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Northern bobwhite (Colinus virginianus) population ecology on reclaimed mined lands

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To the Graduate Council:

I am submitting herewith a thesis written by Evan Philip Tanner entitled "Northern bobwhite (Colinus virginianus) population ecology on reclaimed mined lands." I have examined the final electronic copy of this thesis for form and content and recommend that it be accepted in partial fulfillment of the requirements for the degree of Master of Science, with a major in Wildlife and Fisheries Science.

Patrick Keyser, Major Professor

We have read this thesis and recommend its acceptance:

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(Original signatures are on file with official student records.)

Northern Bobwhite (*Colinus virginianus*) Population Ecology on Reclaimed Mined Lands.

A Thesis Presented for the

Master of Science

Degree

The University of Tennessee, Knoxville

Evan Philip Tanner

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ABSTRACT

The northern bobwhite (*Colinus virginianus*) has experienced range-wide population declines for the past half century. The primary cause has been large-scale habitat loss and fragmentation. Through auspices of the Surface Mining Control and Reclamation Act of 1977 (SMCRA), large tracts of early successional vegetation have been created throughout much of the bobwhite's range that may be managed to increase usable space. Peabody WMA is a reclaimed coal mine in Western Kentucky where bobwhite have been present in the past. To better understand the dynamics of this population and how habitat on Peabody WMA influences these dynamics, my two objectives were to (1) document survival, cause-specific mortality, and assess multi-scale habitat effects on survival of bobwhite, and (2) estimate nest survival, reproductive efforts, and gauge the effects of habitat composition on these parameters across multiple scales. In relation to bobwhite survival, there was not evidence of multi-scale habitat influence (Part II). Survival increased as the amount of forest vegetation increased within a home range. This was likely related to the availability of woody escape cover associated with forest vegetation on our study site. Pooled seasonal survival rates differed between Ken (S = 0.316, SE = 0.027) and Sinclair (S = 0.141, SE = 0.022) sites. This may have been attributed to differences in habitat suitability or predator abundance. Nest survival rate was low relative to other research (S = 0.317, SE = 0.081; Part III). Nest age was the most influential factor relative to nest survival on our study site and had a positive relationship. Evidence of micro-habitat effects on nest survival existed, though these effects were minimal. Nest survival increased as distance to bare ground increased. This is likely related to the importance of nest concealment on our study site. Our results show that reclaimed mined lands can provide usable space to support bobwhite populations. Management efforts should focus on increasing woody cover within reclaimed

vegetation blocks to increase bobwhite survival while increasing the amount of native warm-season grasses for nesting vegetation and litter cover to support reproductive efforts.

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PART I

INTRODUCTION

Region-wide declines of northern bobwhite (*Colinus virginianus*) populations have been reported throughout the bobwhite's range since the early 1900's (Leopold 1931). More recently, the North American Breeding Bird Survey (BBS) has confirmed this population decline, with an annual decrease of 3.8% in the United States between 1966 – 2009 (Sauer et al. 2011). Within Kentucky, an annual decrease of 2.8% during that same period has been recorded. Although there are many factors that may influence such declines, it has become clear that the major causative factor is a loss habitat (Guthery 1997, Brady et al. 1998, Veech 2006). Much of this habitat loss can be attributed to increased use of clean farming practices combined with silvicultural practices that increase tree density within stands (Brennan 1991, Twedt et al. 2006). Another factor influencing the loss of bobwhite habitat is the decreased use of prescribed burning (Twedt et al. 2006). Prescribed burning promotes early successional habitat and increases grassland plant species richness (Collins 1987), both of which contribute to bobwhite population viability (Stoddard 1931, Greenfield et al. 2003). A steady increase in urbanization, along with intensive silvicultural and agricultural practices, has led to substantial fragmentation of early successional habitat (Terhune et al. 2005) essential for sustaining bobwhite populations at a landscape scale. Remaining early successional vegetation in the southeastern United States have lost much of their native grass component, having been converted into row crops or tall fescue (*Schedonorus phoenix*) and other exotic grasses.

An opportunity for increasing bobwhite habitat throughout the eastern United States is management of reclaimed surface mine sites. Although research regarding bobwhite response to habitat associated with reclaimed mine sites is lacking, studies have shown such sites provide habitat for several early successional specialists (Allaire 1978, Whitmore and Hall 1978, Devault et al. 2002, Karo 2009).

Grassland vegetation has been established in the eastern United States under the auspices 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, land that has been impacted by surface mining may qualify for a plan to reclaim the area for environmental improvement following mining. This has led to the reclamation of more than 600,000 ha in the eastern United States, of which more than 200,000 ha are in Kentucky (Table 1.1). However, establishment of dense stands of sericea lespedeza (*Lespedeza cuneata*) and other non-native herbaceous species is common on these reclaimed mine sites. Such vegetation has been used to establish cover that minimizes soil erosion. Surface mine reclamation success has been assessed in the short-term (e.g., <5 years), such that the establishment of plant species diversity was of lower priority compared to the prevention of soil erosion (Holl 2002).

Although these non-native species have been effective in reducing erosion, the resulting habitat may be unfavorable for bobwhite quail (Eddy 1999). Sericea lespedeza is an aggressive perennial legume that out-competes native grasses. 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 of poor quality and may be heavily compacted. The ability of sericea lespedeza to become established and be competitive in a variety of soil types (Ohlenbusch et al. 2007) has also contributed to its domination of reclaimed mine sites. Dense fields of this legume, which often exist on reclaimed sites, provide structure in which bobwhites will seldom nest (Roseberry and Klimstra 1984). Management practices must be focused on removing this uniform structure and restoring the structural variability required by bobwhites for escape, nesting, feeding, and brood rearing cover.

Table 1.1. Eastern US coal-mined land area (ha) reclaimed under SMCRA, 1978-2005[1].

State	Total
E KY	269,627
MD	5,490
OH	83,662
PA	107,029
TN	17,908
VA	38,201
WV	105,358
Total	627,275

[1] Including the interim Surface Mine Control and Reclamation Act program. Source US Office of Surface Mine Reclamation and Enforcement "20th Anniversary of the Surface Mining Law" (http://www.osmre.gov/annivrep.htm) and annual reports to Congress.

Studies monitoring bobwhite population dynamics as a function of vegetation types and quality have been focused at both the local and landscape levels in a number of ecological regions. Studies in the central and western Great Plains have evaluated macro-habitat feature influences on bobwhite summer survival (Taylor et al. 1999), survival of bobwhite chicks (DeMaso et al. 1997), over-winter habitat use and winter survival (Williams et al. 2000, Williams et al. 2004), and population responses to habitat management (Webb and Guthery 1982). Cox et al. (2004) also evaluated survival and mortality of bobwhites within this region. In the Midwest, studies have examined effects of hunting pressure on survival rates (Suchy and Munkel 2000), population dynamics related to weather parameters and hunting pressure (Stanford 1972), effects of habitat use on non-breeding survival (Janke 2011), and detailed ecology of localized bobwhite populations (Roseberry and Klimstra 1984, Burger 1995b). Roseberry and Klimstra (1984) conducted an intensive 26-year population ecology study using banding in Illinois, assessing survival, cause-specific mortality, fecundity, and hunting effects on bobwhite survival. In Missouri, Burger et al. (1995) evaluated general bobwhite population dynamics and specific causes of mortality. Numerous bobwhite studies have been conducted in the Red Hills region of the Gulf Coastal Plain and have included evaluation of effects of research on bobwhite survival (Terhune et al. 2005), demographic responses to different burning scales (Wellendorf and Palmer 2007), over-winter survival in relation to landscape composition (Holt et al. 2009), and evaluation of population dynamics (Pollock et al. 1989, Palmer et al. 2002). Burger et al. (1998) evaluated bobwhite survival and cause-specific mortality within this region on intensively managed plantations. 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 region, 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). Research efforts on population ecology of bobwhites, survival of hunted vs. non-hunted populations, and effects of vegetation on bobwhite survival have been extensively studied throughout many regions within the species' range. However, few population studies have examined northern bobwhites in the Central Hardwoods Conservation Region. Except for a study examining winter survival as a function of landscape composition in western Tennessee (Seckinger et al. 2008), extensive population dynamic studies within this region are entirely lacking.

Although there have been studies monitoring populations of early successional passerines (Whitemore and Hall 1978, Devault et al. 2002) and game birds (Karo 2009) in the context of reclaimed strip mines, none has monitored bobwhite populations. 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, as habitat use and survival has been shown to vary seasonally (Burger et al. 1995, Sisson et al. 2009, Lohr et al. 2011). It is also important to assess bobwhite population responses to large-scale habitat management efforts, as bobwhite home ranges vary with habitat composition and individual reproductive status (Brennan 1999). Studies in the past have focused on management efforts at a relatively small scale. Research must assess large-scale habitat management and its effects on bobwhite populations as it has been suggested that this scale influences bobwhite population dynamics (Williams et al. 2004, Seckinger et al. 2008). The temporal scale can also be influential in population responses to habitat management and should be evaluated.

This research was initiated to evaluate bobwhite population dynamics on a reclaimed surface mine site in western Kentucky. Our objectives were to (1) document survival rates at two scales: home range and landscape scale, and (2) document fecundity, including nest success and nest productivity, as a function of habitat parameters. In Part II, we assessed survival rates as a function of habitat parameters during the winter and summer seasons at the home range and landscape scales. In Part III, we assessed nest success as a function of habitat parameters at the microhabitat and landscape scales. Parts II and III are written as stand-alone manuscripts for future publication.

LITERATURE CITED

Allaire, P. N. 1978. Reclaimed surface mines: new potential for some North American birds. American Birds 32(1):3-5.

Brady, S. J., C. H. Flather, K. E. Church. 1998. Range-wide declines of northern bobwhite (*Colinus virginianus*): Land use patterns and population trends. International Symposium on Partridges Quails and Pheasants in the Western Palearctic and the Nearctic. Gibier Faune Sauvage, Game Wildlife 15: 413-431.

Brennan, L. A. 1991. How can we reverse the northern bobwhite population decline? Wildlife Society Bulletin 19: 544-555.

Brennan, L. A. 1999. Northern Bobwhite. The Birds of North America. No. 397.

Burger, Jr., L. W., T. V. Dailey, E. W. Kurzejeski, M. R. Ryan. 1995. Survival and cause-specific mortality of northern bobwhite in Missouri. The Journal of Wildlife Management 59(2): 401-410.

Burger, Jr., L. W., D. C. Sisson, H. L Stribling, D. W. Speake. 1998. Northern bobwhite survival and cause-specific mortality on an intensively managed plantation in Georgia. Proc. Annu. Conf. Southeast Association of Fish and Wildlife Agencies 52:174-190.

Collins, S. L. 1987. Interaction of disturbances in tallgrass prairie: a field experiment. Ecology 68: 1243-1250.

Cox, S. A., A D. Peoples, S. J. DeMaso, J. J. Lusk, F. S. Guthery. 2004. Survival and cause-specific mortality of northern bobwhites in western Oklahoma. The Journal of Wildlife Management 68:663-671.

Curtis, P.D., B. S. Mueller, P. D. Doerr, and C. F. Robinette. 1988. Seasonal survival of radio-marked northern bobwhite quail from hunted and non-hunted populations. Biotelemetry X: International Radio-telemetry Symposium 10:263-275.

DeMaso, S. J., A. D. Peoples, S. A. Cox, E. S. Parry. 1997. Survival of northern bobwhite chicks in western Oklahoma. The Journal of Wildlife Management 61:846-853.

DeVault, T. L., P. E. Scott, R. A. Bajema, S. L. Lima. 2002. Breeding bird communities of reclaimed coal-mine grasslands in the American Midwest. Journal of Field Ornithology 73(3):268-275.

Dixon, K. R., M. A. Horner, S. R. Anderson, W. D. Henriques, D. Durham, R. J. Kendall. 1996. Northern bobwhite habitat use and survival on a South Carolina plantation during winter. Wildlife Society Bulletin 24(4):627-635.

Eddy, T. A. 1999. Effects of sericea lespedeza infestations on wildlife habitat in Kansas. 61st Midwest Fish and Wildlife Conference, Iowa Conservation Commission, Des Moines.

Eddy, T. A., J. Davidson, B. Obermeyer. 2003. Invasion dynamics and biological control prospects for sericea lespedeza in Kansas. Great Plains Research 13 (Fall 2003): 217-30.

Greenfield, K. C., M. J. Chamberlain, L. W Burger, Jr., E. W. Kurzejeski. 2003. Effects of burning and discing Conservation Reserve Program fields to improve habitat quality for northern bobwhite (*Colinus virginianus*). The American Midland Naturalist 149:344-353.

Guthery, F. S. 1997. A philosophy of habitat management for northern bobwhites. The Journal of Wildlife Management 61:291-301.

Holl, K. D. 2002. Long-term vegetation recovery on reclaimed coal surface mines in the eastern USA. The Journal of Applied Ecology 39:960-970.

Holt, R. D., L. W. Burger, Jr., B. D. Leopold, D. Godwin. 2009. Over-winter survival of northern bobwhite in relation to landscape composition and structure. Pages 432-446 in S. B. Decerbaum, B. C. Faircloth, T. M. Terhune, J. j. Thompson, J. P. Carroll, eds. Gamebird 2006:Quail VI and Perdix XII. 31 May-4 June 2006. Warnell School of Forestry and Natural Resources, Athens Georgia, USA.

Janke, A. K. 2011. Survival and habitat use of non-breeding northern bobwhites on private lands in Ohio. Thesis, Ohio State University, Columbus, USA.

Karo, R. 2009. Sharp-tailed grouse return to mined land. 2009 National Meeting of the American Society of Mining and Reclamation, Revitalizing the Environment: Proven Solutions and Innovative Approaches. Published by ASMR.

Leopold, A. 1931. Report on a game survey of the north central states. Democrat Printing Company, Madison, Wisconsin, USA.

Lohr, M. T., B. M. Collins, C. K. Williams, and P. M. Castelli. 2011. Life on the edge: Northern bobwhite ecology at the northern periphery of their range. Journal of Wildlife Management 75:52-60.

Ohlenbusch, P. D. 2007. Sericea lespedeza: history, characteristics, and identification. Kansas State University.

Palmer, W. E., S. D. Wellendorf, L. A. Brennan, W. R. Davidson, F. E. Kellogg. 2002. Hunting success and northern bobwhite density on Tall Timbers Research Station: 1970-2001. Pages 213-216 *in* S. J. DeMaso, W. P. Kuvlesky, Jr., F. Hernández, and M. E. Berger, eds. Quail V: Proceedings of the Fifth National Quail Symposium. Texas Parks and Wildlife Department, Austin, TX.

Palmer, W. E., S. D. Wellendorf. 2007. Effect of radiotransmitters on northern bobwhite annual survival. The Journal of Wildlife Management 71: 1281-1287.

Pollock, K. H., C. T. Moore, W. R. Davidson, F. E. Kellogg, G. L. Doster. 1989. Survival rates of bobwhite quail based on band recovery analyses. The Journal of Wildlife Management 53:1-6.

Robinette, C. F. and P. D. Doerr. 1993. Survival of northern bobwhite on hunted and non-hunted study areas in the North Carolina Sandhills. Pages 74-78 in K. E. Church and T. V. Dailey, eds. Quail III: National Quail Symposium. Kansas Dept. Wildl. And Parks, Pratt.

Roseberry, J. L., W. D. Klimstra. 1984. Population ecology of the bobwhite. Southern Illinois University, Carbondale, USA.

Sauer, J. R., J. E. Hines, J. E. Fallon, K. L. Pardieck, D. J. Ziolkowski, Jr., and W. A. Link. 2011. The North American Breeding Bird Survey, Results and Analysis 1966-2009. Version 3.23. 2011. USGS Patuxent Wildlife Research Center, Laurel, Maryland, USA.

Seckinger, E. M., L. W. Burger, Jr., R. Whittington, A. Houston, R. Carlisle. 2008. Effects of landscape composition on winter survival of northern bobwhites. The Journal of Wildlife Management 72(4):959-969.

Sisson, D. C., T. M. Terhune, H. L. Stribling, J. F. Sholar, S. D. Mitchell. 2009. Survival and causes of mortality for northern bobwhites in the southeastern USA. Pages 467-478 in Cederbaum, S. B., B. C. Faircloth, T. M. Terhune, J. J. Thompson, J. P. Carroll, eds. Gamebird 2006: Quail VI and Perdix XII. 31 May- 4 June 2006. Warnell School of Forestry and Natural Resources, Athens Georgia, USA.

Stanford, J. A. 1972. Bobwhite quail population dynamics: relationships of weather, nesting, production patterns, fall population characteristics, and harvest in Missouri quail. Proceedings of the National Bobwhite Quail Symposium 1: 115-139.

Stoddard, H. L. 1931. The bobwhite quail: its habits, preservation and increase. Charles Scibner’s Sons, New York, New York, USA.

Suchy, W. J., R. J. Munkel. 2000. Survival rates for northern bobwhites on two areas with different levels of harvest. Pages 140-146 *in* L. A. Brennan, W. E. Palmer, L. W. Burger, Jr., T. L. Pruden (eds.). Quail IV: Proceedings of the Fourth National Quail Symposium. Tall Timbers Research Station, Tallahassee, FL.

Taylor, J. S., K. E. Church, D. H. Rusch, J. R. Cary. 1999. Macrohabitat effects on summer survival, movements, and clutch success of northern bobwhite in Kansas. The Journal of Wildlife Management 63(2):675-685.

