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Individual and landscape-level effects of selective herbicides, mowing, and prescribed fire on habitat quality for northern bobwhite (*Colinus virginianus*)

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INDIVIDUAL AND LANDSCAPE-LEVEL EFFECTS OF SELECTIVE HERBICIDES,
MOWING, AND PRESCRIBED FIRE ON HABITAT QUALITY FOR NORTHERN
BOBWHITE (*COLINUS VIRGINIANUS*)

A Thesis

Submitted to the Graduate Faculty of the
Louisiana State University and
Agricultural and Mechanical College
in partial fulfillment of the
requirements for the degree of
Master of Science

in

The School of Renewable Natural Resources

by

Charles Lynn Kitts
B.S., University of Tennessee at Martin, 1999
May 2004

DEDICATION

This work is dedicated to my wife, Beth. I am grateful to have known such a wonderful person. Without her inspiration and support I would never have made it this far. Her companionship has made my life what it is, and I look forward to spending the rest of it with her.

ACKNOWLEDGEMENTS

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ABSTRACT

Northern bobwhite (*Colinus virginianus*) populations have been declining throughout their range during the past 30 years primarily because of a result of loss of early successional habitat. Specifically, intensive silviculture practices and reduction in the use of prescribed fire has led to this loss. I studied effects of management practices (selective herbicides, mowing, and prescribed fire) on male and covey distribution, and brood-rearing habitats on Jackson-Bienville Wildlife Management Area (JBWMA). Specifically, I used calling surveys to assess landscape characteristics associated bobwhite distribution. I also measured vegetation and arthropod response, using imprinted bobwhite chicks, pitfalls, and sweep nets, to different habitat manipulations. Male bobwhites were closely associated with early successional habitats, and negatively associated with the proportion of landscape variables associated with 16-29 year old pine stands. Several vegetation characteristics were affected by the use of herbicides, mowing, and burning; however, arthropod response was not similar. Imprinted chicks selected arthropod orders similar to wild chicks, although they did not consume a large quantity of arthropods. These data indicates habitats on JBWMA may not be of the quality needed for brood-rearing. Future research should focus on long-term effects of manipulations (selective herbicides, mowing, and burning) on northern bobwhite populations. Managers should focus on creating early successional habitats across forested landscape, and continue to search for methods to enhance these habitats for northern bobwhites.

CHAPTER 1: INTRODUCTION AND STUDY AREA

Introduction

Northern bobwhite (*Colinus virginianus*; hereafter bobwhite) populations have declined over most of their geographic range in the past 3 decades (Droege and Sauer 1990, Brennan 1991). The overall population has declined at a rate of 2.4% per year in the United States since the mid-1960s. The southeastern United States, the geographic center of bobwhite range (Rosene 1969), has seen the most rapid decline (Brennan 1991). Louisiana has experienced an even greater loss over that period. Church et al. (1993) reported a long-term decrease in bobwhite population trend of 5.3% from 1966-91. The same study found that in the short term (1982-91) Louisiana experienced a 7.6% decrease in bobwhite abundance. From 1966-1999 bobwhite populations in Louisiana have experienced a 4.8% decline (Sauer et al. 2000). Despite numerous research efforts and thousands of publications on bobwhites, populations are still on the decline. Effective management strategies have been known for over half a century (Stoddard 1931, Brennan 2002), yet managers are still frustrated with bobwhite population projections.

