

Data Collection

Bobwhites were captured during all months of the year using funnel traps (Stoddard 1931, Palmer et al. 2002) baited with cracked corn and grain sorghum placed throughout the study area at known covey locations as well as locations that appeared to contain suitable cover for bobwhite. Traps were covered with burlap and surrounding vegetation to reduce stress and predation on captured individuals. Additionally, we used a second technique involving netting of radiomarked birds at night during the winter season to increase the number of radiomarked bobwhite in a covey (Truitt and Daily 2000). We recorded the sex, age, weight (g), and overall condition of each captured bobwhite (Rosene 1969). Sex was determined by plumage and age was determined by the presence or absence of buff-tipped primary coverts. Condition was rated on a scale of 1 to 6 with 1 being excellent condition and 6 being extremely lethargic or injured.

Birds were not collared if condition was >4. We fit birds that weighed >120 g with a necklace-style radio transmitter (crystal-controlled, two-stage design, pulsed by a CMOS multivibrator, American Wildlife Enterprise, Monticello, Florida, USA) that weighed ≤6.5 grams. Our trapping and handling methods complied with University of Tennessee Institutional Animal Care and Use Committee Permit (no. 2042-0911) protocol.

Radiomarked birds were located ≥3 times/week using a scanning receiver and a handheld Yagi antenna (Advanced Telemetry Systems, Inc., Isanti, MN). However, nesting birds were located daily to monitor nest success. We used the homing method (White and Garrot 1990) by walking to ≤50 m of the bird to minimize disturbance. Each bird was located at different times throughout the week to capture the variability in daily movements and habitat use. Upon detection of a mortality signal (12-hr signal), we immediately located the collar and determined the fate of the individuals as predation (mammal, avian), investigator-induced (consequence of research efforts), or unknown, based on evidence at the site of recovery and condition of the recovered transmitter (Curtis et al. 1988).

The breeding season was defined as 1 April-30 September (Burger et al. 1995*a*). We assumed birds located 2-3 times at the same site to be potential nesters and placed flagging >10 m away from the presumed nest location. Once the radiomarked bird was away from the nest, we located the nest, and recorded the number of eggs and UTM coordinates. Incubating birds were monitored once daily until hatch, nest failure, or adult mortality occurred (DeVos and Mueller 1993). We determined nest termination date to within 1 day and classified each nest as hatched, abandoned, or destroyed (Burger et al. 1995*b*). A hatched nest were those for which ≥1 egg hatched. Abandoned nests were those for which all eggs remained, but the incubating adult did not return. A destroyed nest was any nest in which ≥1 egg was destroyed and the incubating adult did not return. We assumed nests were found on day 1 of incubation to estimate nest initiation and nest age (Potter et al. 2011).

Data Analysis

We estimated seasonal survival rates using the known fate model with a logit link function in Program MARK (White and Burnham 1999). We censored the first 7 days after a bird was released in our analysis to control for a potential short-term effect of capturing and radio-marking (Guthery and Lusk 2004). We used a staggered-entry method to analyze survival with the known fate model (Pollock et al. 1989). This method left-censors individual's encounter histories until they are captured and enter the monitored population. We right-censored individuals because of emigration from the study area, radio failure or loss, or unknown fate. We assumed birds were randomly sampled, survival times for individuals were independent, and censoring mechanisms were independent of animal fate. Each survival period (winter and summer) consisted of 183 days. Encounter histories were coded as weekly survival periods, where each survival period had 26 encounter history periods. We used model averaging to obtain seasonal survival estimates; these estimates and standard errors take into account model selection uncertainty (White 2008).

We used the nest survival model within Program MARK (White and Burnham 1999) to estimate daily survival rate of nests across the 4 years of our study. Nest survival was defined as the probability of a nest surviving the 23-day incubation period (Rosene 1969, Potter et al. 2011). The delta method (Powell 2007) was used to expand our survival estimates to a temporal scale that encompassed the 23-day incubation period.

We also estimated total clutch size, hatching success, male nesting rate, renesting rate, and double clutching rate for fecundity vital rates (Table C.2). Male nesting rate, renesting rate, and double clutching rate was based on the number of birds documented to have nested during the nesting season. We estimated male nesting rate as the proportion of male-incubated nests per female-incubated nests. Renesting rate was estimated as the proportion of radiomarked females that initiated new clutches after failed nesting attempts. Double clutching rate was estimated as the proportion of radiomarked females that initiated a second nest after a successful nest. After the radiomarked bird and brood had vacated the nest site, nests were examined to determine hatching success (DeMaso et al. 1997). Hatching success was calculated as the proportion of eggs that hatched from successful nesting attempts. Clutch size was the average number of eggs laid per nesting attempt.

We conducted LSA of population growth and fecundity following Sandercock et al. (2008) using 9 demographic parameters (Table C.2). We were able to use empirical estimates (mean ± SD) collected during our study for all parameters except chick survival due to the inability to adequately monitor broods post-hatch. Therefore, we used Suchy and Munkel (2000) estimates of chick survival after adjusting their 38-day period to a 30-day pre-fledging period (Gates et al. 2012). Chick survival was defined as the proportion of chicks hatched that survived until independence at 30 days. We sampled this vital rate from a uniform distribution bounded by 95% confidence intervals after adjusting the upper and lower limits for a 30-day period (range 0.740-0.943) (Gates et al. 2012). We specified mean and standard deviation for each of the other eight parameters and sampled from normal distributions with 1,000 iterations (Figure C.2); the same number of iterations was used for chick survival. Simulations were conducted with R (R Development Core Team 2012). We treated vital rates as independent and did not use a covariance structure or a function with density-dependence to select random draws (Wisdom et al. 2000). We used linear regression and coefficients of determination (r^2) to calculate the amount of variation in finite rate of population growth (λ) explained by simulated variation in each of the 9 demographic parameters. We solved univariate equations from regressions of vital rates on simulated population growth rates for $\lambda = 1$ to identify the magnitude of change necessary to achieve stable populations.

We also used linear regression to calculate the amount of variation in λ explained by total fecundity, which was based on the sum of 6 types of nesting attempts and their contribution during the 6-month summer breeding season (Sandercock et al. 2008). Nesting attempts included (1) first nests incubated by females, (2) second nests laid after successful hatching of a first clutch and incubated by females, (3) renests laid after loss of a first nest and incubated by females, (4) second renests laid after loss of first renests and incubated by females, (5) first nests incubated by males, and (6) renests laid after loss of a first male clutch and incubated by males (Sandercock et al. 2008).