Terhune, T. M., D. C. Sisson, J. B. Grand, H. L. Stribling. 2005. Factors influencing survival of radiotagged and banded northern bobwhites in Georgia. The Journal of Wildlife Management 71: 1288-1297.

Twedt, D. J., R. R. Wilson, A. S. Keister. 2006. Spatial models of northern bobwhite populations for conservation planning. The Journal of Wildlife Management 71: 1808-1818.

Veech, J. A. 2006. Increasing and declining populations of northern bobwhites inhabit different types of landscapes. The Journal of Wildlife Management 70:992-930.

Webb, W. M., F. S. Guthery. 1982. Response of bobwhite to habitat management in northwest Texas. Wildlife Society Bulletin 10(2):142-146.

Wellendorf, S. D., W. E. Palmer. 2007. Effects of two burning scales on northern Bobwhite demographic rates and home range size. National Quail Symposium. *In Press*

Whitmore, R. C., G. A. Hall. 1978. The response of passerine species to a new resource: reclaimed surface mines in West Virginia. American Birds 32:6-9.

Williams, C. K., R. S. Lutz, R. D. Applegate, D. H. Rusch. 2000. Habitat use and survival of northern bobwhite (*Colinus virginianus*) in cropland and rangeland ecosystems during the hunting season. Canadian Journal of Zoology 78:1562-1566.

Williams, C. K., R. S. Lutz, R. D. Applegate. 2004. Winter survival and additive harvest in northern bobwhite coveys in Kansas. The Journal of Wildlife Management 68:94-100.

PART II

INFLUENCE OF MULTI-SCALE HABITAT ATTRIBUTES ON NORTHERN BOBWHITE SURVIVAL ON RECLAIMED MINED LANDS

ABSTRACT Through the auspices of the Surface Mining Control and Reclamation Act of 1977 (SMCRA), large tracts of early successional vegetation have been created throughout much of the northern bobwhite (*Colinus virginianus*) species' range. Such reclaimed lands offer potential habitat for bobwhite. An understanding of multi-scale habitat effects on bobwhite survival and habitat use is essential to successfully managing large tracts of land for viable populations of this species. To date, no study has assessed bobwhite survival, habitat characteristics, or the relationship between them on reclaimed mined land. To better understand this relationship, we used radio telemetry on Peabody Wildlife Management Area (WMA), a 3,330 ha reclaimed surface mine in western Kentucky. We conducted research across two sites on Peabody WMA (Sinclair and Ken). We captured bobwhites from Sep 2009-Sep 2011 during non-breeding (1 Oct-31 March) and breeding (1 Apr-30 Sep) seasons. A total of 841 birds were fitted with necklace-style radio-collars of which 619 were used in analysis. We used the known fate model in Program MARK to estimate seasonal survival rates from 61 *a priori* models at 2 spatial scales, home range and landscape. Seasonal survival differed ($\chi^2 = 7.87$, $P = 0.005$) between sites (Sinclair = 0.141, 95% CI = 0.097-0.184; Ken = 0.316, 95% CI = 0.263-0.368) over the study period. Of all the candidate models, those including weekly time interaction, group (whether birds had estimated home ranges or not), year, season, and the percentage of forest vegetation in a home range effects were best supported (AICc weights = 0.807). Survival was positively related with the amount of forest in a home range ($\beta = 0.024$, CI = 0.003-0.462). The amount of open herbaceous core area at a landscape scale ($\beta = 0.084$, CI = -0.020-0.188) was also in the top model, but this effect did not differ from 0. Based on our results, there was no evidence of multi-scale habitat effects on bobwhite survival. We suggest management efforts should focus on improving habitat at the local scale by providing woody escape cover in large planted blocks of

reclaimed vegetation, while also maintaining forest understory structure characteristic of open woodlands.

Northern bobwhite (*Colinus virginianus*), here after “bobwhite,” have experienced a 3.8% annual decline throughout the species’ range and a 2.8% decline in Kentucky between 1966–2009 (Sauer et al. 2011). Habitat degradation resulting from clean farming practices, urban sprawl, advancement of succession, and the decrease in managed disturbance have been attributed to these range-wide declines in populations (Brennan 1991, Williams et al. 2004, Twedt 2006). Habitat fragmentation has exacerbated these problems by isolating remaining habitat. It is imperative to re-establish early successional vegetation at a landscape scale to reverse declining population trends (Guthery 1997, Dimmick et al. 2002, Williams et al. 2004).

Previous research has stressed the importance of directly assessing habitat attributes and their influence on bobwhite survival to develop strategies that can lead to increased population densities (Taylor et al 1999, Seckinger et al. 2008, Holt et al. 2009). Furthermore, the need to assess these attributes at different spatial scales has been suggested (Brady et al. 1993, Roseberry 1993), as habitat fragmentation has led to the need for assessing habitat attribute effects on survival at a broad scale, rather than just a local scale. Furthermore, understanding broader scale constraints on habitat may provide insight on how to best allocate resources for local-scale habitat improvement efforts. Also, Seckinger et al. (2008) suggested habitat composition at both local and landscape levels may be important in understanding bobwhite mortality in relation to predation.

Reclaimed mined lands offer a unique opportunity to increase the amount of habitat at a large-scale for bobwhite in many areas of the eastern United States. Large tracts of early successional vegetation are often created through the auspices of the Surface Mining Control and Reclamation Act of 1977 (SMCRA). Although much of the area reclaimed is in early successional vegetation, these lands are often vegetated with species that may not provide suitable food, or at seeding rates in which vegetation structure is not ideal (Eddy 1999). To better understand the effects of vegetation composition of reclaimed mined lands on survival, research must focus at both home range and landscape scales.

Although bobwhite is an extensively studied species (Burger et al. 1995), evaluations of survival and cause-specific mortality within the Central Hardwoods Conservation Region are limited. Furthermore, few studies have addressed the potential of reclaimed mined lands for bobwhite (Beckerle 2004), and no studies have related habitat characteristics of these areas to bobwhite survival. Seckinger et al. (2008) and Janke (2011) showed evidence of multi-scale habitat influence on survival. Because mine lands are often quite large, the need to directly assess habitat effects on survival at both home range and landscape levels is necessary. Furthermore, studies relating multi-scale habitat metrics to survival rates tend to focus on a single season over multiple years (Taylor et al. 1999, Seckinger et al. 2008, Holt et al. 2009), and only Lohr et al. (2011) included multiple seasons. As seasonality has been shown to have an influence on bobwhite survival (Curtis et al. 1988, Burger et al. 1995), both breeding and non-breeding season should be assessed when relating survival to multi-scale habitat attributes.

Little is known about the suitability of reclaimed mined lands for supporting populations of bobwhite (Beckerle 2004). The reclamation process can create a unique vegetative landscape, and an understanding how this composition affects bobwhite survival is essential for implementing effective management that optimizes population size. To understand if reclaimed mined lands can support viable bobwhite populations, and how habitat on reclaimed mined lands affected bobwhite survival, we conducted a radio telemetry study on Peabody Wildlife Management Area (WMA), Kentucky, USA from 2009-2011. Because much of Peabody WMA was planted in uniform “blocks” of vegetation during the reclamation process, we hypothesized bobwhite survival would be higher where woody escape cover was available at the local scale and where there was increased interspersion of early successional vegetation and suitable woody cover at the landscape scale. Our primary objective was to determine which vegetation attributes contributed to increased bobwhite survival on reclaimed mined land, and if there was scale dependency related to these vegetation attributes. We also sought to document overall survival and cause-specific mortality of bobwhite on a reclaimed coal mine. To assess the importance of scale on survival in relation to vegetation attributes, we focused our analysis at the home range and landscape scales.

STUDY AREA

We conducted the study on a reclaimed coal mine, Peabody WMA (3,323 ha) in Muhlenberg (37°14'N, 87°15'W) and Ohio (37°17'N, 86°54'W) counties in western Kentucky, USA. The study area consisted of open herbaceous vegetation (36%; Table A.1), which was dominated by sericea lespedeza (*Lespedeza cuneata*) and annual forbs such as, common ragweed (*Ambrosia artemisiifolia*), sumpweed (*Iva annua*), and goldenrod (*Solidago spp.*). Shrub vegetation (25%) was characterized by an abundance of black locust (*Robinia pseudoacacia*), winged sumac (*Rhus copallinum*), and blackberry (*Rubus spp.*). Deciduous forests (22%) primarily consisted of eastern cottonwood (*Populus deltoides*), green ash (*Fraxinus pennsylvaticum*) and American sycamore (*Platanus occidentalis*), and typically had a well-developed understory consisting of blackberry (*Rubus spp.*) and honeysuckle (*Lonicera japonica* and *Lonicera maakii*). More recently, native warm-season grasses (NWSG), including mixtures of big bluestem (*Andropogon gerardii*), little bluestem (*Schizachyrium scoparium*), indiangrass (*Sorghastrum nutans*), and switchgrass (*Panicum virgatum*), have been established (8%). Small lakes, wetlands, and annual grain food plots comprised the remainder (9%) of our study area. Forests on the WMA were established under guidelines in place prior to the passage of the SMCRA ("pre-law"), while all early successional vegetation was established under post-law criteria. Habitat management on both units include dormant-season (January-March) prescribed fire, disking (all months), herbicide spraying, and plantings of food plots and NWSG. Efforts have focused on maintaining early successional vegetation while trying to limit coverage of invasive, non-native plants (particularly sericea lespedeza) that had been established previously.

We conducted our research on two different sites on Peabody WMA (Ken and Sinclair). These two sites are separated by the Green River and are 18 kilometers apart. As no birds were detected moving between Ken and Sinclair, we considered them separate sites. Vegetation on Sinclair (1470 ha) was 45% open herbaceous, 22% scrub-shrub, 22% forest, and 4% NWSG. The remaining 7% consisted of small lakes, wetlands, and annual grain food plots. Ken (1853 ha) consisted of 28% open herbaceous, 28% scrub-shrub, 22% forest, 11% NWSG. The remaining 11% consisted of small lakes, wetlands, and annual grain food plots.

METHODS

Land Cover

Four major vegetation types (forest, scrub-shrub, open herbaceous, and NWSG) were delineated based on aerial imagery in ArcGIS 9.3 (ESRI, Redlands, CA, USA), which constituted 91% of the total land cover on our study site. To delineate between forest, scrub-shrub, 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 1m x 1m 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, scrub-shrub 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 scrub-shrub, 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 scrub-shrub 9.6 m^2/ha (SE = 1.23); scrub-shrub 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.

Data Collection

We captured bobwhites year-round (Sep 2009 - Sep 2011) using funnel traps (Stoddard 1931), which were covered with burlap and vegetation to help reduce stress and predation of captured birds. We defined a biological year as 1 Oct-30 Sep and seasons as non-breeding (1 Oct-31 Mar) and breeding (1 Apr-30 Sep), based on Burger et al. (1995). We strategically placed traps (*n* = 120) in areas thought to have birds and where birds were heard or seen. We fitted captured birds with necklace-style collars weighing 6g (crystal-controlled, two-stage design, pulsed by a CMOS multivibrator, American Wildlife Enterprise, Monticello, Florida, USA) based on meeting a minimum body mass requirement (120g) and availability of collars. We assumed radio transmitters did not affect survival (Palmer and Wellendorf 2007, Terhune et al. 2007). We leg-banded (double) all captured birds. We determined sex, age, and weight of all birds, and released birds at their capture site. During the breeding season, we classified the sex of a bird as unknown if we were not able to determine sex because of the bird's age. We determined if a bird was an adult by the absence of a buff-tipped primary covert (Stoddard 1931). Our trapping and handling methods complied with University of Tennessee Institutional Animal Care and Use Committee Permit (no. 2042-0911) protocol.

We attempted to locate radio-marked individuals at least three times/week using a scanning receiver and a handheld Yagi antenna (Advanced Telemetry Systems, Inc., Isanti, Minnesota, USA). We located birds by homing (White and Garrot 1990) within 50m to avoid flushing birds. Once birds were located, we recorded the distance and azimuth to the actual bird location and recorded the Universal Transverse Mercator (UTM) coordinates of the observer on a GPS unit (Garmin GPSMAP 60CSx, Garmin International, Inc. Olathe, Kansas, USA). We then used the distance and azimuth to estimate the location of each bird. We recorded locations of birds at different times on subsequent days to capture the variability of diurnal patterns. We recorded the vegetation type in which the bird was located based on our four major vegetation categories. We located transmitters emitting a mortality signal (12-hr signal) immediately after detection 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).

Population Estimation

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). 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 potential overlap (Rusk et al. 2007), which provided 47% coverage of our 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 (DeMaso et al. 1992) and ceased monitoring 20 minutes beyond the last call recorded (Guthery 1986). 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 (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 able to identify the covey call (“*koi-lee”*) prior to data collection. We visited each survey point twice per fall. We compared calling rates of coveys located with telemetry equipment prior to each fall survey, thus providing a correction factor for calling rate (Riddle et al. 2008). We estimated average covey size by flushing coveys detected during the survey with bird dogs within 12 hours of completion of the survey. 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). We calculated standard errors based on methods from Ott (1993).

Home Range Estimation

We calculated home ranges for individual birds with ≥20 locations (DeVos and Mueller 1993, Taylor et al. 1999) using the 95% fixed-kernel method (Worton 1989, Seaman et al. 1999) and the Animal Movement Extension (Hooge and Eichenlaub 1997) in ArcView 3.2 (Environmental Systems Research Institute, Inc., Redlands, California, USA). During the non-breeding season, home ranges were estimated for individuals rather than coveys and survival estimates were derived for individual birds.

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 post-release to control for a potential short-term impact associated with capturing and radio-marking (Guthery and Lusk 2004). We used a staggered-entry method, which left-censors individual's encounter histories until they are captured and enter the monitored population, to analyze survival (Pollock et al. 1989). We right-censored individuals because of emigration from the study area, radio failure or loss, or unknown fate. Each survival period (non-breeding and breeding) consisted of 183 days.

Our survival analysis consisted of three hierarchical steps consisting of three different suites of models. These three suites of models represented class metrics and two different spatial scales: home range and landscape. Based on *a priori* models, we used a model-selection approach based on Akaike's Information Criterion (AIC) to determine the model that best explained survival within our suites of models. We used a ΔAIC_c value of <2 (Burnham and Anderson 2002) to determine validity of a model for explaining variance in survival. We summed the weights of models containing parameters of interest to assess the strength of the individual covariate. After analyzing all three suites of models, we computed the model-averaged parameter estimates for daily survival using the survival estimate from each model. We then used the delta method (Powell 2007) to expand estimates to a temporal scale that encompassed a biological season. For survival analysis, the effects of different covariates were assessed based on published studies and biological importance. For the class suite, we included: sex, age, weight, site, year, season, linear time, and weekly time effects (Table A.2).

We also separated birds into two groups based on whether or not they were monitored enough (i.e., >20 locations, typically >7 weeks) to have enabled us to have estimated a home range. Using this approach, we were able to include all birds in our analysis regardless of whether or not they had home ranges and associated vegetation metrics. Had we excluded the short-lived birds because of a lack of associated vegetation metrics, we would have biased our survival estimates. Vegetation metrics were calculated only for birds with estimated home ranges because we were able to establish a reliable, explicit spatial context for these individuals that could then be georeferenced to our vegetation layers. Additive models were also assessed to incorporate any additive effects between multiple covariates. Our top model from the first suite was then used as the baseline model in subsequent analyses.

At the home range scale, we included the proportion of a vegetation type in an individual's home range (Scrub-shrub, Forest, NWSG, and Open Herbaceous) and the home range size as covariates (Table A.2). All models we evaluated within this suite included the effect of the top model from the first stage of our analysis.

At the landscape scale, we included metrics associated with a buffer placed around each home range. Buffers were created in ArcGIS 9.3 using a radius equal to double the average daily movement observed during our study within each season: breeding (128m) and non-breeding (138m). Average daily movement was calculated as the mean distance between consecutive daily locations for an individual, averaged across all individuals (Holt et al. 2009). Similar studies have used a buffer equal to the mean daily movement observed during the study (Holt et al. 2009). We decided to use double the average daily movement to help ensure we captured the landscape where any bird could have theoretically traveled based on their actual locations. We selected, *a priori,* nine landscape-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, Scrub-shrub/open vegetation (both NWSG and Open Herbaceous) edge density, core area of all four major vegetation types (using a 30m edge effect), and a contagion index (Table A.2). The contagion index is a measure of patch-type interspersion and overall patch dispersion (O'Neill et al. 1988), and influences 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 150m moving window. Based on values obtained from the moving window, we averaged metrics within each buffered home range. Using these landscape covariates, we developed our third suite of models. We hypothesized that, at the landscape scale, edge density between woody cover and open vegetation would increase survival of individuals (Leopold 1933). We included models with quadratic relationships between survival and edge density to test for a potential maximum threshold of edge before it begins to decrease survival (Guthery et al. 2001, Duren et al. 2011). We also tested for the effect of patch size and patch dispersion at the landscape level in relation to survival. The top model from our second suite of models was used as an additive effect in all models analyzed at the landscape scale. After incorporating the landscape scale models, the best approximating model based on the ΔAIC_c score was considered our best overall model across all three scales. Chi-square tests were used to compare survival among sites, seasons, and sexes. We assessed the possibility of confounding relationships between sites and vegetation variables from our top model through interaction models. We compared the strength of interaction models to the strength of additive models containing the strongest supported covariates after our hierarchical analysis.