Declines in bobwhite populations have been attributed to deterioration and loss of suitable habitats (Klimstra 1982) for nesting, brood-rearing, fall and winter covey ranges, and escape cover. Bobwhite populations require early to mid-successional habitats, such as grassy and weedy areas, primarily associated with small agricultural fields and fire-maintained forest ecosystems (Rosene 1969). Historically, bobwhites were a byproduct of land use practices. Tenant farming, small-scale agriculture, livestock management, which burned woodlands to promote grasses for grazing, and natural fire produced

abundant bobwhite populations (Landers and Mueller 1986). Burger (2002) points to reduction in landscape heterogeneity associated with monoculture production in agriculture and forestry production as the likely cause of declines across the range of the bobwhite. Changing agricultural practices that maximize production, and large-scale silviculture practices that maximize wood fiber production, generally reduce quality and heterogeneity of habitat for bobwhite (Exum et al. 1982, Fies et al. 1992). Changes in agriculture and forestry practices have reduced brushy weedy edge habitat, and increased large scale agricultural farms and dense pine (*Pinus spp.*) forests. Dense forests canopies reduce light penetration to the understory, which reduces grasses and forbs abundance. This change in land use has resulted in a substantial loss of early successional habitats critical to bobwhite survival. Although other factors, such as predation, have been viewed as the source of region-wide population declines, loss of early successional habitat due to changing agricultural and forestry practices (Roseberry and Klimstra 1984, Roseberry 1993) is the primary cause of bobwhite declines.

Prescribed fire is one of the most economical ways of managing habitats for bobwhites (Landers and Mueller 1986). Burning can be used to stimulate production of critical food plants (Brennan 1991). Stoddard (1931) promoted prescribed fire to reduce undesirable ground litter and hardwood sprouts, while enhancing plants preferred by bobwhites. Fire can be used to enhance herbaceous vegetation (Cain et al. 1998), which is exploited by bobwhites; however, reduction in extent and frequency of fire (DeMaso et al. 1998) has led to pine or mixed pine-hardwood forests developing high basal areas, tree densities, dense hardwood mid- and understory, and low herbaceous plant diversity. Short rotational burns (1-2 years) are needed to effectively manage bobwhite habitats in

pine systems (Brennan 1991). Presently, most burn rotations are in the 5-7 year range (Personal observation). This results in a landscape of dense under and mid-story stands of little benefit to bobwhites. Exclusion of fire in pine forests of the Southeast is potentially the most critical problem bobwhites experience (Burger 2002).

In intensively managed pine ecosystems, bobwhites rely on disturbance-induced habitat changes to reproduce and persist. Disturbance, in the form of timber thinning or clear-cutting, creates openings in the forest canopy. This allows sunlight to reach the ground, promoting the growth of forbs and grasses needed for food by bobwhites. In these ecosystems, early successional communities only exist for a short period (2-5 years) after timber harvest. These communities then are lost through natural succession (Burger 2002). Although early successional plant communities could be maintained by intermediate disturbance, without the use of fire or other disturbance the quality of these habitats for bobwhites declines.

Promoting fire within forested regions not only increases grasses and forbs, but also increases insect abundance. Hurst (1972) pointed out that burned areas had a greater biomass of available insects than unburned areas. Protein-rich invertebrates can comprise >80% of a bobwhite chick's diet for the first 2 weeks of life (Handley and Cottom 1931, Nestler et al. 1945, Hurst 1972, Jackson et al. 1987). Invertebrates are an essential source of amino acids, protein, water, and energy needed by chicks for survival and growth (Nestler et al. 1942, 1945). During the breeding season, DeVos (1986) reported that bobwhites tend to select habitats with higher insect densities. Parsons et al. (1998) also found that bobwhites with broods used areas with greater arthropod biomass and abundance. Hens and chicks will readily feed on a variety of insects, including beetles

(Coleoptera), aphids (Homoptera), ants (Hymenoptera) and plant bugs (Hemiptera), as well as numerous varieties of spiders (Araneae) throughout the spring and summer (Jackson et al. 1987).