RESULTS

Vital Rates

We captured 1,794 bobwhites (866 males, 652 females, and 276 birds for which we were not able to confirm gender) from 1 October 2009 – 30 September 2013. We captured more juveniles (n = 1,443) than adults (n = 351) during the course of our study. Of the 1,794 captured birds, we radiomarked 1,198, but were only able to use 1,131 in our survival analyses because of censoring. Winter survival estimates were 0.224 (n = 151, SE = 0.039) in 2010, 0.125 (n = 229, SE = 0.022) in 2011, 0.243 (n = 102, SE = 0.050) in 2012, and 0.469 (n = 214, SE = 0.042) in 2013. Summer survival estimates were 0.176 (n = 127, SE = 0.040) in 2010, 0.151 (n = 108, SE = 0.039) in 2011, 0.171 (n = 274, SE = 0.026) in 2012, and 0.115 (n = 274, SE = 0.022) in 2013.

The model-averaged, pooled winter survival estimate across all 4 years was 0.279 (n = 696, SE = 0.021) and 0.149 (n = 783, SE = 0.015) during the summer (Table C.2).

A total of 127 nests (30 in 2010, 27 in 2011, 32 in 2012, and 38 in 2013) were found, of which 106 were incubated by females and 21 by males. However, we were only able to use 124 of the nests for nest survival analysis. To avoid disturbance and probability of nest abandonment, the remaining 3 nests, which were located without a radiomarked adult associated with the nest, were not monitored. Overall daily nest survival was 0.956 (SE = 0.009) and 0.352 (SE = 0.062) during the 23-day incubation period (Table C.2). Overall, 48.0% of nests were successful. Clutch size (n = 127) ranged from 2 to 19 and averaged 12.5 eggs (SD = 3.2) (Table C.2). Pooled hatching success was 84.3 ± 2.9% (n = 61, range 8.3-100%; Table C.2). Males incubated 16.5% (n = 21) of nests and no males were observed renesting. We recorded 13.8% (n = 12) of females initiating a new nest after an unsuccessful first nest. There were only 2 (4.5%) instances of females initiating a second nest after completing a successful clutch (Table C.2) during the four nesting seasons.

Life Stage Analyses

The median value of λ after 1,000 simulations was 0.266 (inter-quartile range = 0.213-0.336) implying a declining population. Clutch size was the most dominant vital rate affecting population growth rate, followed by hatching success and nest survival (Table C.3; Figure C.3). Chick survival, renesting rate, double clutching rate, and male nesting rate combined to explain 12.6% of the variation in λ. Both winter and summer survival explained <5% of variation in λ.

No single vital rate was capable of achieving population stability ($\lambda = 1$) when solving univariate equations relating vital rates to λ (Table C.3) except raising summer survival from

14.9% (observed) to 99.4%, which would be biologically infeasible. A 1-egg change in clutch size would produce a 1.9% change in λ and would require a clutch size of 50 to produce λ = 1, which was also biologically impossible. A 1% change in winter and summer survival would produce a change in λ of 0.8% and would require a winter survival rate of 1.21 or summer survival rate of 0.99 to attain population stability, which are both scientifically impossible. Likewise, a 1% change in nest survival would produce a change in λ of 0.6%, and raising nest survival to 1.0 would only raise λ to 0.662.

Total fecundity explained 94% of variation in λ when modeled across all vital rates. Clutch size, followed by hatching success and nest survival had the greatest influence on variation in total fecundity (Table C.4; Figure C.4). Male nesting rate and renesting rate combined to explain 9.6% of variation in total fecundity. Double clutching rate was negligible, explaining <1% of variation in total fecundity.

DISCUSSION

Our LSA revealed that no single vital rate was capable of achieving population stability (λ = 1), when solving univariate equations, suggesting that our demographic parameters were either biased low or depressed below sustainable levels. Clutch size and hatching success, the two most important vital rates in our study, were likely unbiased because they were measured directly from nest inspection (Sandercock et al. 2008). However, other parameters related to fecundity (male nesting, renesting, and double clutching) may have been biased low. Renesting, double clutching, and male incubation have been shown to enhance nesting productivity (Burger et al. 1995*b*); however, there is little evidence these vital rates affect population growth rate compared to seasonal survival (Sandercock et al. 2008, Gates et al. 2012). Stoddard (1931) stated that although bobwhite experienced low individual nest success, the majority of pairs were eventually successful through renesting. This was not the case during our study as no males were documented renesting and only 13.8% of females (n = 12) were known to renest after a failed nesting attempt. Male bobwhite incubated 21 nests (16.5%), which was once again at the lower end of the range of published estimates (13-29%; Stoddard 1931, Klimstra and Roseberry 1975, Suchy and Munkel 1993, Burger et al. 1995*b*). However, our observed male nesting rates were likely biased low because of low sample sizes, and may not have been comparable to male nesting rate estimates from previous studies with larger sample sizes (Burger et al. 1995*b*). Double clutching can have a positive influence on annual production, though it is not necessary to replace populations under normal conditions (Roseberry and Klimstra 1984). There were only two instances of double-clutching that occurred during our study, which was likely an underestimate.

Secondary nesting activities (renesting, double clutching, and male nesting) from our study combined to account for 9.7% of the variation in population growth rate, indicating only a modest contribution to reproductive output. However, Burger et al. (1995*b*) suggested that studies that observe only female reproductive activity underestimate reproductive effort and production by approximately 33%. Late-season first attempts may also be underestimating the number of clutches a female has actually attempted. These late-season nests may actually be renests, second attempts, or even third attempts (Burger et al. 1995*b*). This may possibly underestimate reproductive effort and production by the end of the nesting season.

In opposition to most demographic studies on bobwhite (Sandercock et al. 2008, Gates et al. 2012, Williams et al. 2012), we did not detect much sensitivity of winter survival on population growth rate. Also, our estimates of winter survival were somewhat higher than past estimates (Curtis et al. 1988, Burger et al. 1995*a*, Lohr et al. 2011). This may have been a result of bobwhite inhabiting more forested areas during winter (Tanner 2012) as opposed to more open herbaceous areas associated with nesting activities during the summer months. This is in contrast to the findings by Seckinger et al. (2008) who reported an increase in winter survival after removal of closed-canopy forest vegetation. However, forest vegetation on PWMA was established during reclamation and was very uncharacteristic of typical forested areas within the Mid-South. Forest vegetation rarely had canopy closure and as a result, was more similar to open canopy woodlands. The understory was comprised primarily of blackberry and honeysuckle, which provided ideal escape cover. In Virginia, Tonkovich and Stauffer (1993) reported quail tended to use sites with more Japanese honeysuckle than was generally available. Likewise, Roseberry and Klimstra (1984) noted that Japanese honeysuckle was frequently an understory component of woody headquarters of coveys in Illinois. Additionally, our estimates of summer survival were exceptionally low compared to past research (Burger et al. 1995*a*, Sisson et al. 2009, Lohr et al. 2011), which may have been due to a combination of the cost of reproduction and poor habitat conditions during the summer.