In addition to evaluating multi-scale models, we also tested specific hypotheses related to ongoing management at the study area. First, we tested whether an increase in interspersion of vegetation within a buffered home range increased survival. Roseberry and Sudkamp (1998) suggested that an increase in the contagion index, the measure of patch type interspersion, was associated with higher bobwhite densities on an area. Peabody WMA was re-vegetated in large blocks of vegetation resulting in relatively low interspersion (Contagion Index = 48.3 – 56.8). To test this, we compared a model containing the contagion index calculated for each bird's buffered home range to the null model. We used *t* tests to compare contagion indices between sites and seasons. We hypothesized that there would be an increase in survival with an increase in interspersion (decrease in contagion) of vegetation.

Second, we tested if there was evidence of a diminishing return effect of edge density on bobwhite survival. At a landscape scale, this relationship has been shown to affect habitat suitability for bobwhite (Guthery et al. 2001). Although bobwhite has been considered an "edge species", the presence of too much edge could be detrimental to bobwhite survival. To test this, we compared models containing the scrub-shrub/open vegetation edge density and forest/open vegetation edge density covariates to models containing their quadratic effects.

In addition to the two management hypotheses, we also looked to document if hunting was a significant source of mortality on Peabody WMA. Roseberry and Klimstra (1984) observed that hunting mortality was compensatory on populations early in the hunting season, but became additive as time progressed later into the hunting season. To gauge whether or not hunting was a significant source of mortality on our population, we wanted to document the overall number of mortalities resulting from quota hunts on the property.

RESULTS

We captured and double-banded 841 bobwhites (457 males, 326 females, and 58 birds for which we could not determine sex) from 1 Sep 2009 – 30 Sep 2011. We captured more juveniles ($n = 674$) than adults ($n = 167$). Of the 841 captured birds, we radio-marked 627, but were only able to use 619 in our survival analysis because of censoring. We obtained $\geq$20 locations for 235 birds for which we were able to estimate home ranges and associated habitat metrics. Our trap success (number of birds captured divided by the number of trap nights) was greater ($t = 5.49$, $P = 0.03$) in non-breeding seasons (Ken = 4.1%, Sinclair = 4.2%) than in breeding seasons (Ken = 3.0%, Sinclair = 2.5%).

The 2009 fall population estimate was 934 (SE = 450) on Sinclair and 1518 (SE = 731) on Ken. 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 in 2010 was 0.33 (SE = 0.08). The 2010 fall population estimate was 2163 (SE = 706) on Sinclair and 1682 (SE = 549) on Ken. The average covey size in 2010 was 8.91 (SE = 0.70).

The only difference detected in survival rates was between sites (χ^2 = 7.87, *P* = 0.005; Sinclair = 0.141, 95% CI = 0.097-0.184; Ken = 0.316, 95% CI = 0.263-0.368; Table A.3). There was no difference in survival by sex on Sinclair (χ^2 = 0.881, *P* = 0.347; male = 0.122, 95% CI = 0.076-0.167; female = 0.173, 95% CI = 0.108-0.237) or Ken (χ^2 = 1.347, *P* = 0.245; male = 0.286, 95% CI = 0.223-0.348; female = 0.352, 95% CI = 0.281-0.422). Survival was not different between ages on Sinclair (χ^2 = 0.00005, *P* = 0.994; juvenile = 0.141, 95% CI = 0.095-0.186; adult = 0.141, 95% CI = 0.062-0.219) or Ken (χ^2 = 0.00004, *P* = 0.983; juvenile = 0.316, 95% CI = 0.257-0.374; adult = 0.316, 95% CI = 0.227-0.404). Seasonal survival rates based on model averaging differed among Groups (χ^2 = 32.95, *P* = <0.0001) (Figure 2.1). Model averaged seasonal survival rates were 0.06 (SE = 0.02) for birds without home ranges and 0.49 (SE = 0.01) for birds with home ranges. Mammalian predation accounted for the highest percentage of known mortalities during the non-breeding season (40.3%) while avian predation account for the highest percentage of known mortalities in the breeding season (14.5%) (Table A.4).

In our first suite of models, a weekly temporal effect with an interaction effect between Groups was a better approximating model than other temporal models. Therefore, we included the weekly temporal effect with an interaction effect between Groups in our subsequent models. The best overall approximating model for suite one included Group, year, site, and season effects and had an AIC_c weight of 0.52. Beta estimates for year (β = -0.767, CI = -1.002 to -0.531) and season (β = -0.314, CI = -0.549 to -0.080) were negative, indicating a decrease in survival from the first year to the second, and from non-breeding to breeding seasons. The beta estimate for site (β = 0.633, CI = 0.386-0.880) indicated a higher survival on the Ken unit. The top model (wi+g+year+site+season) was used as the baseline model for our second suite of models.

In our second suite of models, proportion of forest within a home range was present in the top three models, and had a combined $\mathrm{AIC_c}$ weight of 0.547. The top model from this suite only contained the additional forest home range variable, and had an $\mathrm{AIC_c}$ weight of 0.251. This model was 1.54 times more likely than the second best approximating model. The beta estimate for the amount of forest within a home range ($\beta = 0.024$, CI = 0.003-0.046) suggested an increase in survival associated with a higher proportion of forest vegetation in a bird's home range. . The home range scale variable with the most support (FOR) had an $\mathrm{AIC_c}$ weight of 0.80. The top model (wi+g+year+site+season+FOR) was used as the baseline model for our final suite of models.

The best supported model at the landscape scale showed a positive relationship between survival and the amount of open herbaceous core area within the buffered home range (wi+g+year+site+season+FOR+OH_CA) (Table A.5). The beta estimate for the amount of open herbaceous core within a buffered home range ($\beta = 0.08$, CI = -0.02-0.18) suggested a weak positive influence on survival, as the confidence interval for this parameter included 0. Although the final top model included the open herbaceous core area variable at the landscape level, the weight for this top model ($\mathrm{AIC_c}$ weight = 0.13) was only slightly greater than the weight for the next best supporting model containing no landscape variables ($\mathrm{AIC_c}$ weight = 0.10) (Table A.5). The group, site, year, weekly interaction, and season variables all had importance weights of >0.99 (Table A.6), suggesting strong effects of these variables on survival. The home range scale variable with the most support (FOR) had an $\mathrm{AIC_c}$ weight of 0.80, suggesting dependency of survival to a home range scale metric. Landscape scale metrics carried little weight, with the top variable having an $\mathrm{AIC_c}$ weight of 0.13. Additive models containing the best supported vegetation covariates (FOR and OH_CA) had more support than interaction models (ΔAIC = 1.28).

With respect to our specific management hypotheses, survival did not increase with interspersion within a bird's buffered home range (Table A.7). The beta value ($\beta = 0.027$, CI = 0.022-0.032) of the contagion index model when analyzed individually exhibited an increase in survival associated with higher contagion index values, although the contagion index was not a significant covariate in our hierarchical analysis ($\beta = 0.024$, CI = -0.04-0.16). Sinclair had a higher (non-breeding season, t = 2.395, $P = 0.018$; breeding season, t = 4.133, $P < 0.001$) contagion index than Ken. As the contagion index value increased from 0-100, the amount of vegetation interspersion decreased. Hunting mortality was not a significant source of mortality during our study. Though hunting occurred on both sites during 2009-2011, there were no marked birds harvested during the course of our study (Table A.4). Finally, there was no evidence of diminishing returns with respect to edge effect on survival, as the quadratic edge models had much lower AIC values compared to the linear edge models (Table A.8). Based on these models, survival increased linearly as the amount of scrub-shrub/open vegetation (β = 0.041, CI = 0.033-0.049) and forest/open vegetation ($\beta = 0.489$, CI = 0.346-0.632) edge density increased.

DISCUSSION

We did not detect evidence of multi-scale dependency on survival of bobwhites on our study site. Although survival on sites differed, survival increased on both sites as the amount of forest within an individual's home range increased. Only one landscape-scale metric (Open Herbaceous Core Area) influenced survival on Peabody WMA, but the confidence interval for the beta estimate included zero, suggesting a minimal impact on survival. Furthermore, even though the amount of open herbaceous core area was included in the best landscape-scale model, the proportion of open herbaceous vegetation was not in a competitive model at the home range scale. Likewise, landscape-scale forest metrics carried little weight, though the amount of forest within a home range was important to survival.

The greatest difference we detected in survival rates was for Group, a result we expected because birds for which we were able to calculate a home range had to survive long enough to be located $\geq$20 times. However, incorporating all 619 birds in our analysis was important to avoid biasing our overall survival estimates, a concern that was validated by our models. Survival rates from model averaging provided low estimates for birds for which we did not calculate a home range (S = 0.06) and high estimates for birds with an estimated home range (S = 0.49) compared to previous studies (Burger et al. 1995, Palmer and Wellendorf 2007, Seckinger et al. 2008, Holt et al. 2009).

The only other difference in survival we detected was between sites. Bobwhite at Ken had a higher overall survival than Sinclair. Anecdotally, we documented more predation mortalities at Sinclair than Ken (Table A.4). The higher number of identified avian mortalities we documented during the breeding season was similar to results from Sisson et al. (2009). However, they documented lower mammalian predation in the non-breeding season compared to avian predation, which differed from our consistently higher mammalian mortality observations. Rollins and Carroll (2001) suggested that predation is the primary source of bobwhite mortality. As Sinclair had a higher contagion index than Ken, there was less interspersion of vegetation types on Sinclair. This may have resulted in woody escape cover being less available on many parts of Sinclair and, and as a consequence, increased exposure to predation. Janke (2011) observed higher survival related to increases in availability of woody cover in Ohio. He contributed this to a decrease in predation related to the availability of woody escape cover. Likewise, Flock (2006) suggested that lower bobwhite survival on CRP fields was the result of a lack of woody cover. As Roseberry and Sudkamp (1998) found that the contagion index was completely related to edge density (R^2 = 1.00), the lower amount of edge on Sinclair may have resulted in increased exposure to predators. Although we did not measure predator abundance during the scope of our study, we postulate that the higher amount of observed mortalities on Sinclair (as a result of less interspersion) may have accounted for survival rates lower than observed survival rates from previous studies.

However, based on our model testing this hypothesis, survival did not increase with increased interspersion. The influence of vegetation interspersion on survival was minimal based on the model's beta value (β = 0.027, CI = 0.022-0.032) when analyzed individually, and was not an influential factor in survival models from our overall hierarchical analysis, as the effect did not differ from 0. Our contagion index measurements were relatively high (poorer) in relation to suitable conditions for quail densities based on Roseberry and Sudkamp's (1998) results. With regards to edge, there was a linear relationship between edge density and survival, and no evidence of diminishing returns. This may be related to a lack of edge resulting from lower vegetation interspersion. Since interspersion differed among sites, we explored the possibility of differing influences of the best supported vegetation covariates (FOR and OH_CA) to survival between Ken and Sinclair. By assessing interaction models, we found that additive models containing these covariates had more support. This suggested that the effects of forest composition within a home range and the amount of open herbaceous vegetation at a landscape scale on survival were similar between sites, with both covariates having a slightly positive relationship to survival across sites.

Compared to previously published estimates, survival rates at Ken ($S_{\text{pooled}} = 0.316$) were consistent, but those on Sinclair ($S_{\text{pooled}} = 0.141$) were relatively low. Burger et al. (1995) estimated non-breeding survival at 0.159 (SE = 0.008) and breeding survival at 0.332 (SE = 0.027) in northern Missouri. In western Tennessee, Seckinger et al. (2008) reported pooled non-breeding survival estimates of 0.38 (SE = 0.02), while Holt et al. (2009) reported large annual variation in non-breeding survival rates in Mississippi, having 0.06 (SE = 0.019) survival the first year and 0.465 (SE = 0.110) survival the second year. Sisson et al. (2009) reported breeding season survival estimates averaged 0.352 (SE = 0.013) during a 13-year study in southern Georgia and eastern Alabama, USA.

In contrast to the findings of Seckinger et al. (2008), who reported an increase in bobwhite survival after removal of closed-canopy forest vegetation, our results suggest a weak positive relationship between survival and the amount of forest vegetation within a home range. These results appear to conflict with long-held knowledge regarding bobwhite habitat requirements (Rosene 1969). However, forest vegetation on Peabody WMA was established during reclamation and was not typical of forests in the Central Hardwoods Conservation Region. Forests on our study area rarely had canopy closure and, as a result, were more similar to open-canopy woodlands with an understory that provided woody escape cover and food. Increasing the availability of woody cover has been suggested as a means for increasing survival of bobwhites, primarily during the non-breeding season (Yoho and Dimmick 1972, Roseberry and Klimstra 1984, Williams et al. 2000). This importance of escape cover in relation to habitat quality and reduced mortality has been suggested by Roseberry and Klimstra (1984). As forest vegetation was able to provide woody escape cover and food sources, habitat security may have been higher in forest vegetation compared to our other vegetation types.

Our top landscape-scale model included the additive effects of forest vegetation (home range) and open herbaceous core area (landscape). This suggests that survival increased when woody cover was available at a local scale within the context of a large, open herbaceous matrix. Although open herbaceous vegetation consisted primarily of non-native species, the structure provided by this vegetation may explain its influence on survival. As described by Kopp et al. (1998), ideal bobwhite habitat consists of multiple components, including exposure to bare ground and canopy coverage of herbaceous vegetation. The structure provided by species within the Open Herbaceous vegetation type on our study site exhibited the presence of these two components, and apparently provided usable cover. The availability of woody cover in large open areas such as our Open Herbaceous vegetation type helps determine the suitability of this vegetation for bobwhite habitat use (Guthery 1999). By having forest vegetation (woody escape cover) adjacent or near large areas of open vegetation, usable space (Guthery1997) and habitat suitability for bobwhites may have increased on our study area. This may explain why an increase in survival was associated with increased forest coverage (home range) and open herbaceous coverage (landscape).

Models containing landscape-scale metrics were ambiguous, having similar weights among the highest competing models. Similarly, the beta estimates for landscape metrics were low and all confidence intervals contained 0. Although the top model contained the Open Herbaceous Core Area variable, the influence of this variable on survival was not different from 0. Instead, group, year, site, and season were all shown to have more influence on survival than any landscape metric. Likewise, a temporal (weekly) interaction effect between Groups had the most influence on survival, suggesting variation in survival was most related to weekly temporal changes. In general, survival was shown to decrease as weeks progressed through the breeding and non-breeding season (Figure A.1). A strong relationship between survival and a temporal effect has been documented by others (Terhune et al. 2007, Brinkley 2011, Janke 2011). Change in habitat suitability, habitat management, and predator abundance through seasons has been suggested to explain variation in survival through time. Since habitat management was similar on both areas throughout the scope of our study, change in predator abundance may have been a driving factor in explaining this temporal relationship. Although similar studies have identified the importance of multi-scale habitat effects in both the breeding (Taylor et al. 1999) and non-breeding (Seckinger et al. 2008, Janke et al. 2011) seasons, we saw no evidence that such dynamics were operative on our study area. Our results suggest that though time, site, seasonal, and temporal variation may be the main driving factors in survival of bobwhite on reclaimed mined lands, the amount of woody escape cover and the presence of open vegetation adjacent to a bird's home range can influence survival across spatial scales.

MANAGEMENT IMPLICATIONS

Although variation from temporal and site effects are often unavoidable, management of reclaimed mined lands should focus on providing woody escape cover available throughout large areas of open herbaceous vegetation that often occur on reclaimed land. Native species that provide desirable structure and can replace sericea lespedeza should be promoted within open herbaceous areas on reclaimed mined lands. We suggest habitat management on reclaimed mined lands should focus at a local scale. Forested areas should be managed to maintain woody escape cover in the understory while preventing canopy closure and natural succession to proceed. This may be the most effective way to manage large tracts of open vegetation that is released during the mine reclamation process.

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LITERATURE CITED

Beckerle, L. T. 2004. Techniques for encouraging native plants and creating bobwhite quail habitat on drastically disturbed land. Proceedings of the National Meeting of American Society of Mining and Reclamation 21:127.

Brady, S. J., C. H. Flather, K. E. Church, and E. W. Schenck. 1993. Correlates of northern bobwhite distribution and abundance with land use characteristics in Kansas. Pages 115-125 *in* K. Church and T. Dailey, editors. Quail III: National Quail Symposium. Kansas Department of Wildlife and Parks, Pratt, Kansas, USA.

Brennan, L. A. 1991. How can we reverse the northern bobwhite population decline? Wildlife Society Bulletin 19: 544-555.

Burger, Jr., L. W., T. V. Dailey, E. W. Kurzejeski, M. R. Ryan. 1995. Survival and cause-specific mortality of northern bobwhite in Missouri. The Journal of Wildlife Management 59(2): 401-410.

Burnham, K. P., and D. R. Anderson. 2002. Model selection and inference: a practical information-theoretic approach. Springer-Verlag, New York, New York, USA.

Curtis, P.D., B. S. Mueller, P. D. Doerr, and C. F. Robinette. 1988. Seasonal survival of radio-marked northern bobwhite quail from hunted and non-hunted populations. Biotelemetry X: International Radio-telemetry Symposium 10:263-275.

DeMaso, S. J., F. S. Guthery, G. S. Spears, S. M. Rice. 1992. Morning covey calls as an index of northern bobwhite density. Wildlife Society Bulletin 20: 94-101.