Fuller (1994) stated that bobwhite habitat management in the southeast is very similar to Red-cockaded Woodpecker (*Picoides borealis*; hereafter RCW) habitat management. RCWs are a federally endangered species associated with mature pine forests of the southeastern United States (Federal Register, 13 October 1970, Volume 35 [199:16047], United States Forest Service, 1995). Intensive management for RCW includes maintaining mature pine forests, short rotation (2-3 years) prescribed burning, and removal of hardwood mid-story woody vegetation using mechanical (mowing) and herbicide application (Conner and Rudolph 1991, United States Forest Service 1995). RCW management practices have been shown to be beneficial to bobwhite (Burger et al. 1998, Bowman et al.1999). Reduction in mid-story hardwood stems has led to increased grasses and forbs beneficial to bobwhite for nesting, brood-rearing, and foraging (Bowman et al.1999). Fuller (1994) found that bobwhite use of RCW colonies was in greater proportion than habitat availability within their home range. Arthropod abundance and biomass were greater in RCW colonies than in unmanaged habitat (Fuller 1994). Therefore, management practices directed at reducing mid-story hardwood stems in RCW colonies also should provide greater quality habitat for bobwhites. This habitat has the potential to provide an essential source of arthropods needed for bobwhite chicks.

A variety of management practices can be used to help combat understory succession. Mechanical techniques (roller-drum chopping and mowing) can reduce mid-story density and allow light penetration to the under-story, but are time consuming and

labor intensive, and also can be detrimental to bobwhites. For instance, mowing during late spring and summer has the potential to destroy bobwhite nests (Stoddard 1931). These mechanical practices only help to build up the hardwood root stock, while doing nothing to promote grasses and legumes (Welch 2000). Disking can be used to promote grasses and legumes (Rosene 1969), but must be timed properly and requires a quality seed bank for vegetation to respond. While beneficial plants such as blackberries (*Rubus spp.*) and grasses can be promoted, depending on the timing of disking, detrimental pest species such as sicklepod (*Cassia obtusifolia*) also may arise (Welch 2000).

Researchers have suggested using herbicides to manage vegetation to enhance wildlife habitat (Goodrum 1960). Herbicides can be applied for conifer site preparation, conifer release from competition, right-of-ways, and wildlife openings (Ahrens 1994, McComb and Hurst 1987). Herbicides may provide an alternative and supplement to prescribed fire, resulting in a change in composition and productivity of plant communities for bobwhites (Guthery et al. 1987, Washburn et al. 2000, Madison et al. 2001). Using selective herbicides may reduce competing hardwood mid-story vegetation and promote early successional plant communities such as grasses and legumes that are exploited by bobwhites. McComb and Hurst (1987) found that re-vegetation following herbicide applications can improve bobwhite habitat quality by increasing preferred bobwhite food plants. Miller et al. (1989) found that sites treated with Velpar® had a greater abundance of legumes than other mechanically or chemically treated sites. In Mississippi, following site preparation with Velpar, Hurst and Palmer (1988) saw abundant legume growth that provided excellent quail habitat.

Arsenal® is another selective herbicide that can be used to control hardwoods, and annual and perennial weeds (Ahrens 1994). Feken (1995) found an increase in herbaceous plants and bare ground following Arsenal application in South Carolina. One year after treatment in east-central Mississippi, the number of forbs, legumes, and vines increased with the use of Arsenal (Watkins et al. 1989). Brooks et al. (1993) found that diversity of herbaceous plant species was richer in plots treated with Arsenal than with Tordon+Garlon®, or Velpar. In Georgia, bobwhite plant foods were more abundant with the treatment of Arsenal than with Tordon®, Garlon, or Velpar 2-4 years following treatment (Witt et al. 1993).

Most previous studies have focused on plant community responses following chemical site-preparation treatments. Recently, efforts have been made to study effects of herbicide applications on existing vegetation conditions and whether applications could enhance nesting and brood rearing habitats (Welch 2000, Greenfield et al. 2002). Madison et al. (2001) found that herbicide treated plots in one year of their study satisfied most bobwhite nesting requirements. In the Red Hills region of south Georgia and north Florida, Welch (2000) found a >3 fold increase in forb coverage combined with a decrease in hardwood stem density with treatment of Arsenal, compared to no increase of forb coverage and an increase in hardwood stem density with mechanical treatment. Jones and Chamberlain (2004) found that Arsenal in combination with fire increased the quality of nesting and brood rearing habitat in pine forests relative to burning alone. These studies show that selective herbicides can be used, with and without prescribed fire, to enhance habitat quality for bobwhites.