Our LSA demonstrated that population growth rate was mostly affected by fecundity parameters. Specifically, clutch size, hatching success, and nest survival were major factors explaining variation in λ. A high contribution of fecundity is consistent with Wisdom and Mills (1997) and Wisdom et al. (2000) who found similar impacts of fecundity for Greater Prairie-Chicken (*Tympanuchus cupido*) population growth. In contrast to our results, LSA studies in Ohio (Gates et al. 2012), New Jersey (Williams et al. 2012), and a comprehensive study of bobwhite data across their range (Sandercock et al. 2008) found winter survival was the most important vital rate. Guthery (1997) and Guthery et al. (2000) suggested northern populations of bobwhites tend to have higher fecundity and lower annual survival compared to southern populations. Although the LSA conducted by Williams et al. (2012) in New Jersey found winter survival to be the most important vital rate contributing to variance in λ, components of fecundity (clutch size, nest success, young produced per nest) also had a large effect on variation in λ. These results suggest elements of fecundity may be more critical on population growth in more marginal habitat, including New Jersey at the northern periphery of the bobwhites' range as well as reclaimed surface mines.

The importance of the contributions fecundity may make to bobwhite population dynamics also lies in their fundamental biology as an r-selected species (MacArthur and Wilson 1967). Elasticity patterns of birds (Saether and Bakke 2000) and mammals (Heppell et al. 2000) suggest that population growth rates of r-selected species will respond to higher fecundity and rapid development of young, while growth rates of K-selected species respond better to improved adult or juvenile survival rates. Saether and Bakke (2000) used elasticity analysis to analyze published data on 49 species of birds, including 27 species from the order Cicconiformes, 16 species from the order Passeriformes, 2 species from each of the orders Anseriformes and Strigiformes, and 1 species each from the orders Coraciiformes and Piciformes, to determine how λ is influenced by variation in different demographic traits. Across species, they found that the contribution of fecundity to population growth rate increased with increasing clutch size and decreasing adult survival, while the greatest contribution of adult survival occurred among long-lived species that matured late and laid few eggs. In populations with relatively low adult survival, such as bobwhite, the elasticity of λ to changes in fecundity or juvenile survival may dictate that they are key vital rates (Heppell 1998). Indeed, bobwhites exhibit a reproductive system that responds to and enables recovery from high annual mortality and catastrophic events (Stanford 1972, Roseberry and Klimstra 1984, Suchy and Munkel 2000).

Furthermore, Williams (1966) observed that the central biological problem is not survival, but design for survival; the set of adapted reproductive traits sensitive to particular ecological problems (Stearns 1976). Of those adaptive traits, clutch size is thought to be a good measure of hen fitness (Eldrigde and Krapu 1988). Our clutch size (12.5 ± 3.2) was similar to that of previous studies (range 11.5-14.0 eggs; Sandercock et al. 2008). However, our rates were somewhat lower than the 14.4 and 13.7 reported by Stoddard (1931) and Roseberry and Klimstra (1984), respectively. Similarly, our hatching success (85.5 ± 2.9%), which was lower than reported by most studies (85-95%; Sandercock et al. 2008), may have also been related to hen condition. Furthermore, the average clutch size of early nests (13.47; May and June) was higher than late nests (11.40; July, August, and September) suggesting the importance of body condition as well as quality nesting conditions early in the nesting season.

Caloric deficiencies can lead to delays in egg laying, reduced egg production, and ovarian degeneration in female fowl (Breitenbach et al. 1963, King 1973). Eldrigde and Krapu (1988) reported variation in diet quality of captive Mallards (*Anas platyrynchos*) significantly affected clutch size, egg mass, egg composition, laying rate, renesting interval, nesting attempts, and total eggs laid. Similarly, Erikstad et al. (1993) found a positive correlation between body mass and clutch size in Common Eider (*Somateria mollissima*). They postulated that a high body mass during incubation may increase nest attentiveness and decrease the incubation period, as well as reduce predation risk (Erikstad et al. 1993). Giuliano et al. (1996) found that low protein and/or energy in the diet negatively affected egg production and ovary mass in both northern bobwhite and scaled quail.

Our study site was comprised primarily of sericea lespedeza as a result of reclamation efforts aimed at quickly reducing erosion. Although this structural arrangement can provide acceptable cover for bobwhite, sericea lespedeza seeds are virtually indigestible and provide no nutritional benefit (Davison 1958) compared to many native species (Blocksome 2006). Bugg and Dutcher (1989) found that invertebrate use of sericea lespedeza was lowest of all plant species they studied. Similarly, Menhinick (1967) reported that insect biomass was less on sericea lespedeza than on other vegetation. Eddy (1999) concluded that invertebrate and vertebrate species declined by 73 and 55% respectively in sericea lespedeza sites. High stem density, lack of singing perches, fewer canopy openings and a decrease in the seasonal availability of foliage, flowers, seeds, and prey insects were given as factors in the decline.

During the nesting and chick-rearing season, hens require a large proportion of invertebrate protein in their diet (Wiens and Rotenberry 1979). Nestler et al. (1944) and Nestler (1949) concluded that the physiological requirements of female quail during the nesting season demand a much greater intake of high protein foods than males. Likewise, bobwhite chicks require a high protein diet during the first two weeks of life, with invertebrate species comprising 94.1% of all foods eaten (Eubanks and Dimmick 1974). Insects have more than four times the protein as compared to plants and contain essential amino acids not present in plant protein. Insect protein is also more easily assimilated compared to plant protein (Moreby 2003).

Previous studies suggest that plants classified as forbs tend to be associated with higher invertebrate populations than grass, bare ground and shrubs (Burger et al. 1993, Jamison et al. 2002). Sericea lespedeza litter accumulation has been associated with reduced forb establishment and species richness (Foster and Gross 1998) and may actively interfere with germination of other species through allelopathic compounds generated in leaf and stem tissue (Adams et al. 1973, Wade 1989). With this is mind, more desirable forb species may have been lacking on our site because of the dominance of sericea lespedeza; thus, reducing invertebrate species present and leading to lower hen condition as well as sub-optimal foraging conditions for newly hatched chicks.