DeVos, T., and B. S. Mueller. 1993. Reproductive ecology of northern bobwhite in north Florida. Proceedings of the National Quail Symposium 3: 83-90.

Dimmick, R. W., M. J. Gudlin, and D. F. McKenzie. 2002 The northern bobwhite conservation initiative. Miscellaneous Publication of the Southeastern Association of Fish and Wildlife Agencies, South Carolina, USA.

Duren, K. R., J. J. Buler, W. Jones, and C. K. Williams. 2011. An improved multi-scale approach to modeling habitat occupancy of northern bobwhite. Journal of Wildlife Management 75:1700-1709.

Eddy, T. A. 1999. Effects of sericea lespedeza infestations on wildlife habitat in Kansas. 61st Midwest Fish and Wildlife Conference, Iowa Conservation Commission, Des Moines, Iowa, USA.

Flock, B. E. 2006. Effects of landscape configuration on Northern Bobwhite in Southeastern Kansas. Dissertation, Kansas State University, Manhattan, KS, USA.

Guthery, F.S. 1986. Beef, brush and bobwhites: Quail Management in Cattle Country. Caesar-Kleberg Wildlife Research Institute, Kingsville, Texas, USA.

Guthery, F. S. 1997. A philosophy of habitat management for northern bobwhites. The Journal of Wildlife Management 61:291-301.

Guthery, F. S. 1999. Slack in the configuration of habitat patches for northern bobwhites. Journal of Wildlife Management 63:245-250

Guthery, F. S., M. C. Green, R. E. Masters, S. J. DeMaso, H. M. Wilson, and F. B. Steubing. 2001. Land cover and bobwhite abundance on Oklahoma farms and ranches. Journal of Wildlife Management 65:838-849.

Guthery, F. S., and J. J. Lusk. 2004. Radiotelemetry studies: are we radio-handicapping northern bobwhites? Wildlife Society Bulletin 32:194-201.

Holt, R. D., L. W. Burger, Jr., B. D. Leopold, D. Godwin. 2009. Over-winter survival of northern bobwhite in relation to landscape composition and structure. Pages 432-446 in S. B. Cederbaum, B. C. Faircloth, T. M. Terhune, J. j. Thompson, J. P. Carroll, eds. Gamebird 2006:Quail VI and Perdix XII. 31 May-4 June 2006. Warnell School of Forestry and Natural Resources, Athens Georgia, USA.

Hooge, P. N., and B. Eichenlaub. 1997. Animal Movement Extension to ArcView, ver. 1.1. Alaska Biological Science Center, U. S. Geological Survey, Anchorage, Alaska, USA.

Janke, A. K. 2011. Survival and habitat use of non-breeding northern bobwhites on private lands in Ohio. Thesis, Ohio State University, Columbus, Ohio, USA.

Kopp, S. D., F. S. Guthery, N. D. Forrester, and W. E. Cohen. 1998. Habitat selection modeling for northern bobwhites on subtropical rangeland. Journal of Wildlife Management 62:884-895.

Kozicky, E. L., R. J. Jessen, G. O. Hendrickson, E. B. Speaker. 1956. Estimation of fall quail populations in Iowa. The Journal of Wildlife Management 20: 97-104.

Lebreton, J. –D., K. P. Burnham, J. Clobert, and D. R. Anderson. 1992. Modeling survival and testing biological hypotheses using marked animals: a unified approach with case studies. Ecological Monographs 62:67-118.

Leopold, A. 1933. Game management. Charles Scribner’s Sons, New York, New York, USA.

Lohr, M. T., B. M. Collins, C. K. Williams, and P. M. Castelli. 2011. Life on the edge: Northern bobwhite ecology at the northern periphery of their range. Journal of Wildlife Management 75:52-60.

McGarigal, K., and B. J. Marks. 1994. FRAGSTATS: spatial analysis program for quantifying landscape structure. Version 2.0. Forest Science Department, Oregon State University, Corvallis, USA.

O’Neill, R. V., J. R. Krummel, R. H. Gardner, G. Sugihara, B. Jackson, D. L. DeAngelis, B. T. Milne, M. G. Turner, B. Zygmunt, S. W. Christensen, V. H. Dale, and R. L. Graham. 1988. Indices of landscape pattern. Landscape Ecology 1(3):153-162.

Ott, R. L. 1993. Introduction to statistical methods and data analysis. Duxbury Press, Belmont, California, USA.

Palmer, W. E., S. D. Wellendorf. 2007. Effect of radiotransmitters on northern bobwhite annual survival. The Journal of Wildlife Management 71: 1281-1287.

Pollock, K. H., C. T. Moore, W. R. Davidson, F. E. Kellogg, G. L. Doster. 1989. Survival rates of bobwhite quail based on band recovery analyses. The Journal of Wildlife Management 53:1-6.

Powell, L. A. 2007. Approximating variance of demographic parameters using the delta method: a reference for avian biologists. The Condor 109:949-954.

Riddle, J. D., C. E. Moorman, and K. H. Pollock. 2008. A comparison of methods for estimating northern bobwhite covey detection probabilities. The Journal of Wildlife Management 72: 1437-1442.

Rollins, D., and J. P. Carroll. 2001. Impacts of predation on northern bobwhite and scaled quail. Wildlife Society Bulletin. 29:39-51.

Roseberry, J. L. 1982. Bobwhite. Pages 68-70 in D. E. Davis, editor. Handbook of census methods for terrestrial vertebrates. CRC Press, Boca Raton, Florida, USA.

Roseberry, J. L. 1993. Bobwhite and the "new" biology. Pages 16-20 *in* K. E. Church and T. V. Dailey, editors. Quail III: National Quail Symposium, 14-17 July 1992. Kansas Department of Wildlife and Parks, Pratt, Kansas, USA.

Roseberry, J. L., W. D. Klimstra. 1984. Population ecology of the bobwhite. Southern Illinois University, Carbondale, USA.

Roseberry, J. L., S. D. Sudkamp. 1998. Assessing the suitability of landscapes for northern bobwhite. Journal of Wildlife Management 62:895-902.

Rusk, J. P., F. Hernández, J. A. Arredondo, F. Hernández, F. C. Bryant, D. G. Hewitt, E. J. Redeker, L. A. Brennan, R. L. Bingham. 2007. An evaluation of survey methods for estimating northern bobwhite abundance in southern Texas. The Journal of Wildlife Management 71: 1336-1343.

Rusk, J. P., J. L. Scott, F. Hernández, F. C. Bryant. 2009. Refining the morning covey-call survey to estimate northern bobwhite abundance. Pages 38-45 in Cederbaum, S. B., Faircloth, B. C., Terhune, T. M., Thompson, J. J., Carroll, J. P., eds. Gamebird 2006: Quail

VI and Perdix XII. 31 May-4 June 2006. Warnell School of Forestry and Natural Resources, Athens, Georgia, USA.

Sauer, J. R., J. E. Hines, J. E. Fallon, K. L. Pardieck, D. J. Ziolkowski, Jr., and W. A. Link. 2011. The North American Breeding Bird Survey, Results and Analysis 1966-2009. Version 3.23. 2011. USGS Patuxent Wildlife Research Center, Laurel, Maryland, USA.

Seaman, D. E., J. J. Millspaugh, B. J. Kernohan, G. C. Brundige. 1999. Effects of sample size on kernel home range estimates. The Journal of Wildlife Management 63(2):739-747.

Seckinger, E. M., L. W. Burger, Jr., R. Whittington, A. Houston, R. Carlisle. 2008. Effects of landscape composition on winter survival of northern bobwhites. The Journal of Wildlife Management 72(4):959-969.

Sisson, D. C., T. M. Terhune, H. L. Stribling, J. F. Sholar, S. D. Mitchell. 2009. Survival and causes of mortality for northern bobwhites in the southeastern USA. Pages 467-478 in Cederbaum, S. B., B. C. Faircloth, T. M. Terhune, J. J. Thompson, J. P. Carroll, eds. Gamebird 2006: Quail VI and Perdix XII. 31 May- 4 June 2006. Warnell School of Forestry and Natural Resources, Athens, Georgia, USA.

Stoddard, H. L. 1931. The bobwhite quail: its habits, preservation and increase. Charles Scibner’s Sons, New York, New York, USA.

Stokes, A. W. 1967. Behavior of the bobwhite, (*Colinus virginianus*). The Auk 84: 1-33.

Taylor, J. S., K. E. Church, D. H. Rusch, J. R. Cary. 1999. Macrohabitat effects on summer survival, movements, and clutch success of northern bobwhite in Kansas. The Journal of Wildlife Management 63(2):675-685.

Terhune, T. M., D. C. Sisson, J. B. Grand, H. L. Stribling. 2007. Factors influencing survival of radiotagged and banded northern bobwhites in Georgia. The Journal of Wildlife Management 71: 1288-1297.

Twedt, D. J., R. R. Wilson, A. S. Keister. 2006. Spatial models of northern bobwhite populations for conservation planning. The Journal of Wildlife Management 71: 1808-1818.

Wellendorf, S. D., W. E. Palmer, P. T. Bromley. 2004. Estimating call rates of northern bobwhite coveys and censusing population. The Journal of Wildlife Management 68: 672-682.

White, G. C., and K. P. Burnham. 1999. Program MARK: survival estimation from populations of marked animals. Bird Study 46 (suppl): S120-S139.

White, G. C., and R. A. Garrot. 1990. Analysis of wildlife radio-tracking data. Academic Press, Inc. San Diego, CA, USA.

Williams, C. K., R. S. Lutz, R. D. Applegate, D. H. Rusch. 2000. Habitat use and survival of northern bobwhite (*Colinus virginianus*) in cropland and rangeland ecosystems during the hunting season. Canadian Journal of Zoology 78:1562-1566.

Williams, C. K., R. S. Lutz, R. D. Applegate. 2004. Winter survival and additive harvest in northern bobwhite coveys in Kansas. The Journal of Wildlife Management 68:94-100.

Worton, B. J. 1989. Kernel methods for estimating the utilization distribution in home-range studies. Ecology 70:164-168.

Yoho, N. S., and R. W. Dimmick. 1972. Habitat utilization by bobwhite quail during winter. Pages 90-99 *in* J. A. Morrison and J. C. Lewis, editors. Proceedings of the First National Bobwhite Quail Symposium. Oklahoma State University, Research Foundation, Stillwater, Oklahoma, USA.

APPENDIX A

Table A.1: Delineated vegetation types and total coverage (ha) on Peabody WMA, Ohio and Muhlenberg Counties, Kentucky, USA, 2009–2011.

Site	Vegetation	Hectares	Total (ha)
Ken	Annual Grain	25.8	1853.1
	Forest Deciduous	405.7	
	Native Warm-Season Grass	205.7	
	Open Herbaceous	524.4	
	Scrub Shrub	519.4	
	Water	165.4	
	Wetland Emergent	6.8	
Sinclair	Annual Grain	6.5	1470.6
	Forest Deciduous	327.2	
	Native Warm-Season Grass	58.0	
	Open Herbaceous	671.6	
	Scrub Shrub	321.7	
	Water	69.7	
	Wetland Emergent	16.0	

Table A.2. List and description of class, home range, and landscape metrics used to assess effects on survival of radio-marked northern bobwhite on Peabody WMA, Ohio and Muhlenberg Counties, Kentucky, USA, 1 Sep 2009-30 Sep 2011.

Metric	Scale	Description
t	Group	Time
T	Group	Linear time
W	Group	Weekly time
wi	Group	Weekly time interaction
g	Group	Group: birds with habitat metrics and birds without habitat metrics
year	Group	Year
sex	Group	Sex, either male or female
age	Group	Age, either juvenile or adult
weight	Group	Weight of bird
site	Group	Site, either Ken or Sinclair
season	Group	Season, either breeding or non-breeding
FOR	Home Range	% forest vegetation within a home range
SS	Home Range	% scrub-shrub vegetation within a home range
NWSG	Home Range	% NWSG vegetation within a home range
OH	Home Range	% OH vegetation within a home range
HRS	Home Range	Home range size
ED_FOR	Landscape	Forest to open vegetation edge density
ED_SS	Landscape	Scrub-shrub to open vegetation edge density
FOR_CA	Landscape	Core area of forest vegetation
SS_CA	Landscape	Core area of scrub-shrub vegetation
NWSG_CA	Landscape	Core area of NWSG vegetation
OH_CA	Landscape	Core are of OH vegetation
CI	Landscape	Contagion index

Table A.3. Non-breeding (1 Oct-31Mar) and breeding (1 Apr-30 Sep) season survival (*S*) estimates of radio-marked male, female, juvenile, and adult northern bobwhite by site on Peabody WMA, Ohio and Muhlenberg Counties, Kentucky, USA, 1 Sep 2009-30 Sep 2011.

		Pooled			Male			Female			Juvenile			Adult		
Site	Season	*n*	*S*	SE	*n*	*S*	SE	*n*	*S*	SE	*n*	*S*	SE	*n*	*S*	SE
Sinclair	Non-breeding	155	0.142	0.025	86	0.124	0.026	69	0.176	0.036	127	0.142	0.026	28	0.143	0.043
	Breeding	127	0.138	0.030	79	0.118	0.030	48	0.169	0.039	89	0.138	0.031	38	0.139	0.042
	Pooled	282	0.141	0.022	165	0.122	0.023	117	0.173	0.033	216	0.141	0.023	66	0.141	0.040
Ken	Non-breeding	209	0.318	0.030	111	0.289	0.034	98	0.355	0.039	165	0.317	0.033	44	0.319	0.049
	Breeding	128	0.312	0.039	80	0.280	0.043	44	0.346	0.045	90	0.312	0.043	38	0.313	0.051
	Pooled	337	0.316	0.027	191	0.286	0.032	142	0.352	0.036	255	0.316	0.030	82	0.316	0.045

Table A.4. Non-breeding (1 Oct-31Mar) and breeding (1 Apr-30 Sept) causes of mortality for 467 radio-marked northern bobwhite by site on Peabody WMA, Ohio and Muhlenberg Counties, Kentucky, USA, 1 Sep 2009-30 Sep 2011.

		Causes of mortality						
Site	Season	Avian	Mammal	Harvest	Investigation	Other	Unknown	Total
Sinclair	Non-breeding	43	68	0	3	0	23	137
	Breeding	16	10	0	6	3	42	77
Ken	Non-breeding	10	54	0	4	1	96	165
	Breeding	8	6	0	4	13	57	88

Table A.5. Highest ranking models from 61 *a priori* models based on ΔAIC$_c$ values and AIC$_c$ weights used to assess the influence of class, home range, and landscape metrics on northern bobwhite survival on Peabody WMA, Ohio and Muhlenberg Counties, Kentucky, USA, 1 Sep 2009-30 Sep 2011[a].

Model	AIC$_c$	ΔAIC$_c$	AIC$_c$ Weights	Model Likelihood	*k*	Deviance
{wi+g+year+site+season+FOR+OH_CA}	3642.2111	0	0.13664	1	57	3528.057
{wi+g+year+site+season+FOR}	3642.7004	0.4893	0.10699	0.783	56	3530.552
{wi+g+year+site+season+FOR+NWSG_CA}	3643.2935	1.0824	0.07953	0.582	57	3529.14
{wi+g+year+site+season+FOR+NWSG}	3643.5731	1.362	0.06915	0.5061	57	3529.419
{wi+g+year+site+season+FOR+ED_SS^2}	3643.7167	1.5056	0.06436	0.471	57	3529.563
{wi+g+year+site+season+FOR+HRS}	3643.9411	1.73	0.05753	0.421	57	3529.787
{wi+g+year+site+season+FOR+CI}	3644.1182	1.9071	0.05266	0.3854	57	3529.964
{wi+g+year+site+season+FOR+SS_CA}	3644.1507	1.9396	0.05181	0.3792	57	3529.997
{wi+g+year+site+season+FOR+FOR_CA}	3644.168	1.9569	0.05136	0.3759	57	3530.014
{wi+g+year+site+season+FOR+ED_SS}	3644.1973	1.9862	0.05062	0.3705	57	3530.044
{wi+g+year+site+season+FOR+ED_FOR^2}	3644.4575	2.2464	0.04444	0.3252	57	3530.304
{wi+g+year+site+season+FOR+ED_FOR}	3644.5544	2.3433	0.04234	0.3099	57	3530.401
{wi+g+year+site+season+NWSG}	3644.5799	2.3688	0.0418	0.3059	56	3532.431
{wi+g+year+site+season}	3644.8141	2.603	0.03718	0.2721	55	3534.671
{wi+g+year+site+season+HRS}	3645.3728	3.1617	0.02812	0.2058	56	3533.224
{wi+g+year+site+season+(FOR x NWSG)}	3645.8608	3.6497	0.02203	0.1612	56	3533.712
{wi+g+year+site+season+sex}	3646.453	4.2419	0.01639	0.12	56	3534.305
{wi+g+year+site+season+SS}	3646.5849	4.3738	0.01534	0.1123	56	3534.436
{wi+g+year+site+season+OH}	3646.7392	4.5281	0.0142	0.1039	56	3534.591
{wi+g+year+site+season+age}	3646.8127	4.6016	0.01369	0.1002	56	3534.664
{wi+g+year+site}	3650.154	7.9429	0.00258	0.0189	54	3542.016
{wi+g+year+site+sex}	3651.6093	9.3982	0.00124	0.0091	55	3541.466

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

Table A.6. Importance weights for parameters used to assess the influence of class, home range, and landscape metrics on northern bobwhite survival on Peabody WMA, Ohio and Muhlenberg Counties, Kentucky, USA, 1 Sep 2009-30 Sep 2011

Parameter	Number of candidate models	Importance weight[a]
G	48	1
Site	39	1
Year	35	1
Wi	30	1
Season	30	0.99618
FOR	12	0.80743
OH_CA	1	0.13664
ED_SS	2	0.11498
NWSG	2	0.11095
ED_FOR	2	0.08678
NWSG_CA	1	0.07953
CI	1	0.05266
SS_CA	1	0.05181
FOR_CA	1	0.05136
(FOR x NWSG)	1	0.02203
Sex	9	0.01763
SS	2	0.01534
Age	8	0.0142
OH	1	0.0142
Weight	5	0
HRS	2	0
t	1	0
T	1	0
W	1	0

[a] Importance weight of a parameter is estimated as the sum of Akaike weights from candidate models containing the parameter.