A reduction in the role of prescribed fire has hampered efforts by landowners to effectively manage for bobwhites. Removal of hardwood species and reducing density of canopy closure in pine forests may allow managers to reclaim areas and convert them to early successional communities beneficial to bobwhite. To do this, effects of herbicides, along with other practices (i.e. mowing) on bobwhite habitat quality, needs to be better understood. Few studies, however, have addressed the application of these management techniques in a pine dominated system. Managed pine systems continue to increase across the southeast, and thus, management strategies must be formulated to effectively manage for bobwhites in these systems. Information on effects of herbicides and mechanical practices has the potential to provide managers with tools needed to make decisions about how to reduce hardwood mid-story and restore habitats that have been lost through succession. Landowners desiring to manage for bobwhites by reducing mid-story hardwood species and canopy coverage may be able to use selective herbicides to accomplish this objective.

My objectives were to examine effects of selective herbicide (Arsenal) applications and other management strategies on bobwhite populations at the individual and population scale. Specifically, I examined male distribution in the spring and covey distribution in the fall in relation to landscape characteristics to determine macro-habitat characteristics associated with bobwhite abundance and distribution. I measured brood habitat quality at sites treated with Arsenal, with and without prescribed burning, mechanical control, prescribed fire, and untreated areas. Arthropod, vegetation, and brood response were examined to assess the suitability of each method for bobwhite habitat management.

Study Area

This study was conducted on the 13,136 ha Jackson-Bienville Wildlife Management Area (JBWMA) located in Jackson and Bienville parishes in north-central Louisiana. JBWMA is owned by Weyerhaeuser Company and is managed intensively for wood fiber production. JBWMA is bordered on the east by US highway 167 and on the west by LA highway 147. The area is primarily composed of short-rotation (1-30 years old) loblolly pine (*P. taeda*) plantations. JBWMA is intersected with bottomland hardwoods along streamsides and contains active RCW colonies throughout the area surrounded by uneven-aged mature pine stands. Stands containing RCW colonies are managed using mechanical treatments (mowing) to control woody mid- and understory hardwood species and to promote early successional vegetation. JBWMA is also traversed by several kilometers of gas pipelines, which are planted annually for wildlife use.

Overstory species across JBWMA included shortleaf pine (*P. echinata*), water oak (*Quercus nigra*), post oak (*Q. stellata*), southern red oak (*Q. falcata*), mockernut hickory (*Carya tomentosa*), and sweetgum (*Liquidambar styraciflua*). Understory species consisted mainly of blackberry (*Rubus spp.*), greenbrier (*Smilax spp.*), trumpet creeper (*Campsis radicans*), poison ivy (*Toxicodendron radicans*), chinese privet (*Ligustrum sinense*), sumacs (*Rhus spp.*), lespedezas (*Lespedeza spp.*), japanese honeysuckle (*Lonicera japonica*), and beautyberry (*Callicarpa americana*). There were no primary roads on JBWMA, but several secondary gravel and dirt roads were available across the area. All terrain vehicle (ATV) trails also were distributed across the area for public use.

Application of Arsenal was conducted during September 2001 (63 ha) and October 2002 (79 ha) at a rate of 1120 g/ha (16 oz/acre) broadcast sprayed using a t-boom system mounted on a skidder. Herbicides mixtures were prepared immediately prior to application to minimize hydrolysis and degradation of the herbicide in the tanks (Miller and Glover 1991). Mowing was conducted on appropriate stands during May 2002 and April 2003. A prescribed burn was performed during April-June 2003 on all stands treated with Arsenal (142 ha) and several additional stands not treated with Arsenal (~400 ha).