Low occurrences of desirable food plants on severely disturbed, reclaimed mine sites may offer new challenges to managers. We believe the site's plant community should be evaluated when assessing vital rates of bobwhites on severely-disturbed lands. Evaluation of food items used on reclaimed lands may yield lower than expected native food use rates, because of low native food plant colonization rates. Food item availability and/or occurrence within the habitat area should be ascertained when making any inferences about vital rates related to fecundity on reclaimed habitats which are dominated by perennial, agronomic plant species (Jones et al. 1994).

The right management prescription for increasing bobwhite survival may vary according to the particular landscape and region of interest. We conducted our study on a reclaimed surface mine, where conditions were considered sub-optimal based on dominance of non-native vegetation, limited woody cover, low survival rates, and low nesting success. For that reason, a thorough demographic analysis has been needed to allow managers to identify the primary vital rates affecting bobwhite population growth. With the extensive land area being reclaimed under SMCRA, surface mines could provide a great opportunity to provide habitat for a species in long-term decline. Furthermore, as a non-migratory species, a single large landscape such as our study area, could be manipulated to provide all habitat needs for a sustainable population, one that may be able to contribute to more isolated patches on surrounding areas (i.e., function as a source population).

MANAGEMENT IMPLICATIONS

As modern landscapes become increasingly fragmented and bobwhite populations continue to decline, an understanding of the basic mechanisms that determine population growth rates will be increasingly important. Management on reclaimed surface mines should focus on removing non-native monocultures, such as sericea lespedeza. Specifically, management (disking, burning, and herbicide application) should be conducted on a short return interval (e.g., 2 years) to combat the aggressive regrowth of sericea lespedeza. In addition, seeding native forbs and grasses would circumvent the problem and yield much more desirable results. Optimally, initial reclamation efforts on mine sites should incorporate seeding of native plants over non-native species. A more diverse mixture of forbs and native plant species would allow better feeding opportunities for both adults and chicks during the breeding season and, thus lead to greater reproductive effort. Consequently, if summer survival is enhanced concurrent to increases in nesting success and brood survival, the largest positive effects on population growth may be realized.

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APPENDIX C

Table C.1. Eastern US coal-mined land area (ha) reclaimed under SMCRA, 1978-2005[a].

State	Phase III Released	Phase I Released (2001-2005)[b]	Total
E KY[c]	243,533	26,094	269,627
MD	5,372	118	5,490
OH	74,167	9,495	83,662
PA	93,670	13,359	107,029
TN	14,962	2,946	17,908
VA	37,076	1,125	38,201
WV	93,685	11,673	105,358
Total	562,465	64,810	627,275

[a] Including the interim SMCRA program. Source US OSMRE "20th Anniversary of the Surface Mining Law" (http://www.osmre.gov/annivrep.htm) and annual reports to Congress.

[b] As reported by states to OSMRE; these figures overestimate total affected areas due to double-counting of areas that were both mined and re-mined under SMCRA.

[c] Estimated from total Kentucky areas, as proportionate to the east-west distribution of surface coal tonnage.

Table C.2. Vital rates used in life-stage simulation analyses of northern bobwhite population growth rates (λ) on Peabody WMA, Ohio and Muhlenberg Counties, Kentucky, USA, 1 Oct 2009-30 Sep 2013.

Parameter	Description	Estimate ± SD
Winter survival	Survival from 1 Oct - 31 Mar	0.279 ± 0.021
Summer survival	Survival from 1 April - 30 Sept	0.149 ± 0.015
Nest survival	Probability of a nest surviving 23-day incubation period	0.352 ± 0.062
Chick survival	Survival of chicks during 30-day pre-fledging period	0.842[a]
Clutch size	Average incubated clutch size	12.5 ± 3.2
Hatching success	Proportion of eggs that hatched from successful clutches	0.855 ± 0.171
Male nesting rate	Proportion of male-incubated nests per female-incubated nests	0.228 ± 0.070
Renesting rate	Proportion of radiomarked females that initiated new clutches after failed nesting attempts	0.138 ± 0.089
Double clutching rate	Proportion of radiomarked females that initiated a second nest after a successful nest	0.045 ± 0.038

[a] Suchy and Munkel (2000) estimate sampled from a uniform distribution.

Table C.3. Univariate regression model results for nine northern bobwhite vital rates (independent variable) and estimates of population growth rates (λ; dependent variable) calculated from life-stage simulation analyses following Sandercock et al. (2008) and Gates et al. (2012). Vital rates were randomly selected (n = 1,000) from normal distributions with mean and standard deviations based on data collected on northern bobwhites on Peabody WMA, Ohio and Muhlenberg Counties, Kentucky, USA, 1 Oct 2009-30 Sep 2013.

Parameter	Intercept	Regression coefficient (*b*)	SE (b)	r^2
Winter survival	0.064	0.775	0.137	0.030
Summer survival	0.156	0.849	0.200	0.017
Nest survival	0.076	0.586	0.046	0.141
Chick survival	0.031	0.299	0.053	0.030
Clutch size	0.044	0.019	0.001	0.384
Hatching success	0.022	0.313	0.019	0.207
Male nesting rate	0.233	0.217	0.043	0.024
Renesting rate	0.241	0.295	0.034	0.070
Double clutching rate	0.274	0.174	0.091	0.003

Table C.4. Univariate regression model results for northern bobwhite nesting productivity vital rates (independent variable) and estimates of fecundity rates (dependent variable) calculated from life-stage simulation analyses following Sandercock et al. (2008) and Gates et al. (2012). Vital rates were randomly selected (n = 1,000) from normal distributions with mean and standard deviations based on data collected on northern bobwhites on Peabody WMA, Ohio and Muhlenberg Counties, Kentucky, USA, 1 Oct 2009-30 Sep 2013.