Table A.7. Summary of model-selection results from hypothesis of the effects of the contagion index to survival of northern bobwhite on Peabody WMA, Ohio and Muhlenberg Counties, Kentucky, USA, 1 Sep 2009-30 Sept 2011.

Model	AIC_c	ΔAIC_c	AIC_c Weights	Model Likelihood	*k*	Deviance
CI	3764.26	0	1	1	2	3760.26
Null	3903.42	139.153	0	0	1	3901.42

Table A.8. Summary of model-selection results from hypothesis relating edge density to survival of northern bobwhite on Peabody WMA, Ohio and Muhlenberg Counties, Kentucky, USA, 1 Sep 2009-30 Sept 2011.

Model	AIC$_c$	ΔAIC$_c$	AIC$_c$ Weights	Model Likelihood	*k*	Deviance
ED_SS	3778.944	0	1	1	2	3774.944
ED_SS2	3810.346	31.4018	0	0	2	3806.345
ED_FOR	3839.942	60.9981	0	0	2	3835.942
ED_FOR2	3873.605	94.6612	0	0	2	3869.605
Null	3903.415	124.4715	0	0	1	3901.415

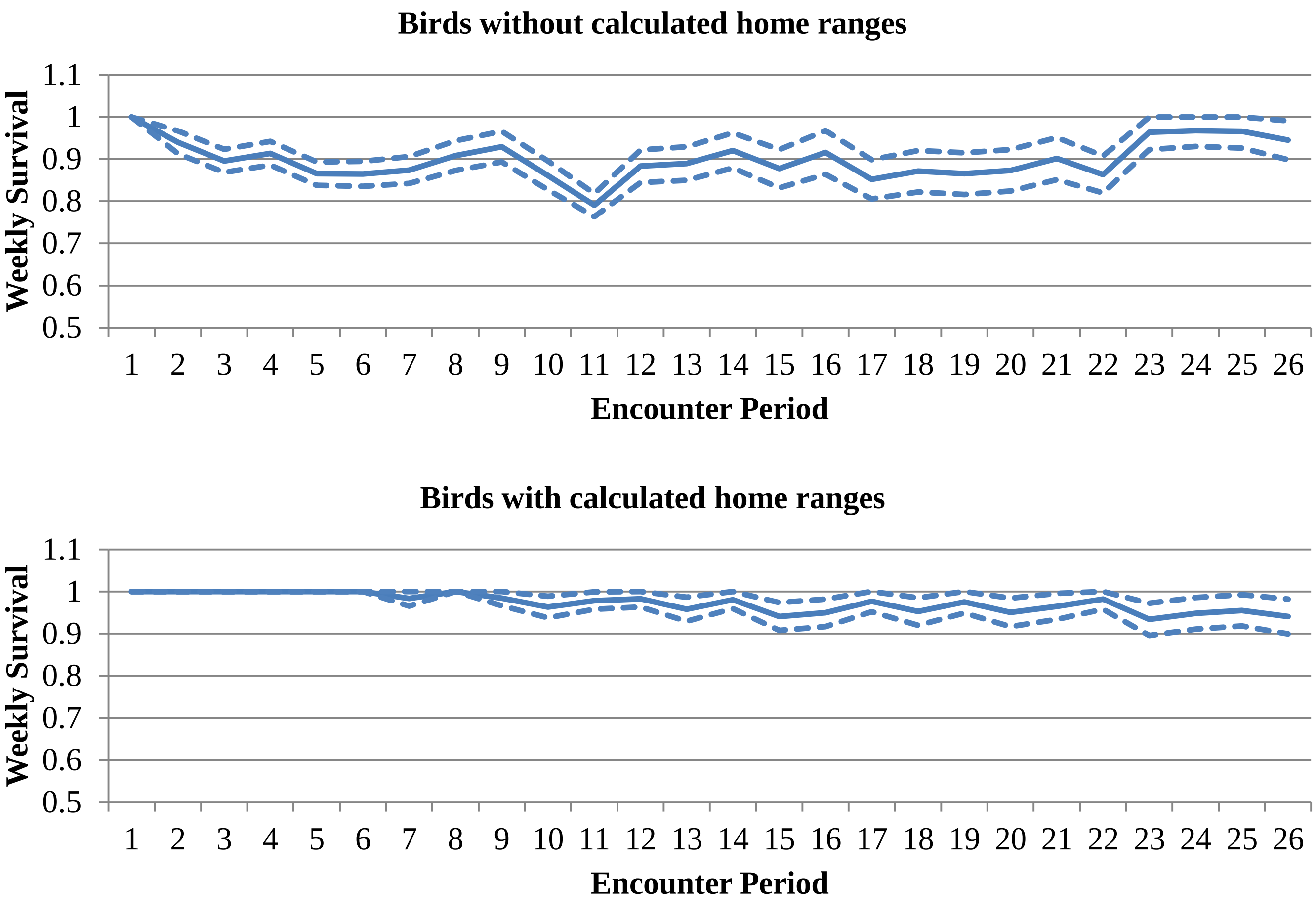

Figure A.1. Encounter period survival rate estimates (solid line) and 95% confidence intervals (dotted lines) for each group of northern bobwhite on Peabody WMA, Ohio and Muhlenberg Counties, Kentucky, USA, 1 Sep 2009-30 Sep 2011.

Table A.9. Age and sex structure of captured northern bobwhite by site and season on Peabody WMA, Ohio and Muhlenberg Counties, Kentucky, USA, 1 Sep 2009-30 Sep 2011.

		Sex			Age	
Site	Season	Males	Females	Unknown	Adult	Juvenile
Sinclair	Non-breeding	129	95	0	29	195
	Breeding	96	50	22	39	129
Ken	Non-breeding	117	115	0	55	177
	Breeding	115	66	36	44	173

Table A.10. Summary of northern bobwhite trapping success (number of birds caught divided by number of trap nights) by site and season on Peabody WMA, Ohio and Muhlenberg Counties, Kentucky, USA, 1 Sep 2009-30 Sep 2011.

Site	Season		Trapping Success (%)
Sinclair	Winter		4.2
	Summer		2.5
Ken	Winter		4.1
	Summer		3.0

Table A.11. Summary of home range and FRAGSTATS landscape habitat metrics, t test statistics (α = 0.05), and probability values (*P*)[a] used to evaluate survival of northern bobwhite during the non-breeding seasons on Peabody WMA, Ohio and Muhlenberg Counties, Kentucky, USA, 1 Sep 2009-30 Sep 2011.

Parameter	Site	Scale	$\bar{x}$	SE	*t*	*P*
NWSG (%)	Ken	Home Range	17.4	1.7	-7.178	< **0.001**
	Sinclair		2.9	0.6		
OH (%)	Ken		34.6	2.6	4.972	< **0.001**
	Sinclair		53.2	2.6		
Scrub-Shrub (%)	Ken		36.7	2.4	-2.622	**0.010**
	Sinclair		28.0	2.1		
Forest (%)	Ken		5.8	1.0	3.062	**0.003**
	Sinclair		13.2	2.4		
Forest and Open Edge Density (m/ha)	Ken	Landscape	1.0	0.1	4.180	< **0.001**
	Sinclair		2.1	0.2		
Scrub-shrub and Open Edge Density (m/ha)	Ken		33.8	1.7	-3.179	**0.002**
	Sinclair		26.8	1.2		
Forest Core Area (ha)	Ken		1.2	0.2	1.487	0.140
	Sinclair		1.6	0.2		
Scrub-shrub Core Area (ha)	Ken		3.7	0.2	-6.382	< **0.001**
	Sinclair		2.4	0.1		
NWSG Core Area (ha)	Ken		2.0	0.2	-6.855	< **0.001**
	Sinclair		0.5	0.1		
Open Herbaceous Core Area (ha)	Ken		3.2	0.2	8.234	< **0.001**
	Sinclair		5.6	0.2		
Contagion Index (%)	Ken		48.4	1.0	2.395	**0.018**
	Sinclair		52.5	1.4		

[a] Bolded *P* values indicate statistically different variables between sites.

Table A.12. Summary of home range and FRAGSTATS landscape habitat metrics, t test statistics ($\alpha = 0.05$), and probability values (*P*)[a] used to evaluate survival of northern bobwhite during the breeding seasons on Peabody WMA, Ohio and Muhlenberg Counties, Kentucky, USA, 1 Sep 2009-30 Sep 2011.

Parameter	Site	Scale	$\bar{x}$	SE	*t*	*P*
NWSG (%)	Ken	Home Range	18.1	1.7	-5.108	< **0.001**
	Sinclair		6.3	1.1		
OH (%)	Ken		34.7	2.6	9.103	< **0.001**
	Sinclair		70.6	2.7		
Scrub-Shrub (%)	Ken		32.9	2.4	-5.438	< **0.001**
	Sinclair		15.5	1.4		
Forest (%)	Ken		8.7	1.2	-3.001	**0.003**
	Sinclair		3.4	1.2		
Forest and Open Edge Density (m/ha)	Ken	Landscape	1.3	0.2	-0.263	0.793
	Sinclair		1.3	0.2		
Scrub-shrub and Open Edge Density (m/ha)	Ken		30.7	1.7	-2.173	**0.032**
	Sinclair		25.4	1.3		
Forest Core Area (ha)	Ken		1.5	0.2	-3.017	**0.003**
	Sinclair		0.8	0.2		
Scrub-shrub Core Area (ha)	Ken		3.2	0.2	-5.456	< **0.001**
	Sinclair		1.9	0.1		
NWSG Core Area (ha)	Ken		1.9	0.2	-6.712	< **0.001**
	Sinclair		0.5	0.1		
Open Herbaceous Core Area (ha)	Ken		3.3	0.2	11.970	< **0.001**
	Sinclair		7.1	0.2		
Contagion Index (%)	Ken		48.3	1.3	4.133	< **0.001**
	Sinclair		56.8	1.5		

[a] Bolded *P* values indicate statistically different variables between sites.

Table A.13. Summary of home range 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 2011.

		Home Range Size (ha)		Home range buffer size (ha)	
Site	Season	x̄	SE	x̄	SE
Sinclair	Non-breeding	22.6	2.5	95.7	8.5
	Breeding	35.3	7.1	136.5	8.3
Ken	Non-breeding	26.9	2.1	107.1	6.4
	Breeding	38.9	5.0	137.5	14.1

Table A.14. Ranking of all 61 *a priori* models based on ΔAIC_c values and AIC_c weights used to assess the influence of class, home range, and landscape metrics on northern bobwhite survival on Peabody WMA, Ohio and Muhlenberg Counties, Kentucky, USA, 1 Sep 2009-30 Sept 2011[a].

Model	AIC_c	ΔAIC_c	AIC_c Weights	Model Likelihood	*k*	Deviance
{wi+g+year+site+season+FOR+OH_CA}	3642.2111	0	0.13664	1	57	3528.0573
{wi+g+year+site+season+FOR}	3642.7004	0.4893	0.10699	0.783	56	3530.5519
{wi+g+year+site+season+FOR+NWSG_CA}	3643.2935	1.0824	0.07953	0.582	57	3529.1397
{wi+g+year+site+season+FOR+NWSG}	3643.5731	1.362	0.06915	0.5061	57	3529.4193
{wi+g+year+site+season+FOR+ED_SS2}	3643.7167	1.5056	0.06436	0.471	57	3529.5629
{wi+g+year+site+season+FOR+HRS}	3643.9411	1.73	0.05753	0.421	57	3529.7873
{wi+g+year+site+season+FOR+CI}	3644.1182	1.9071	0.05266	0.3854	57	3529.9644
{wi+g+year+site+season+FOR+SS_CA}	3644.1507	1.9396	0.05181	0.3792	57	3529.9969
{wi+g+year+site+season+FOR+FOR_CA}	3644.168	1.9569	0.05136	0.3759	57	3530.0142
{wi+g+year+site+season+FOR+ED_SS}	3644.1973	1.9862	0.05062	0.3705	57	3530.0435
{wi+g+year+site+season+FOR+ED_FOR2}	3644.4575	2.2464	0.04444	0.3252	57	3530.3037
{wi+g+year+site+season+FOR+ED_FOR}	3644.5544	2.3433	0.04234	0.3099	57	3530.4006
{wi+g+year+site+season+NWSG}	3644.5799	2.3688	0.0418	0.3059	56	3532.4314
{wi+g+year+site+season}	3644.8141	2.603	0.03718	0.2721	55	3534.6708
{wi+g+year+site+season+HRS}	3645.3728	3.1617	0.02812	0.2058	56	3533.2243
{wi+g+year+site+season+(FOR x NWSG)}	3645.8608	3.6497	0.02203	0.1612	56	3533.7123
{wi+g+year+site+season+sex}	3646.453	4.2419	0.01639	0.12	56	3534.3045
{wi+g+year+site+season+SS}	3646.5849	4.3738	0.01534	0.1123	56	3534.4364
{wi+g+year+site+season+OH}	3646.7392	4.5281	0.0142	0.1039	56	3534.5907
{wi+g+year+site+season+age}	3646.8127	4.6016	0.01369	0.1002	56	3534.6642
{wi+g+year+site}	3650.154	7.9429	0.00258	0.0189	54	3542.0158
{wi+g+year+site+sex}	3651.6093	9.3982	0.00124	0.0091	55	3541.466

Continued

Table A.14 Continued

Model	AIC$_c$	ΔAIC$_c$	AIC$_c$ Weights	Model Likelihood	*k*	Deviance
{wi+g+year}	3672.2219	30.0108	0	0	53	3566.089
{wi+g+site}	3684.8729	42.6618	0	0	53	3578.74
{g+year+site+season}	3695.9529	53.7418	0	0	5	3685.952
{g+year+site}	3697.3419	55.1308	0	0	4	3689.341
{g+year+site+season+sex}	3697.5818	55.3707	0	0	6	3685.58
{g+year+site+season+weight}	3697.8567	55.6456	0	0	6	3685.855
{g+year+site+season+age}	3697.9433	55.7322	0	0	6	3685.941
{g+year+site+sex}	3698.8414	56.6303	0	0	5	3688.84
{g+year+site+weight}	3699.2768	57.0657	0	0	5	3689.275
{g+year+site+age}	3699.3227	57.1116	0	0	5	3689.321
{wi+g+weight}	3707.4965	65.2854	0	0	45	3617.4
{wi+g+sex}	3708.1768	65.9657	0	0	45	3618.081
{wi+g+season}	3717.613	75.4019	0	0	53	3611.48
{g+year}	3721.015	78.8039	0	0	3	3715.014
{g+site}	3721.9816	79.7705	0	0	3	3715.981
{wi}	3722.2726	80.0615	0	0	52	3618.144
{wi+g+year}	3723.5643	81.3532	0	0	53	3617.431
{wi+g+age}	3724.2039	81.9928	0	0	53	3618.071
{W}	3755.026	112.8149	0	0	26	3702.993
{g+season}	3757.0176	114.8065	0	0	3	3751.017
{g}	3758.7241	116.513	0	0	2	3754.724
{g+sex}	3758.7857	116.5746	0	0	3	3752.785
{g+weight}	3760.3669	118.1558	0	0	3	3754.366
{g+age}	3760.3981	118.187	0	0	3	3754.398
{site+year}	3877.1281	234.917	0	0	3	3871.128
{site+sex}	3881.9839	239.7728	0	0	3	3875.983
{site}	3882.5902	240.3791	0	0	2	3878.59
{site+season+sex	3883.9418	241.7307	0	0	4	3875.941
{site+season}	3884.5741	242.363	0	0	3	3878.574
{site+age}	3884.5904	242.3793	0	0	3	3878.59
{site+season+age}	3886.5735	244.3624	0	0	4	3878.573

Continued

Table A.14 Continued

Model	AIC_c	ΔAIC_c	AIC_c Weights	Model Likelihood	*k*	Deviance
{site+season+age}	3886.5735	244.3624	0	0	4	3878.573
{year}	3893.6637	251.4526	0	0	2	3889.663
{sex}	3901.5623	259.3512	0	0	2	3897.562
{null}	3903.4153	261.2042	0	0	1	3901.415
{weight}	3904.7085	262.4974	0	0	2	3900.708
{age}	3904.7562	262.5451	0	0	2	3900.756
{season}	3905.3187	263.1076	0	0	2	3901.318
{T}	3905.3708	263.1597	0	0	2	3901.371

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

PART III

MULTI-SCALE FACTORS AFFECTING NESTING ECOLOGY OF NORTHERN BOBWHITE ON RECLAIMED MINED LAND

ABSTRACT Large tracts of early successional vegetation are being created throughout much of the northern bobwhite (*Colinus virginianus*) species' range under the auspices of the Surface Mining Control and Reclamation Act of 1977 (SMCRA). Monitoring populations and understanding limiting factors on reclaimed mined lands is essential if these vast tracts are to be managed successfully for northern bobwhite. A potential limiting factor on these areas is reproductive success. To better understand bobwhite reproduction on reclaimed mined lands, we used radio telemetry on Peabody WMA, a 3,330 ha reclaimed surface mine in western Kentucky, to monitor nesting northern bobwhite. During 2010- 2011 we captured northern bobwhite (n = 385) using baited funnel traps and monitored them (n = 210 fitted with necklace-style radio-collars) during the breeding (1 Apr-30 Sep) season. We located 57 nests, of which 47.4% were successful and 52.6% were unsuccessful. We used the nest survival model in Program MARK to estimate daily nest survival rates from 20 *a priori* models at 2 spatial scales: micro-habitat and landscape. Daily nest survival rate (DSR) was 0.951 (SE = 0.010) and nest survival (beginning at the onset of incubation) was 0.317 (SE = 0.081). Nest age was the most influential factor for nest survival (β = 0.17, CI = 0.07-0.26). We found no evidence that landscape metrics or vegetation composition within a 210-m nest buffer influenced nest survival. Distance to unvegetated bare ground was included in the top model, but had a minimal effect on nest survival (β = 0.82, CI = -0.07-1.72) at the micro-habitat scale. DSR was higher for nesting substrate comprised of NWSG (S = 0.95, SE = 0.01) than sericea lespedeza (S = 0.94, SE = 0.01). Our results suggest reclaimed mined lands can sustain successful breeding efforts of northern bobwhite. Management should focus on decreasing sericea lespedeza coverage and increasing native grass coverage to increase nesting success.