Parameter	Intercept	Regression coefficient (*b*)	SE (b)	r^2
Nest survival	0.113	2.120	0.158	0.153
Clutch size	-0.005	0.069	0.003	0.417
Hatching success	-0.074	1.119	0.067	0.220
Male nesting rate	0.682	0.772	0.149	0.025
Renesting rate	0.713	1.035	0.118	0.071
Double clutching rate	0.827	0.634	0.318	0.003

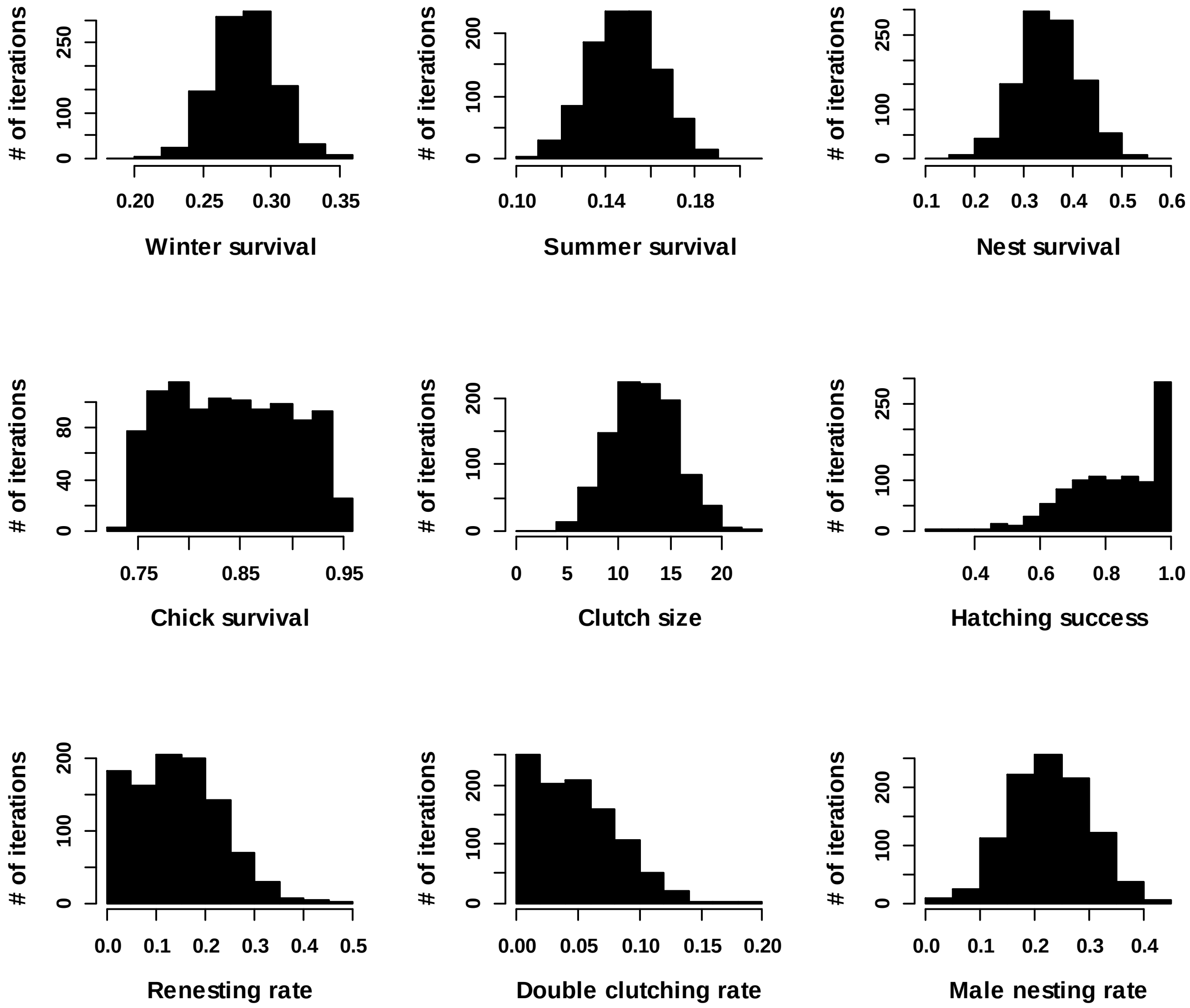

Figure C.1. Frequency and distribution based on 1000 simulations for values of nine vital rates selected from normal and uniform (chick survival only) distributions for life stage simulations of population growth rates (λ) of northern bobwhites on Peabody WMA, Ohio and Muhlenberg Counties, Kentucky, USA, 1 Oct 2009-30 Sep 2013.