With northern bobwhite (*Colinus virginianus*) (hereafter, bobwhite) experiencing a range-wide, 3.8% annual decline from 1966-2009 (Sauer et al. 2011), understanding limiting factors related to population declines is critical. As with many ground-nesting species, bobwhite experience high nesting losses (Martin 1993, Rollins and Carrol 2001), which may equate to lower densities, especially in isolated populations (Errington and Stoddard 1938, Roseberry and Klimstra 1984). Low nesting success and a lack of vegetation suitable for nesting successfully has limited bobwhite populations (Rosene 1969, Dimmick et al. 2002).

Although scale-dependent habitat metrics have been linked to bobwhite survival (Seckinger et al. 2008, Holt et al. 2009, Janke et al. 2011), little research has addressed multi-scale habitat effects on bobwhite nest survival (Taylor et al. 1999*a*, Potter et al. 2011). In Kansas, micro- and macro-habitat characteristics were assessed in relation to nest-site selection and nest survival (Taylor et al. 1999*a*, Taylor et al. 1999*b*). Successful nests had more native grass hayfields surrounding them at a landscape scale, but had less coverage of native grass at a local scale. At the micro-habitat scale, successful nests were mainly associated with areas containing less shrub cover, taller vegetation, and less litter. In Iowa, Potter et al. (2011) observed no evidence of multi-scale habitat influences on nest success. The percentage of forb canopy cover positively influenced nest success at the nest site level, but this influence was minimal based on its beta value and was only documented on one of two sites studied.

Reclaimed mined lands offer a unique opportunity to increase habitat on a large scale for bobwhite in many areas of the eastern United States. Large tracts of early successional vegetation are often created under the auspices of the Surface Mining Control and Reclamation Act of 1977 (SMCRA). However, these lands often are re-vegetated with plant species that may not provide suitable structure for nesting bobwhite, such as sericea lespedeza (*Lespedeza* *cuneata*; Eddy 1999). To better understand the effects of vegetation composition and structure for bobwhite reproduction on reclaimed mined lands, research must focus on nest success as a function of these habitat attributes at both local and landscape scales.

Although nesting ecology of bobwhite and multi-scale habitat effects on nest success have been studied, no contemporary research has been conducted within the Central Hardwoods Conservation Region and none has been conducted on reclaimed mined lands. To better understand how vegetation on reclaimed mined lands affected bobwhite nest success, we conducted a large-scale radio telemetry study on Peabody Wildlife Management Area (WMA), Kentucky, USA from 2010-2011. Since much of Peabody WMA was planted in uniform "blocks" of vegetation during the reclamation process, grasses ideal for nesting are not always readily available at the local scale. We hypothesized nest success would be greater with an increase in the distance to bare ground, an increase in native grass coverage at the local scale, and a decrease of deciduous forest coverage at the landscape scale. Our primary goal to was to determine which vegetation attributes contributed to increased nest success on reclaimed mined land, and if there was scale-dependency related to these vegetation attributes. We also wanted to document overall nesting ecology of bobwhite on reclaimed mine land, such as nesting rate, success rate, daily nest survival rate (DSR), and re-nesting rate. To assess the importance of scale on nest success in relation to vegetation attributes, we focused analysis at the micro-habitat and landscape scales.

STUDY AREA

We conducted the study on a reclaimed coal mine, Peabody WMA (3,323 ha) in Muhlenberg (37°14'N, 87°15'W) and Ohio (37°17'N, 86°54'W) counties in western Kentucky, USA. The study area consisted of open herbaceous vegetation (36%; Table B.1), which was dominated by sericea lespedeza (*Lespedeza cuneata*) and annual forbs such as, common ragweed (*Ambrosia artemisiifolia*), sumpweed (*Iva annua*), and goldenrod (*Solidago spp.*). Shrub vegetation (25%) was characterized by an abundance of black locust (*Robinia pseudoacacia*), winged sumac (*Rhus copallinum*), and blackberry (*Rubus spp.*). Deciduous forests (22%) primarily consisted of eastern cottonwood (*Populus deltoides*), green ash (*Fraxinus pennsylvaticum*) and American sycamore (*Platanus occidentalis*), and typically had a well-developed understory consisting of blackberry (*Rubus spp.*) and honeysuckle (*Lonicera japonica* and *Lonicera maakii*). More recently, native warm-season grasses (NWSG), including mixtures of big bluestem (*Andropogon gerardii*), little bluestem (*Schizachyrium scoparium*), indiangrass (*Sorghastrum nutans*), and switchgrass (*Panicum virgatum*), have been established (8%). Small lakes, wetlands, and annual grain food plots comprised the remainder (9%) of our study area. Forests on the WMA were established under guidelines in place prior to the passage of the SMCRA (“pre-law”), while all early successional vegetation was established under post-law criteria. Habitat management on both units include dormant-season (January-March) prescribed fire, disking (all months), herbicide spraying, and plantings of food plots and NWSG. Efforts have focused on maintaining early successional vegetation while trying to limit coverage of invasive, non-native plants (particularly sericea lespedeza) that had been established previously.

We conducted our research on two different sites on Peabody WMA (Ken and Sinclair). These two sites are separated by the Green River and are 18 kilometers apart. As no birds were detected moving between Ken and Sinclair, we considered them separate sites. Vegetation on Sinclair (1470 ha) was 45% open herbaceous, 22% scrub-shrub, 22% forest, and 4% NWSG. The remaining 7% consisted of small lakes, wetlands, and annual grain food plots. Ken (1853 ha) consisted of 28% open herbaceous, 28% scrub-shrub, 22% forest, 11% NWSG. The remaining 11% consisted of small lakes, wetlands, and annual grain food plots.

METHODS

Land Cover

Four major vegetation types (forest, scrub-shrub, open herbaceous, and NWSG) were delineated based on aerial imagery in ArcGIS 9.3 (ESRI, Redlands, CA, USA), which constituted 91% of the total land cover on our study site. To delineate between forest, scrub-shrub, 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 1m x 1m 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, scrub-shrub 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 scrub-shrub, 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 scrub-shrub 9.6 m^2/ha (SE = 1.23); scrub-shrub 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.

Data Collection

As trapping efforts supported a larger radio-telemetry project, we captured bobwhites year-round (Sep 2009 - Sep 2011) using funnel traps (Stoddard 1931), which were covered with burlap and vegetation to help reduce stress and predation of captured birds. We defined the breeding season as 1 Apr-30 Sep, based on Burger et al. (1995*a*). We strategically placed traps (*n* = 120) in areas thought to have birds and where birds were heard or seen. We fitted captured birds with necklace-style collars weighing 6g (crystal-controlled, two-stage design, pulsed by a CMOS multivibrator, American Wildlife Enterprise, Monticello, Florida, USA) based on meeting a minimum body mass requirement (120g) and availability of collars. We assumed radio transmitters did not affect survival (Palmer and Wellendorf 2007, Terhune et al. 2007). We leg-banded (double) all captured birds. We determined sex, age, and weight of all birds, and released birds at their capture site. During the breeding season, we classified the sex of a bird as unknown if we were not able to determine sex because of the bird's age. We determined if a bird was an adult by the absence of a buff-tipped primary covert (Stoddard 1931). Our trapping and handling methods complied with University of Tennessee Institutional Animal Care and Use Committee Permit (no. 2042-0911) protocol.

We attempted to locate radio-marked individuals at least three times/week using a scanning receiver and a handheld Yagi antenna (Advanced Telemetry Systems, Inc., Isanti, MN). We located birds by homing (White and Garrot 1990) within 50m to avoid flushing birds. Once birds were located, we recorded the distance and azimuth to the actual bird location and recorded the Universal Transverse Mercator (UTM) coordinates of the observer on a GPS unit (Garmin GPSMAP 60CSx, Garmin International, Inc. Olathe, KS, USA). We used the distance and azimuth to estimate the location of each bird. We recorded locations of birds at different times on subsequent days to capture the variability of diurnal patterns. We considered birds with identical subsequent locations to be nesting (Burger et al. 1995*b*). We located the actual nest and counted eggs when the radio-marked bird was away from the nest. Once nest location was determined, we recorded UTM coordinates on a GPS unit to the nearest meter. Once a bird was considered to be nesting, we monitored the incubation status daily by locating the radiocollared adult. If incubating adults were located away from the nest, we returned to the actual location of the nest to monitor the clutch (Taylor et al. 1999*a*) every 7-10 days.

For micro-habitat vegetation metrics, we measured distance to bare ground (m) (DtoBG), distance to edge (m) (DtoED), and recorded the vegetation type in which the nest was located and the nest substrate (sericea lespedeza, cool-season grass, or native warm-season grass) within 7 days of nest termination. We considered bare ground to be exposed soil with no vegetative coverage and edge to be where two different delineated vegetation types met. We used ArcGIS 9.3 to create a buffer with a 210 m radius (Taylor et al. 1999*a*, Potter et al. 2011) radius around each nest to account for landscape-scale habitat metrics. Within each buffer we calculated landscape metrics using FRAGSTATS (McGarigal and Marks 1994) with a 150m moving window. We used the 150m moving window to capture variation from adjacent vegetation cells. We included our four major vegetation types within our FRAGSTATS analysis. We also analyzed eight landscape metrics: forest/open vegetation edge density, scrub-shrub/open vegetation edge density, core area of all four major vegetation types (using a 30m edge effect), and a contagion index (Table B.2). In addition, we calculated the percent of each of our four vegetation types within the buffer for vegetation composition covariates at the landscape level (Table B.2).

Reproductive Effort

We estimated nesting rates, success rates, and re-nesting rates for each sex based on the number of birds radio-marked and entering the spring population (Burger et al. 1995*b*) at the beginning of our nesting season, which was 7 May. As noted by Burger et al. (1995*b*), we assumed our estimates of nest success and re-nesting rates were over- and under-estimated, respectively. This is because we were typically not able to detect nesting activity until the beginning of the incubation period. We estimated nesting rate as the percentage of radio-marked birds surviving past 7 May that attempted to incubate $\geq$1 nest. We estimated success rate as the percentage of radio-marked birds surviving past 7 May that successfully hatched $\geq$1 nest. We estimated re-nesting rate as the percentage of birds that failed on their initial nesting attempt and initiated a second nest. We used a Chi-square test to compare reproductive effort rates between sites and nest types.

Nest Survival

We estimated DSR of nests and the influence of vegetation covariates on DSR using the nest survival model with a logit link function in Program MARK (White and Burnham 1999). On Peabody WMA, we had a 122-day nesting period, which encompassed 7 May-7 Sep across both years. We assumed a 23-day incubation period (Rosene 1969, Potter et al. 2011), and defined nest survival as the probability of a nest surviving the incubation period.

Our nest survival analysis consisted of two hierarchical stages consisting of four suites of models. These four suites of models represented class, landscape scale, micro-habitat, and vegetation composition metrics. Based on *a priori* models, we used a model-selection approach and Akaike's Information Criterion (AIC) to determine the model that best explained survival within our suites of candidate models. We used a ΔAIC_c value of <2 (Burnham and Anderson 2002) to determine the usefulness of a model for explaining variance in survival. We summed the weights of models containing parameters of interest to assess the strength of the individual covariate. After analyzing all four suites of models, we computed the model-averaged parameter estimates for DSR using the survival estimate from each model. We used the delta method (Powell 2007) to expand estimates to a temporal scale that encompassed the 23-day incubation period.

For our first stage of analysis, we included: nest age, nest initiation date, site, year, linear time, and constant time effects (Table B.2). Additive models were also assessed to incorporate any additive effects between multiple covariates. Our top model from the first suite (class) was then used as the baseline model in the subsequent analyses. We assumed nests were found on day 1 of incubation to estimate nest initiation and nest age (Potter et al. 2011), unless the actual starting date of nest initiation was known. We estimated nest initiation date as (onset of incubation date) – (1.2 x clutch size) (Klimstra and Roseberry 1975, Burger et al. 1995*b*).

For our second stage of analysis, we used the top model from our first stage of analysis and added covariates from the three remaining suites of models (landscape, micro-habitat, and vegetation composition). After incorporating these models, we considered the best approximating model based on the ΔAIC_c score to be our best overall model across all three scales. We used this top model for estimating DSR and overall nest survival. We separately tested nesting substrate to assess which substrate was best for increased nest survival. Burger et al. (1995*b*) and Taylor et al. (1999*a*) suggested that nest survival rates may not be comparable to their observed rates if less suitable nesting vegetation (NWSG) was available. Thus, we compared survival rates of nests based on a model containing a covariate with nesting substrate (sericea lespedeza, cool season grass, or NWSG) to assess if nests built with NWSG had higher survival rates compared to other substrates.

RESULTS

We captured 385 birds during the breeding season (211 males, 116 females, 58 unknown), of which 210 were radio-marked. We used 47 male and 45 female radio-marked bobwhite to estimate reproductive efforts. We located a total of 57 nests, of which 46 were incubated by females and 11 were incubated by males. Of the 57 nests, 54 were used for survival models, as the remaining 3 were located without a radio-marked adult associated with the nest.

Nesting rate did not differ between sites for females ($\chi^2 = 2.19$, $P = 0.13$) but did for males ($\chi^2 = 8.55$, $P = 0.003$; Table B.3). Success rate differed between sites for females ($\chi^2 = 5.46$, $P = 0.01$) but not males ($\chi^2 = 2.90$, $P = 0.08$). Clutch size did not differ between first female nests, first male nests, or second female nests ($\chi^2 = 0.66$, $P = 0.71$; Table B.4). Likewise, there was no difference in clutch size among sites for first female nests ($\chi^2 = 0.03$, $P = 0.84$), second female nests ($\chi^2 = 0.80$, $P = 0.37$), or first male nests. Of the 57 nests, 47.4% were successful and 52.6% were unsuccessful (Table B.5). Of the successful nests, 74.1% were first female nests, 11.1% were second female nests, and 14.8% were first male nests (Table B.6).

The model (Site+NestA) was the best model from our first stage of analysis based on the ΔAIC_c value and AIC weight. Of these two variables, nest age (NestA) was most important ($\beta = 0.17$, CI = 0.07-0.26) having been included in the top 5 models from the first stage. Despite Site being included in the top model, its beta value was not different from 0 ($\beta = 0.82$, CI = -0.07-1.72). The (Site+NestA) model was used as the baseline model for our second stage of analysis.

From our second stage of analysis, the model receiving the most support was (Site+NestA+DtoBG) (Table B.7). Although the top model from this second stage included DtoBG, the baseline model (Site+NestA) from our first stage of analysis had a ΔAIC_c value of 0.24 and was only 1.12 times less likely than the model including DtoBG. The beta value for the DtoBG covariate ($\beta = 0.011$, CI = -0.006-0.039) suggests that the effect of this covariate is minimal, as the beta value is not different from 0. Based on this top model, DSR for nests was 0.951 (SE = 0.010), and the probability of a nest successfully hatching after the 23-day incubation period was 0.317 (SE = 0.081). DSR estimated from model averaging of the final stage of analysis had a range of 0.950-0.951 and showed a weak negative trend in DSR as time increased across the nesting season. Nest age was still the most influential covariate after two stages of analysis. DSR of nests increased as nest age increased (Figure 3.1). Models with landscape and vegetation composition metrics measured within nest buffers were not considered likely models as only one covariate (ED_FOR) from these two suites was contained in a model with a $\Delta AIC_c \leq 2$ (Table B.7).

A total of 19 nests were built of sericea lespedeza substrate, 24 were built with cool season grasses, and 11 were built with NWSG (Table B.8). Although the effect of nesting substrate on nest survival did not differ from 0 ($\beta = 0.172$, CI = -0.451-0.795), DSR of nests having a NWSG substrate (S = 0.958, SE = 0.017) was highest, while nests with cool season grass substrate had the second highest DSR (S = 0.951, SE = (0.010), and nests having a sericea substrate (S = 0.942, SE = 0.017) had the lowest DSR.

DISCUSSION

In our study, there was no evidence of multi-scale dependency of nest survival to habitat composition. The influence of landscape-scale habitat effects was negligible with only one landscape metric (ED_FOR) included in a model with a $\Delta AIC_c \leq 2$. Although distance to unvegetated bare ground at the micro-habitat level was included in our top model, this effect was not different from 0. There was no support for landscape-scale metrics or vegetation composition metrics influencing nest survival on either site in our study. Instead, nest age was the most influential effect on nest survival with DSR increasing through the incubation period. Potter et al. (2011) also documented that nest age was the most influential effect on nest survival. However, in contrast to our results, they documented a decrease in nest survival as the incubation period progressed. They suggested this was related to daily feeding excursions by the incubating adult, which may have increased scent and sign around nest locations. Conversely, Klett and Johnson (1982) argued that in most precocial avian species, survival of nests is expected to increase as nest age increases. This is because nests that are ill-placed or in locations of higher risk will likely be predated earlier in the incubation period. Dinsmore et al. (2002) tested this hypothesis in mountain plovers and documented an overall increase in DSR of nests as nest age increased, an outcome that supports our results for bobwhite nest survival on reclaimed mined land. Bobwhite nests that were at higher risk of predation were likely destroyed or abandoned earlier during the incubation period, whereas nests further into the incubation period had a higher chance of successfully hatching.

Our overall nest survival estimate (S = 0.317) was lower than the range of estimates (0.384-0.476) observed in northern Missouri (Burger et al. 1995*b*), east-central Mississippi (Taylor and Burger 1997), southern New Jersey (Collins et al. 2009), southern Texas (Rader et al. 2007), Florida (Brinkley 2011), and on one site in southeast Iowa (Potter et al. 2011). Our estimate of nest survival was higher than the observed survival on a second southeast Iowa site (0.277; Potter et al. 2011). Female nesting rates (Pooled = 60%) on Peabody WMA were comparable to the range of estimates (58%-66%) observed in northern Missouri (Burger et al. 1995*b*), southern Georgia (Terhune et al. 2006), and Florida (Brinkley 2011). Male nesting rates were likely biased low because of low sample sizes, and were not comparable to male nesting rate estimates from previous studies. We suspect lower nest survival on Peabody WMA may be related to a lack of quality nesting vegetation such as native grasses (Collins et al. 2009, Potter et al. 2011). Burger et al. (1995*b*) observed approximately 40% available native grass nesting vegetation coverage on their study area, and Potter et al. (2011) observed approximately 50%. Within our study site, there was 4% available native grass nesting vegetation coverage on Sinclair and 11% on Ken. This percentage was estimated from our land cover delineation techniques. This difference in available nesting vegetation may have caused increased disturbance or predator pressure on nests within our study site. The model including nesting substrate revealed that nest survival was highest with native grass nest substrate and lowest with sericea lespedeza. This suggests that the limited amount of native grasses on Peabody WMA may be related to our relatively low nest survival rates.

The micro-habitat covariate DtoBG was in the top model, though its influence on nest survival appeared to be limited. Nest survival increased as the distance to bare ground from the nest location increased. This may be related to the importance of nest concealment on Peabody WMA; nests closer to areas of bare ground (i.e., freshly disced blocks, fire lines, and roads) may be more prone to disturbance and predation. Lusk et al. (2006) and Collins et al. (2009) suggested any site containing vegetation characteristics that improved nest concealment was important to nest site selection of bobwhite. The reclamation process, establishment of fast-growing, non-native plants to help prevent erosion and densely planted native grasses (Fitzgerald et al. 2004), may have limited bare ground on our study site. As such, areas with bare ground may be more readily visited by predators (Townsend et al. 2001). Townsend et al. (2001) reported bobwhite selected areas with less bare ground for nesting in Oklahoma. Taylor et al. (1999*b*) observed no preference with regard to bare ground in nest site selection in Kansas, stating litter presence was a more important site component. Conversely, Lusk et al. (2006) reported a higher mean percentage of bare ground associated with successful nests compared to unsuccessful nests in north Texas. Although this association was observed, the relationship they observed between percentage of bare ground cover and nest survival was negative and a site became unsuitable for a nest once bare ground exceeded 30%. Furthermore, mammalian nest predation was higher than snake predation as the amount of bare ground increased at nest sites (Lusk et al. 2006). Because mammals were the primary cause of known nest predation in our study (Table B.5), a nest further from bare ground may have had less of a chance of mammalian predation.

Only one model containing a landscape metric (ED_FOR) had a $\Delta AIC_c \leq 2$, and all other models containing landscape and vegetation composition variables within the nest buffer were not considered likely models. This suggests the landscape metrics we measured did not influence nesting success. Staller et al. (2002) observed only minor differences in macro-habitat composition between successful and failed nests, and nests compared to all random locations. Taylor et al. (1999*a*) also documented insensitivity of clutch success to landscape composition between cropland-dominated and rangeland-dominated sites. Likewise, Potter et al. (2011) observed no support for landscape scale or vegetation composition metrics on influencing nest success between managed and unmanaged landscapes.

MANAGEMENT IMPLICATIONS

Managers of reclaimed mined lands interested in improving bobwhite nest success should increase coverage of native species that provide suitable nesting cover (40-50% of the area). Rather than focusing habitat management at a landscape scale, efforts on reclaimed mined lands should focus on improving micro-habitat conditions, such as providing ground litter in association with desirable nesting vegetation. Future research should investigate other micro-habitat metrics that may increase nest survival, while also documenting predator-habitat interactions on reclaimed mined lands and its effects on nest success.

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LITERATURE CITED

Brinkley, S. K. 2011. Factors related to nest survival and over-winter survival of a northern bobwhite (*Colinus virginianus*) population in southwest Florida. Thesis. The University of Tennessee, Knoxville, Tennessee, USA.

Burger, Jr., L. W., T. V. Dailey, E. W. Kurzejeski, M. R. Ryan. 1995*a*. Survival and cause-specific mortality of northern bobwhite in Missouri. The Journal of Wildlife Management 59(2): 401-410.

Burger, L. W., Jr., M. R. Ryan, T. V. Dailey, and E. W. Kurzejeski. 1995*b*. Reproductive strategies, success, and mating systems of northern bobwhite in Missouri. Journal of Wildlife Management 59:417-426.

Burnham, K. P., and D. R. Anderson. 2002. Model selection and inference: a practical information-theoretic approach. Springer-Verlag, New York, New York, USA.

Collins, B. M., C. K. Williams, and P. M. Castelli. 2009. Reproduction and microhabitat selection in a sharply declining northern bobwhite population. The Wilson Journal of Ornithology 121:688-695.

Dimmick, R. W., M. J. Gudlin, and D. F. McKenzie. 2002. The Northern Bobwhite Conservation Initiative. Miscellaneous publication of the Southeastern Association of Fish and Wildlife Agencies, South Carolina, USA.

Dinsmore, S. J., G. C. White, and F. L. Knopf. 2002. Advanced techniques for modeling avian nest survival. Ecology 83:3476-3488.

Eddy, T. A. 1999. Effects of sericea lespedeza infestations on wildlife habitat in Kansas. 61st Midwest Fish and Wildlife Conference, Iowa Conservation Commission, Des Moines, Iowa, USA.

Errington, P. L., and H. L. Stoddard. 1938. Modifications in predation theory suggested by ecological studies of the bobwhite quail. Transactions of the North American Wildlife Conference 3:736:740.

Fitzgerald, J. L., J. L. Bell, and M. D. Kinney. 2004. Establishing and maintaining native warm-season grasses on mined lands dominated by sericea lespedeza. Proceedings of the Fourth Eastern Native Grass Symposium. October 4-6, 2004, University of Kentucky, Lexington, KY.

Holt, R. D., L. W. Burger, Jr., B. D. Leopold, D. Godwin. 2009. Over-winter survival of northern bobwhite in relation to landscape composition and structure. Pages 432-446 in S. B. Cederbaum, B. C. Faircloth, T. M. Terhune, J. j. Thompson, J. P. Carroll, eds. Gamebird 2006:Quail VI and Perdix XII. 31 May-4 June 2006. Warnell School of Forestry and Natural Resources, Athens Georgia, USA.

Janke, A. K. 2011. Survival and habitat use of non-breeding northern bobwhites on private lands in Ohio. Thesis, Ohio State University, Columbus, Ohio, USA.

Klett, A. T., and D. H. Johnson. 1982. Variability in nest survival rates and implications to nesting studies. Auk 99:77-87.

Klimstra, W. D., and J. L. Roseberry. 1975. Nesting ecology of the bobwhite in southern Illinois. Wildlife Monographs 41. 37pp.

Lebreton, J. –D., K. P. Burnham, J. Clobert, and D. R. Anderson. 1992. Modeling survival and testing biological hypotheses using marked animals: a unified approach with case studies. Ecological Monographs 62:67-118.

Lusk, J. J., S. G. Smith, S. D. Fuhlendorf, and F. S. Guthery. 2006. Factors influencing northern bobwhite nest-site selection and fate. Journal of Wildlife Management 70:564-571.

Martin, T. E. 1993. Nest predation among vegetation layers and habitat types: revising the dogmas. American Naturalist 141:897-913.

McGarigal, K., and B. J. Marks. 1994. FRAGSTATS: spatial analysis program for quantifying landscape structure. Version 2.0. Forest Science Department, Oregon State University, Corvallis, USA.

Palmer, W. E., S. D. Wellendorf. 2007. Effect of radiotransmitters on northern bobwhite annual survival. The Journal of Wildlife Management 71: 1281-1287.

Potter, L. M., D. L. Otis, and T. R. Bogenschutz. 2011. Nest success of northern bobwhite on managed and unmanaged landscapes in southeast Iowa. Journal of Wildlife Management 75:46-51.

Powell, L. A. 2007. Approximating variance of demographic parameters using the delta method: a reference for avian biologists. The Condor 109:949-954.

Rader, M. J., L. A. Brennan, F. Hernández, N. J. Silvy, and B. Wu. 2007. Nest-site selection and nest survival of northern bobwhite in southern Texas. Wilson Journal of Ornithology 119:392-399.

Rollins, D. and J. P. Carroll. 2001. Impacts of predation on northern bobwhite and scaled quail. Wildlife Society Bulletin 29:39-51.

Roseberry, J. L., and W. D. Klimstra. 1984. Population ecology of the bobwhite. Southern Illinois University Press, Carbondale, Illinois, USA.

Rosene, W. 1969. The Bobwhite Quail: its life and management. Rutgers University Press, New Brunswick, New Jersey, USA.

Sauer, J. R., J. E. Hines, J. E. Fallon, K. L. Pardieck, D. J. Ziolkowski, Jr., and W. A. Link. 2011. The North American Breeding Bird Survey, Results and Analysis 1966-2009. Version 3.23. 2011. USGS Patuxent Wildlife Research Center, Laurel, Maryland, USA.

Seckinger, E. M., L. W. Burger, Jr., R. Whittington, A. Houston, R. Carlisle. 2008. Effects of landscape composition on winter survival of northern bobwhites. The Journal of Wildlife Management 72(4):959-969.

Staller, E. L., W. E. Palmer, and J. P. Carroll. 2002. Macrohabitat composition surrounding successful and depredated northern bobwhite quail nests. Pages 61-64 *in* S. J. DeMaso, J. W.

P. Kuvlesky, F. Hernández, and M. E. Berger, editors. Quail V: Proceedings of the Fifth National Quail Symposium. Texas Parks and Wildlife Department, Austin, TX.

Stoddard, H. L. 1931. The bobwhite quail: its habits, preservation and increase. Charles Scibner's Sons, New York, New York, USA.

Taylor, J. D., II, and L. W. Burger, Jr. 1997. Reproductive effort and success of northern bobwhite in Mississippi. Proceedings of the Annual Conference Southeastern Association of Fish and Wildlife Agencies 51:329-341.

Taylor, J. S., K. E. Church, D. H. Rusch, J. R. Cary. 1999*a*. Macrohabitat effects on summer survival, movements, and clutch success of northern bobwhite in Kansas. The Journal of Wildlife Management 63(2):675-685.

Taylor, J. S., K. E. Church, D. H. Rusch, J. R. Cary. 1999*b*. Microhabitat selection by nesting and brood rearing northern bobwhite in Kansas. The Journal of Wildlife Management 63:686-694.

Terhune, T. M., D. C. Sisson, and H. L. Stribling. 2006. The efficacy of relocating wild northern bobwhites prior to breeding season. Journal of Wildlife Management 70:914-921.

Terhune, T. M., D. C. Sisson, J. B. Grand, H. L. Stribling. 2007. Factors influencing survival of radiotagged and banded northern bobwhites in Georgia. The Journal of Wildlife Management 71: 1288-1297.

Townsend, D. E., II, R. E. Masters, R. L. Lochmiller, D. M. Leslie, Jr., S. J. DeMaso, and A. D. Peoples. 2001. Characteristics of nest sites of northern bobwhites in western Oklahoma. Journal of Range Management 54:260-264.

White, G. C., and K. P. Burnham. 1999. Program MARK: survival estimation from populations of marked animals. Bird Study 46 (suppl): S120-S139.

White, G. C., and R. A. Garrot. 1990. Analysis of wildlife radio-tracking data. Academic Press, Inc. San Diego, CA, USA.

APPENDIX B

Table B.1: Delineated vegetation types and total coverage (ha) on Peabody WMA, Ohio and Muhlenberg Counties, Kentucky, USA, 2009–2011.

Site	Vegetation	Hectares	Total (ha)
Ken	Annual Grain	25.8	1853.1
	Forest Deciduous	405.7	
	Native Warm-Season Grass	205.7	
	Open Herbaceous	524.4	
	Scrub Shrub	519.4	
	Water	165.4	
	Wetland Emergent	6.8	
Sinclair	Annual Grain	6.5	1470.6
	Forest Deciduous	327.2	
	Native Warm-Season Grass	58.0	
	Open Herbaceous	671.6	
	Scrub Shrub	321.7	
	Water	69.7	
	Wetland Emergent	16.0	

Table B.2. List and description of class, micro-habitat, and landscape metrics assess effects on nest survival of radio-marked northern bobwhite on Peabody WMA, Ohio and Muhlenberg Counties, Kentucky, USA, 1 Apr 2009-30 Sep 2011.

Metric	Scale	Description
T	Group	linear time
NestA	Group	nest age
NestI	Group	nest initition date
Year	Group	year
Site	Group	site, either Ken or Sinclair
Null	Group	contant time
DtoBG	Micro-habitat	distance to bare ground (m)
DtoED	Micro-habitat	distance to edge (m)
Substrate	Micro-habitat	nesting substrate (sericea lespedeza, cool season grass, NWSG)
Vegtype	Micro-habitat	vegetation type (Forest, scrub-shrub, open herbaceou, NWSG)
FOR	Landscape	% forest vegetation within landscape buffer
SS	Landscape	% scrub-shrub vegetation within landscape buffer
NWSG	Landscape	% NWSG vegetation within landscape buffer
OH	Landscape	% OH vegetation within landscape buffer
ED_FOR	Landscape	Forest to open vegetation edge density
ED_SS	Landscape	Scrub-shrub to open vegetation edge density
FOR_CA	Landscape	Core area of forest vegetation
SS_CA	Landscape	Core area of scrub-shrub vegetation
NWSG_CA	Landscape	Core area of NWSG vegetation
OH_CA	Landscape	Core are of OH vegetation
CI	Landscape	Contagion index

Table B.3. Reproductive efforts of radio-marked male and female northern bobwhite surviving past 7 May on Peabody WMA, Ohio and Muhlenberg Counties, Kentucky, USA, 1 Apr 2009-30 Sep 2011.

	n		Nesting rate (%)		Success rate (%)		Renest rate (%)	
Site	Male	Female	Male	Female	Male	Female	Male	Female
Sinclair	12	11	0.0	72.7	0.0	18.2	0.0	25.0
Ken	35	34	5.7	55.9	2.9	35.3	0.0	10.5
Pooled	47	45	4.3	60.0	2.1	31.1	0.0	14.8

Table B.4. Mean clutch size for female incubated first nests, female incubated renests, and male incubated first nests of radio-marked northern bobwhite on Peabody WMA, Ohio and Muhlenberg Counties, Kentucky, USA, 1 Apr 2009-30 Sep 2011.

		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		21	12.6	0.7		4	8.0	2.3		7	12.0	1.1
Ken		19	13.6	0.5		2	12.0	3.0		4	12.0	0.8
Pooled		40	13.1	0.5		6	9.3	1.8		11	12.0	0.7

Table B.5. Nest fates of radio-marked northern bobwhite on Peabody WMA, Ohio and Muhlenberg Counties, Kentucky, USA, 1 Apr 2009-30 Sep 2011.

	Site					
	Sinclair		Ken		Total	
Fate	*n*	%	*n*	%	*n*	%
Successful	11	34.4	16	64.0	27	47.4
Unsuccessful	21	65.6	9	36.0	30	52.6
Abandoned	4	19.0	1	11.1	5	16.7
Nest Depredation	12	57.2	8	88.9	20	66.6
Mammalian	6		2		8	40.0
Snake	2		1		3	15.0
Unknown	4		5		9	45.0
Adult mortality	5	23.8	0	0.0	5	16.7
Mammalian	1		0		1	20.0
Avian	2		0		2	40.0
Unknown	2		0		2	40.0
Total	32	100.0	25	100.0	57	100.0

Table B.6. 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 2009-30 Sep 2011.