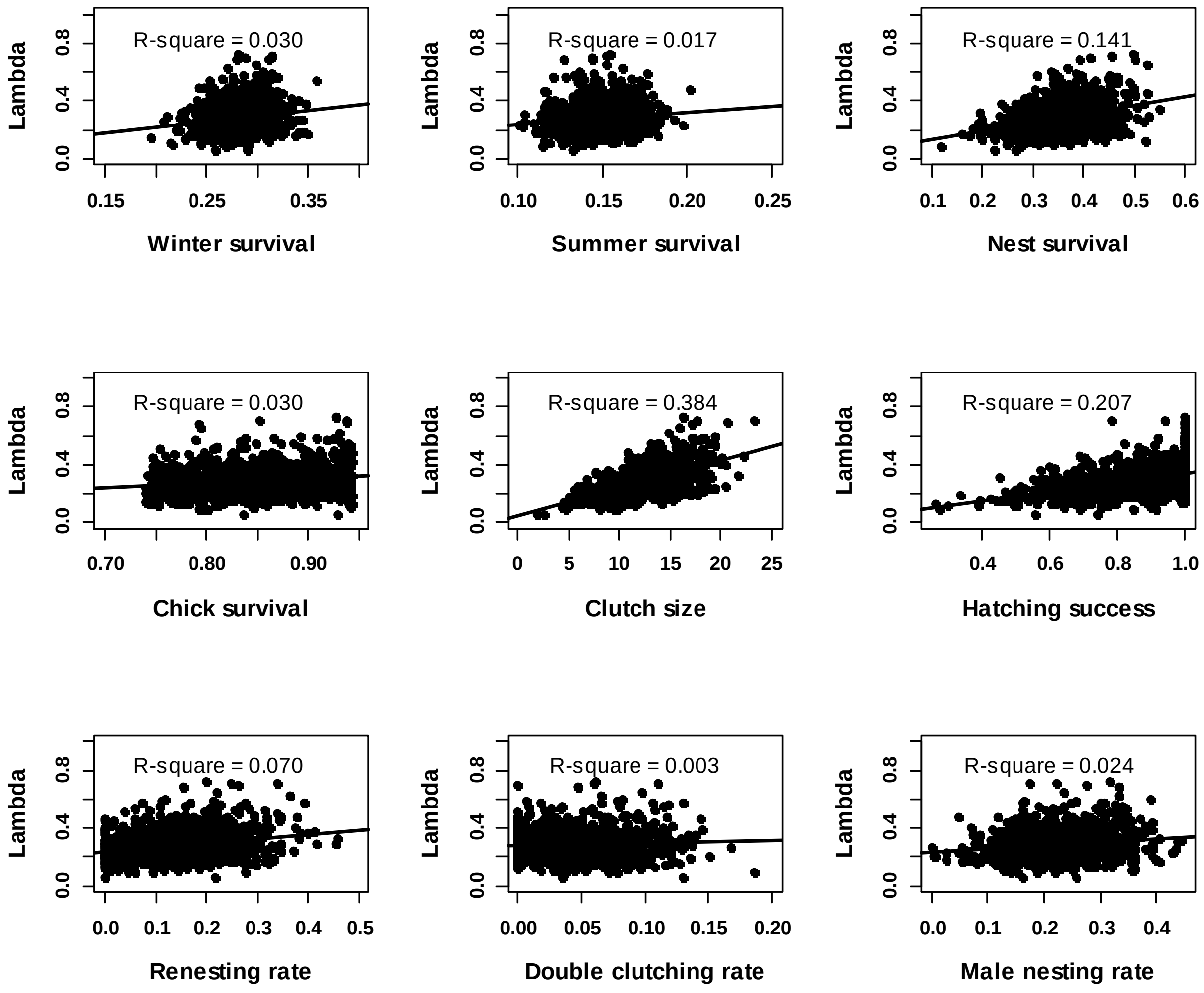

Figure C.2. Plots of simulated growth rates (λ; dependent variable) versus each of nine vital rates (independent variable) and associated linear regression models used for life stage analyses of northern bobwhites on Peabody WMA, Ohio and Muhlenberg Counties, Kentucky, USA, 1 Oct 2009-30 Sep 2013.

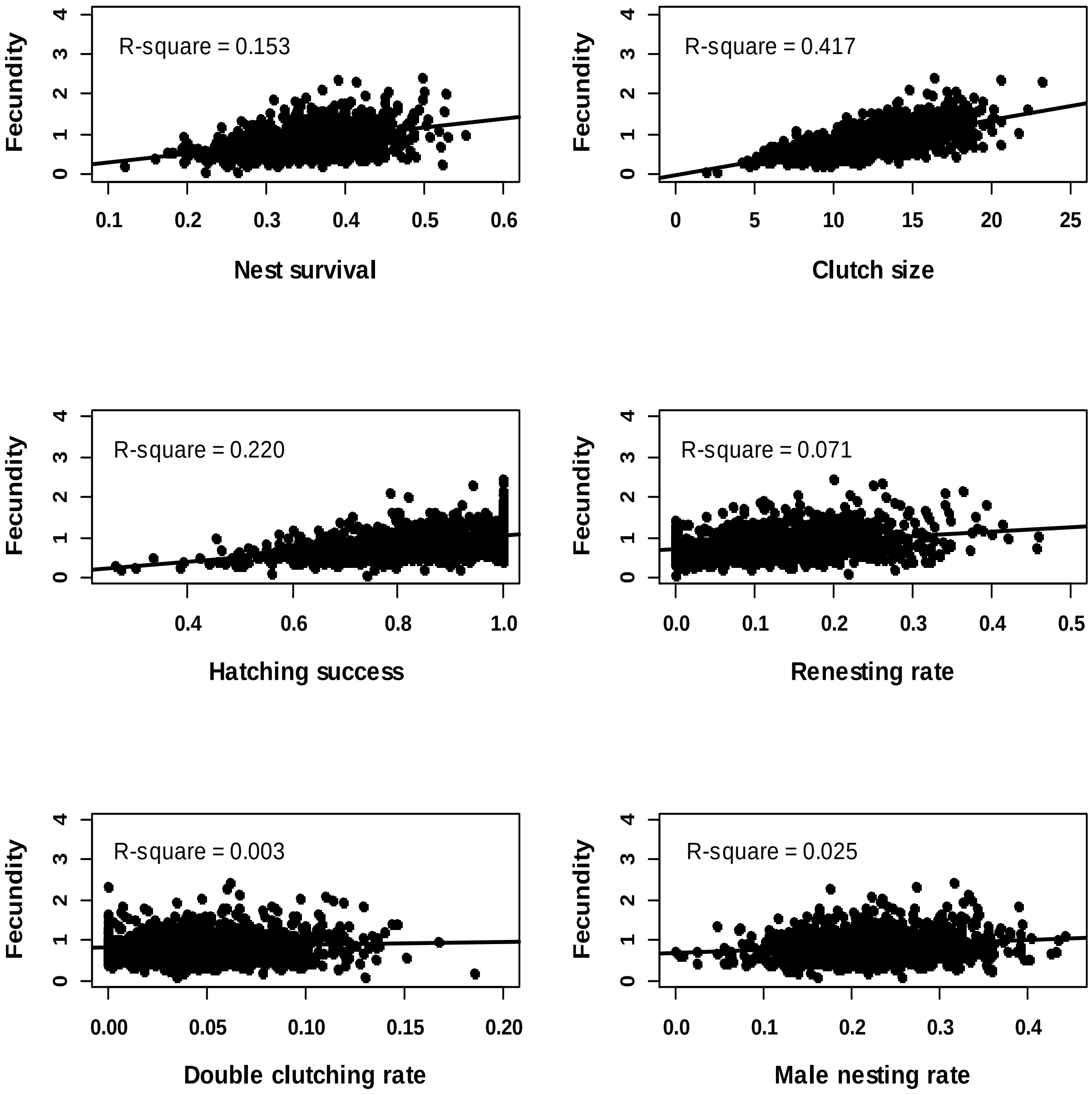

Figure C.3. Plots of simulated total fecundity (dependent variable) versus each of six reproductive vital rates (independent variable) and associated linear regression models used for life stage analyses of northern bobwhites on Peabody WMA, Ohio and Muhlenberg Counties, Kentucky, USA, 1 Oct 2009-30 Sep 2013.

APPENDIX D

METHODS

Data Collection

We estimated fall population densities using a fall covey survey. We counted the number of coveys giving the “koi-lee” call early in the morning before leaving roost locations (Stoddard 1931, Stokes 1967). We systematically placed survey points throughout the study area to maximize coverage and efficiency. Studies in the past have used a range of audibility radii, from >900 m (Rusk et al. 2009) to as little as 400 m (Roseberry 1982); therefore, we selected a 500-m radius, a conservative figure that is well within this published range. We placed survey points (n = 20) at least 1000 m apart to avoid any potential overlap (Rusk et al. 2007), which provided 47% coverage across the entire study area. Survey points were located at ridge tops along roads to facilitate access while allowing maximum probability of detection. We conducted the survey 45 minutes before sunrise and ceased monitoring 20 minutes beyond the last call recorded (DeMaso et al. 1992). Surveys were not conducted during extreme weather conditions or rain (Kozicky et al. 1956, Wellendorf et al. 2004). Because individuals can separate at night, there is a chance one covey may be recorded as multiple coveys. To avoid double counting, we considered covey calls from the immediate vicinity (<30 m) of another call as one covey only (Wellendorf et al. 2004). Once a covey call was heard, we took an azimuth using a hand-held compass and estimated the distance from the point ocularly. We measured call intensity and the number of covey calls/call events. Call events were defined as calls from a covey separated by >1 minute (Wellendorf et al. 2004). To minimize observer effects, all participants were exposed to and were able to identify the covey call (“*koi-lee”*) prior to data collection. We visited each survey point twice per fall. Prior to each fall survey we located collared coveys with telemetry equipment to compare calling rates of coveys, thus providing a correction factor for calling rate

(Riddle et al. 2008). We estimated average covey size by flushing coveys detected during the census with telemetry within 12 hours of completion of the census.

Data Analysis

We calculated fall population size by multiplying the average covey size by the total number of coveys heard on each site and dividing that number by the estimated calling rate multiplied by the percentage of area that was surveyed across the property (Holt et al. 2009). Population variance was calculated using the delta method (Powell 2007) by combining the coefficient of variation across all variables used (mean detection probability, mean covey size, mean covey count, and mean call rate) to generate an overall coefficient of variation. From that we were able to calculate an upper and lower confidence interval for population size for each year based on our sample area (S. D. Wellendorf and T. M. Terhune, Tall Timbers Research Station, personal communication).

RESULTS

Population Estimates

Population estimates increased by 15.0% from 2009 to 2010 (Figure D.1). The average covey size was 7.87 (SE = 0.75) during the fall of 2009. We did not estimate a calling rate during the 2009 fall covey survey, so the estimated calling rate from 2010 was used for both years. The estimated calling rate was 0.33 (SE = 0.08), and the average covey size was 8.91 (SE = 0.70) in 2010. Population estimates remained relatively stable from 2010 to 2011 (Figure D.1). The average covey size was 8.95 (SE = 0.95), and the estimated calling rate was 0.46 (SE = 0.07) in 2011. Population estimates increased by 8.4% from 2011 to 2012 (Figure D.1). The average covey size was 7.29 (SE = 0.59) and the estimated calling rate was 0.47 (SE = 0.07) in 2012. There was an increase of 55.7% in fall population size from 2012 to 2013 (Figure D.1). The average covey size was 8.57 (SE = 0.76), and the estimated calling rate was 0.34 (SE = 0.07) in 2013.

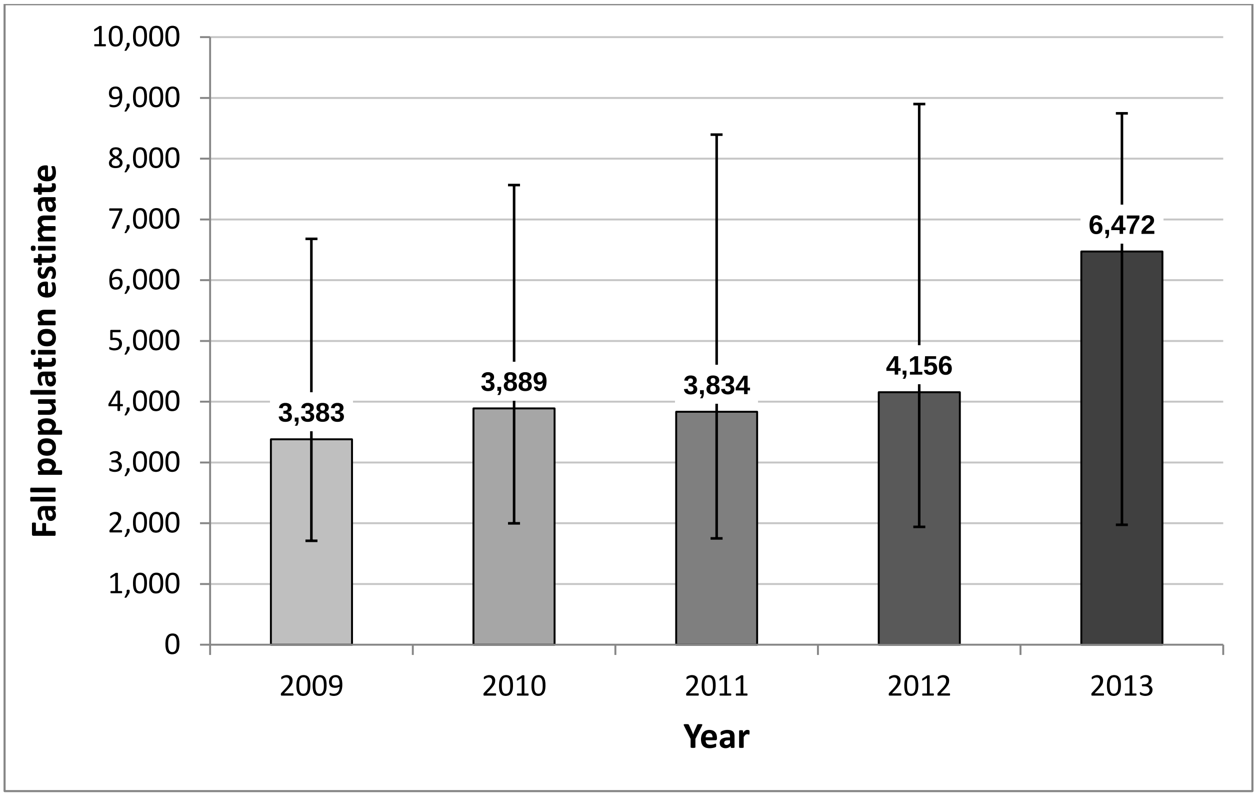

Figure D.1. Fall population estimates and confidence intervals of northern bobwhite on Peabody WMA, Ohio and Muhlenberg Counties, Kentucky, USA, 1 Oct 2009-30 Sep 2013.

PART V

CONCLUSIONS

The main objectives of my research were to determine the effects of habitat management on (1) seasonal (Part II) and (2) nest survival (Part III), how (3) multi-scale habitat contributed to seasonal (Part II) and nest survival (Part III), and (4) conduct life stage simulation analyses (LSA) to determine which vital rates were affecting population growth rate the most (Part IV) on a reclaimed surface mine (PWMA) in Western Kentucky. My key conclusions from the analyses are briefly discussed below.