		Nest Type					
		F-incubated first nest		F-incubated renest		M-incubated nest	
	Site	*n*	%	*n*	%	*n*	%
Incubated nests	Sinclair	21	65.6	4	12.5	7	21.9
	Ken	19	76.0	2	8.0	4	16.0
	Pooled	40	70.2	6	10.5	11	19.3
Successful nests	Sinclair	9	75.0	1	8.3	2	16.7
	Ken	11	73.3	2	13.3	2	13.3
	Pooled	20	74.1	3	11.1	4	14.8

Table B.7. Ranking *a priori* models based on ΔAIC_c values and AIC_c weights used to assess the influence of class, micro-habitat, and landscape metrics on northern bobwhite nest survival on Peabody WMA, Ohio and Muhlenberg Counties, Kentucky, USA, 1 Apr 2009-30 Sept 2011[a].

Model	AICc	Delta AICc	AICc Weights	Model Likelihood	Num. Par	Deviance
{Site+NestA+DtoBG}	126.876	0	0.11969	1	4	118.796
{Site+NestA}	127.116	0.2403	0.10614	0.8868	3	121.068
{Site+NestA+DtoED}	128	1.1244	0.06822	0.57	4	119.92
{SitexNestA}	128.269	1.3932	0.05964	0.4983	2	124.245
{NestA}	128.66	1.7847	0.04904	0.4097	2	124.637
{Site+NestA+ED_FOR}	128.761	1.8849	0.04664	0.3897	4	120.681
{Site+T+NestA}	128.844	1.9684	0.04473	0.3737	4	120.764
{Site+NestA+Vegtype}	128.849	1.9728	0.04463	0.3729	4	120.769
{Site+NestA+Substrate}	128.853	1.977	0.04454	0.3721	4	120.773
{Site+NestA+OH}	128.966	2.0907	0.04208	0.3516	4	120.887
{Site+NestA+SS}	129.001	2.1254	0.04136	0.3456	4	120.921
{Site+NestA+SS_CA}	129.059	2.1832	0.04018	0.3357	4	120.979
{Site+NestA+OH_CA}	129.061	2.185	0.04014	0.3354	4	120.981
{Site+NestA+NWSG}	129.086	2.2106	0.03963	0.3311	4	121.006
{Site+NestA+ED_SS}	129.102	2.2258	0.03933	0.3286	4	121.022
{Site+NestA+FOR}	129.112	2.2366	0.03912	0.3268	4	121.032
{Site+NestA+CI}	129.113	2.2373	0.03911	0.3268	4	121.033
{Site+NestA+FOR_CA}	129.124	2.2481	0.03889	0.3249	4	121.044
{Site+NestA+NWSG_CA}	129.148	2.2718	0.03844	0.3212	4	121.068
{Site+Year+NestA}	130.658	3.7827	0.01806	0.1509	5	120.538
{Site+T}	141.18	14.3046	0.00009	0.0008	3	135.133
{Site+NestI}	141.7	14.824	0.00007	0.0006	3	135.652
{Site}	141.831	14.9556	0.00007	0.0006	2	137.808
{T}	142.424	15.5482	0.00005	0.0004	2	138.4
{NestI}	143.026	16.1503	0.00004	0.0003	2	139.002
{Site+Year}	143.191	16.3149	0.00003	0.0003	4	135.111
{Null}	143.579	16.7029	0.00003	0.0003	1	141.571
{Year}	144.432	17.5567	0.00002	0.0002	3	138.385

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

Table B.8. Summary of nesting substrate use by site used to evaluate nest survival of northern bobwhite on Peabody WMA, Ohio and Muhlenberg Counties, Kentucky, USA, 1 Apr 2009-30 Sep 2011.

		Nesting Substrate		
Site		Sericea lespedeza (*n*)	Cool season grasses (*n*)	Native warm-season grasses (*n*)
Sinclair		12.0	16.0	3.0
Ken		7.0	8.0	8.0

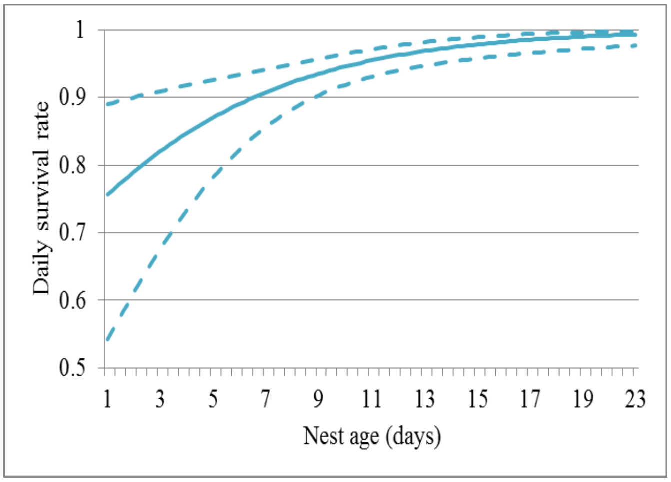

Figure B.1. Daily survival rates and confidence intervals (dotted lines) of northern bobwhite nests as a function of nest age on Peabody WMA, Ohio and Muhlenberg Counties, Kentucky, USA, 1 Apr 2009-30 Sep 2011.

Table B.9. Summary of micro-habitat and landscape metrics by site used to evaluate nest survival of northern bobwhite on Peabody WMA, Ohio and Muhlenberg Counties, Kentucky, USA, 1 Apr 2009-30 Sep 2011.

Parameter	Site	Scale	x̄	SE
Distance to bare ground (m)	Ken	Micro-habitat	17.4	3.5
	Sinclair		17.0	3.8
Distance to edge (m)	Ken	Micro-habitat	30.7	6.2
	Sinclair		34.1	7.7
Forest (%)	Ken	Landscape	8.0	4.0
	Sinclair		3.0	1.1
Scrub-Shrub (%)	Ken	Landscape	28.9	3.9
	Sinclair		15.2	2.1
NWSG (%)	Ken	Landscape	27.4	4.7
	Sinclair		9.0	1.7
OH (%)	Ken	Landscape	32.0	4.3
	Sinclair		69.7	2.9
Forest Core Area (ha)	Ken	Landscape	0.9	0.4
	Sinclair		0.4	0.1
Scrub-shrub Core Area (ha)	Ken	Landscape	3.2	0.4
	Sinclair		1.8	0.2
NWSG core area (ha)	Ken	Landscape	2.9	0.5
	Sinclair		0.8	0.2
Open herbaceous core area (ha)	Ken	Landscape	3.5	0.5
	Sinclair		7.7	0.3
Contagion index (%)	Ken	Landscape	47.5	2.8
	Sinclair		54.1	2.9
Forest and Open Edge Density (m/ha)	Ken	Landscape	0.5	0.2
	Sinclair		1.7	0.5
Scrub-shrub and Open Edge Density (m/ha)	Ken	Landscape	36.0	3.5
	Sinclair		33.4	2.6

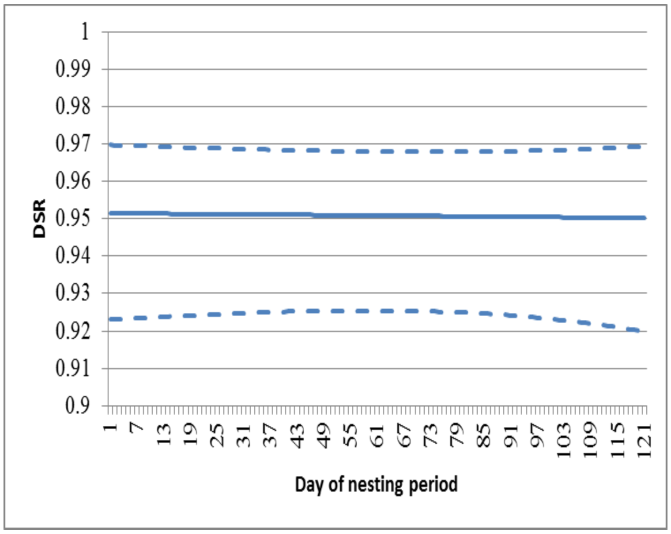

Figure B.2. Daily survival rates and confidence intervals (dotted lines) of northern bobwhite nests from model averaging on Peabody WMA, Ohio and Muhlenberg Counties, Kentucky, USA, 1 Apr 2009-30 Sep 2011.

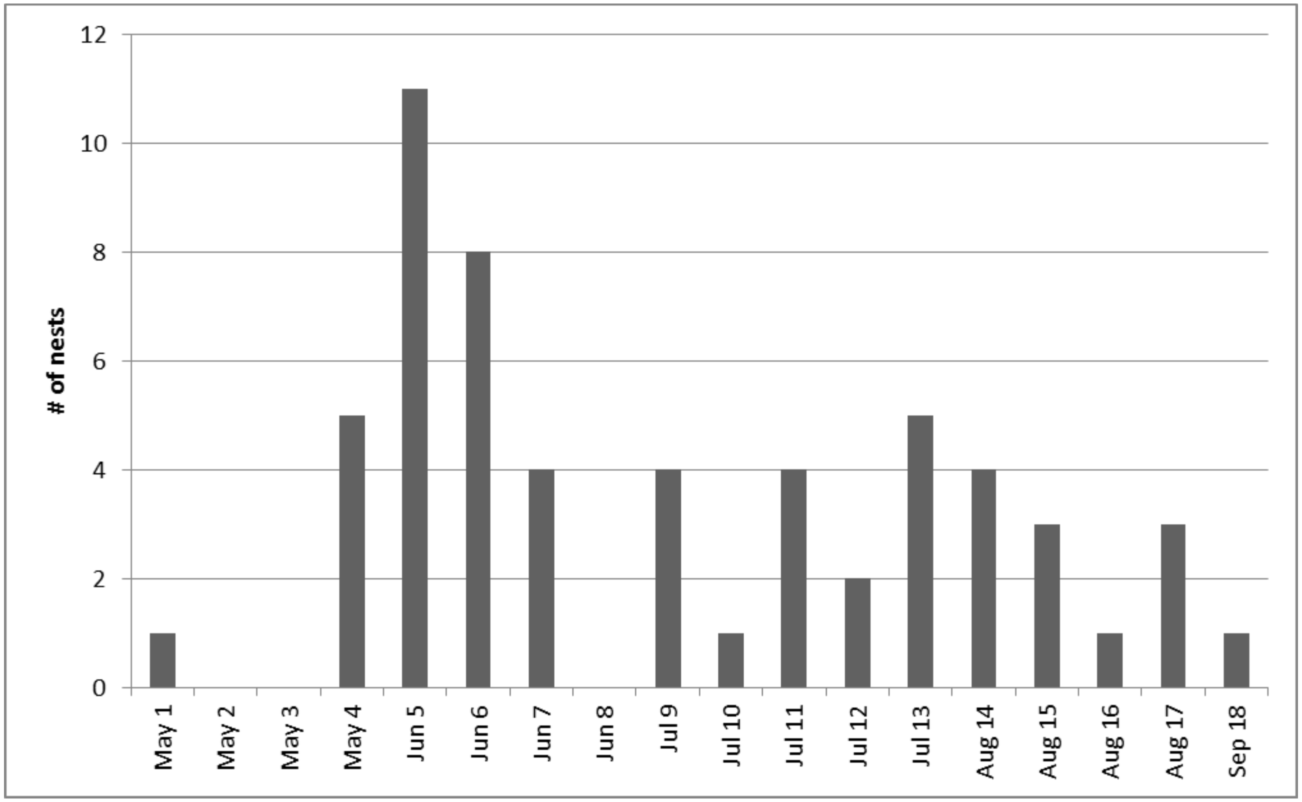

Figure B.3. Nest incubation activity for the 122-day nesting period for northern bobwhite on Peabody WMA, Ohio and Muhlenberg Counties, Kentucky, USA, 1 Apr 2009-30 Sep 2011.

PART IV

CONCLUSIONS

The two primary objects of my research were to (1) document survival, cause specific mortality, and assess multi-scale vegetation effects on survival of northern bobwhite (*Colinus virginianus*) on Peabody WMA, a reclaimed coal mine in Western Kentucky (Part II); and (2) document reproductive efforts, success, and assess multi-scale vegetation effects on nest survival of northern bobwhite on reclaimed mined land in Western Kentucky (Part III). Key conclusions are described briefly below.

Multi-scale habitat effects have been shown to be important to bobwhite survival in previous research (Seckinger et al. 2008, Janke et al. 2011). We detected no evidence of multi-scale influences of vegetation components to survival of bobwhite on our study site. At the home range scale, the amount of forest within a home range positively influenced survival rates. We attributed this to the presence of woody cover at a local scale increasing survival by providing available escape cover. The amount of open herbaceous vegetation at the landscape scale also had a positive influence on survival, though this effect was not different from 0. Survival was statistically different between groups and site. We expect the difference in survival between groups, as we grouped birds based on whether or not they had home ranges. Birds had to survive longer to have a home ranged associated with them. Ken survival rates were consistent with previous research but were relatively low on the Sinclair site (Burger et al. 1995*a*, Seckinger et al. 2008, Holt et al. 2009). Differences in survival among sites may exist because of the variation in survival across space caused by factors such as habitat suitability and predator abundance (Terhune et al. 2007).

Estimated nest survival rates on our study site were relatively low compared to the range of estimates observed in previous research throughout the species' range (Burger et al. 1995*b*, Taylor and Burger 1997, Collins et al. 2009). Vegetation composition and landscape scale vegetation metrics estimated within a 13.8 ha nest buffer were not shown to be influential to nest survival. Instead, nest age was shown to be the most important factor to survival. Daily survival rates of nests were shown to increase and nest age increases. This is to be expected in precocial species, because nests that are ill placed or are in locations of higher risk will likely be predated earlier in the incubation period (Klett and Johnson 1982). At the micro-habitat scale, distance to bare ground was shown to have a positive influence on nest survival. Nests closer to bare ground had lower survival rates, likely because of the increased exposure to disturbance or predation.

Management efforts on reclaimed mined land should focus on creating open, early succession vegetation at a landscape scale, while providing woody escape cover at the local scale. When possible, management should focus on native plant species to provide this structure for bobwhite, as non-natives often planted on reclaimed lands are not ideal bobwhite food. Although bare ground is essential for brooding success, managers should also focus on providing areas of herbaceous canopy coverage and available ground litter to increase nesting success.

Future research should identify specific vegetation differences between Ken and Sinclair that may be driving differences in survival rates. Also, efforts should be taken to assess relative predator levels and monitor chick survival, as this may be influencing population dynamics on Peabody WMA.

LITERATURE CITED

Burger, Jr., L. W., T. V. Dailey, E. W. Kurzejeski, M. R. Ryan. 1995*a*. Survival and cause-specific mortality of northern bobwhite in Missouri. The Journal of Wildlife Management 59(2): 401-410.

Burger, L. W., Jr., M. R. Ryan, T. V. Dailey, and E. W. Kurzejeski. 1995*b*. Reproductive strategies, success, and mating systems of northern bobwhite in Missouri. Journal of Wildlife Management 59:417-426.

Collins, B. M., C. K. Williams, and P. M. Castelli. 2009. Reproduction and microhabitat selection in a sharply declining northern bobwhite population. The Wilson Journal of Ornithology 121:688-695.

Holt, R. D., L. W. Burger, Jr., B. D. Leopold, D. Godwin. 2009. Over-winter survival of northern bobwhite in relation to landscape composition and structure. Pages 432-446 in S. B. Cederbaum, B. C. Faircloth, T. M. Terhune, J. j. Thompson, J. P. Carroll, eds. Gamebird 2006:Quail VI and Perdix XII. 31 May-4 June 2006. Warnell School of Forestry and Natural Resources, Athens Georgia, USA.

Janke, A. K. 2011. Survival and habitat use of non-breeding northern bobwhites on private lands in Ohio. Thesis, Ohio State University, Columbus, Ohio, USA.

Klett, A. T., and D. H. Johnson. 1982. Variability in nest survival rates and implications to nesting studies. Auk 99:77-87.

Seckinger, E. M., L. W. Burger, Jr., R. Whittington, A. Houston, R. Carlisle. 2008. Effects of landscape composition on winter survival of northern bobwhites. The Journal of Wildlife Management 72(4):959-969.

Taylor, J. D., II, and L. W. Burger, Jr. 1997. Reproductive effort and success of northern bobwhite in Mississippi. Proceedings of the Annual Conference Southeastern Association of Fish and Wildlife Agencies 51:329-341.

VITA

Evan was born in Slidell, Louisiana. There, he quickly learned to love the outdoors, spending much of his time hiking, camping, and fishing. At the age of 7, he moved to the outskirts of Mt. Juliet, Tennessee. There, he became heavily involved in the Boy Scouts of America. Through his experiences with Scouts, he gained an appreciation of camping and high adventure trips all throughout North America, and eventually received his Eagle Scout award at the age of 15. Evan attended the University of Tennessee and received his Bachelor of Science degree in Forestry, Resource Management in 2009. He worked as a field technician for the Tennessee Tree Improvement Program during his undergraduate studies. After graduating, he immediately pursued 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. Evan is engaged to his wonderful fiancée Ashley, who also shares a passion for wildlife, hunting and fishing, and exploring the amazing world we live in.