In our experimental approach, we altered the landscape structure on PWMA by applying treatments (prescribed fire, disking, and herbicide application) to approximately 60% of treatment areas (78% of non-forested portions of treatment areas). Management efforts were focused on improving vegetation composition while trying to reduce the dominance of invasive species, such as sericea lespedeza. Subsequently, our results suggest that applied treatments (T/C) had an effect on increasing survival (Part II). Specifically, higher survival was associated with treatment during the summer. However, survival was lower on treatment during the winter compared to control, suggesting management activities may have negatively affected winter habitat conditions for bobwhite. In addition, we suspected that summer survival would be greater than winter survival due to the fact that most management was focused on improving summer habitat conditions for bobwhite. However, this was not the case as survival was higher during the winter (0.281) than summer (0.148), which is in opposition to most reported studies on bobwhite (Burger et al. 1995, Lohr et al. 2011).

Multi-scale habitat effects have been shown to be important to bobwhite survival in previous research (Seckinger et al. 2008, Janke 2011). Likewise, we detected evidence of multi-scale influences of habitat on seasonal survival at the microhabitat scale. Litter depth had a negative influence on survival. We attributed these findings to the importance of bare ground (Stoddard 1931, Brown and Samuel 1978, Klimstra and Zicarrdi 1963, Ellis et al. 1969).

Our data suggest that nest success (0.352 ± 0.062, 23-day period) on a reclaimed surface mine was low compared to that reported in other studies (Parent et al. 2012), and was not influenced in a meaningful manner by our habitat treatments (Part III). We did not detect evidence of multi-scale influences of habitat on nest survival. Instead, we found that the age of a nest during incubation and time a nest was initiated during the nesting season had the greatest effect on nest survival. The probability of a successful nest increased throughout the 23-day incubation period as well as the nesting season. Specifically, our model results {NestA^2} showed that DSR of nests increased significantly from day 1 to day 10 and then gradually leveled off to day 23 (Klett and Johnson 1982). This may have been due to differences in vulnerability of nest sites to predators (Ricklefs 1969); nests at high-risk sites are quickly found and destroyed by predators, while well-placed nests have less of a chance of being located by a predator.

Our LSA revealed that no single vital rate was capable of achieving population stability ($\lambda = 1$), when solving univariate equations, suggesting that our demographic parameters were either biased low or depressed below sustainable levels (Part IV). In contrast to past demographic studies conducted on bobwhite (Sandercock et al. 2008, Gates et al. 2012, Williams et al. 2012) that found winter survival to be the most important vital rate, our LSA demonstrated that population growth rate was affected the most by fecundity parameters. Specifically, clutch size (r^2 =0.384), hatching success ($r^2 = 0.207$), and nest survival ($r^2 = 0.141$) were major factors explaining variation in λ. In populations with relatively low adult survival, such as bobwhite, the elasticity of λ to changes in fecundity or juvenile survival may dictate that they are key vital rates (Heppell 1998). Indeed, bobwhites exhibit a reproductive system that responds to and enables recovery from high annual mortality and catastrophic events (Stanford 1972, Roseberry and Klimstra 1984, Suchy and Munkel 2000).

The fact that fecundity parameters made a large contribution to variance in λ as well as our exceptionally low summer survival may have been a result of our study being conducted on a reclaimed surface mine. Our study site was comprised primarily of sericea lespedeza as a result of reclamation efforts aimed at quickly reducing erosion. Although sericea lespedeza provides usable space (Guthery 1997) and acceptable cover for bobwhite, sericea seeds are virtually indigestible and provide no nutritional benefit (Davison 1958). In addition, sericea lespedeza litter accumulation has been associated with reduced forb establishment and species richness (Foster and Gross 1998) and may actively interfere with germination of other species through allelopathic compounds generated in leaf and stem tissue (Adams et al. 1973, Wade 1989). Bugg and Dutcher (1989) reported invertebrate use of sericea lespedeza was lowest of all plant species studied. During the nesting and chick-rearing season, hens require a large proportion of invertebrate protein in their diet (Wiens and Rotenberry 1979). Likewise, bobwhite chicks require a high protein diet during the first two weeks of life, with invertebrate species comprising 94.1% of all foods eaten (Eubanks and Dimmick 1974). With this is mind, more beneficial plant species may have been lacking on our site; thus, reducing invertebrate density and seed production. In regards to low survival, these factors may have contributed to lower adult body condition as well as sub-optimal foraging conditions for newly hatched chicks during the summer.

Given the importance of loss and fragmentation of habitat, management of reclaimed surface mines may provide biologists and managers with an atypical means of impacting declining bobwhite populations on a large scale. Low occurrences of desirable food plants on severely disturbed, reclaimed habitats may offer new challenges to managers. Food item availability and/or occurrence within the habitat area should be ascertained when making any inferences about survival on reclaimed habitats which are dominated by perennial plant species (Jones et al. 1994). Although yearly and seasonal variation are unavoidable, management on reclaimed surface mines and other cover-limited sites should maintain existing woody cover while adding new cover through shrub plantings taking care to increase interspersion of this cover. Furthermore, management efforts that promote the growth and existence of valuable food plants for bobwhite may further increase summer survival. A faster return interval of applied treatments may be the most effective way to promote the persistence of native plant species and reduce the coverage of non-native species, such as sericea lespedeza. Additional experiments across a range of habitat conditions over a longer period of time may be required to determine management intensity and duration thresholds required to elicit greater changes in survival for northern bobwhite populations.

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VITA

David was born and raised in southwest Virginia. There, he grew up on a cattle and tobacco farm where he gained a value of the land, spending much of his time hunting, hiking, and fishing. It was here that he gained a true passion for the outdoors. His love of nature and the outdoors led him to Lincoln Memorial University in Harrogate, TN, where he earned a B.S. in Wildlife & Fisheries Management. He worked as a wildlife research technician for Kentucky Department of Fish and Wildlife Resources through the SCA during his undergraduate studies. After graduating, David worked as an avian field technician for Virginia Tech University on a coal mine in Wise, Virginia. He also worked as a field technician on a deer and turkey telemetry project in Delaware prior to pursuing his Master of Science degree at the University of Tennessee in wildlife science, studying northern bobwhite population ecology on reclaimed mined lands. There, he gained an extreme passion for management of upland game birds and early successional ecosystems.