Although bobwhites are a primary management species on JBWMA, numerous other species are managed for as well. White-tail deer (*Odocoileus virginianus*), eastern wild turkey (*Meleagris gallopavo*), wild hogs (*Sus scrofa*), eastern cottontail rabbits (*Sylvilagus floridanus*), and gray squirrels (*Sciurus carolinensis*) are all common game species on the area. Predators of the bobwhite on JBWMA include the red fox (*Vulpes vulpes*), coyote (*Canis latrans*), bobcat (*Lynx rufus*), raccoon (*Procyon lotor*), red-tailed hawk (*Buteo jamaicensis*), and cooper's hawk (*Accipiter cooperii*).

CHAPTER 2: RELATIONSHIP BETWEEN LANDSCAPE CHARACTERISTICS AND NORTHERN BOBWHITE ABUNDANCE AND DISTRIBUTION

Landscape ecology has experienced rapid developments in both theory and application (Turner et al. 2001, Wu and Hobbs 2002). The fundamental process of landscape ecology is the emphasis of interactions between spatial patterns and ecological processes (Turner et al. 2001, Bissonette 2003). Emerging technologies, such as Geographic Information Systems (GIS), allow managers to examine habitat patches and patterns across landscapes. Turner et al. (2001) pointed out that land management activities, such as timber harvest and land-use changes, provide opportunities to apply advancing technologies in studying ecological phenomena at the landscape scale. Application of technology will allow managers to develop more robust quantitative models (Burger 2002) to characterize the distribution of habitat patches and their influences on wildlife.

Managers throughout the southeastern United States have been frustrated by persistent declines in northern bobwhite populations. Long-term, region-wide declines in bobwhite populations have been attributed to several factors, most notably habitat loss. Large-scale, intensive monoculture agriculture and forest production has led to a loss of early successional communities and a reduction in landscape heterogeneity (Burger 2002). A reduction in the frequency and extent of prescribed fire as a management tool (Brennan et al. 1998) has led to advanced stages of succession culminating in a reduction in habitat quality for bobwhite populations (Roseberry et al. 1979).

In pine-dominated ecosystems, early successional plant communities may only persist as ephemeral patches following timber harvest (Burger 2002). These patches persist for brief periods (2-5 years), then only in patches facilitated by thinning, clear-

cutting, and site preparation (Burger 2002). The composition and juxtaposition of these patches in the landscape may greatly influence the ability of bobwhites to thrive. Planning and implementation of management objectives for bobwhites will require the ability to inventory, analyze, and interpret habitat patches at several differing spatial scales (Flather et al. 1992, Roseberry and Sudkamp 1998). Roseberry (1993) called for quail biologists to incorporate landscape ecology into their plans and implement management regimes at a broader spatial scale. However, relationships between bobwhite population dynamics and landscape composition and structure remain poorly understood (Michener et al. 2000, Schairer et al. 1999). To assess the response of bobwhites to landscape characteristics and patterns, I examined the relationships among spring male distribution, fall covey distribution, and landscape characteristics on JBWMA. Spatially explicit models were developed to examine the role of landscape characteristics and patterns on bobwhite abundance and distribution.

Methods

Spring/summer call counts were conducted weekly on JBWMA from 15 April to 30 June 2002 and 2003 to examine distribution and abundance of male bobwhites. Sixty stations (2 routes, 30 stations each) were established 0.8 km apart along roads and stations were monitored for 5 minutes, with each route beginning 30 minutes before sunrise and continuing until completed. Sampling began at a randomly selected station on each route. Every calling male heard was recorded to provide information on breeding activity and distribution of males during the breeding period (DeMaso et al. 1992). Fall covey counts were conducted once during October and November of each year. Surveys were conducted using the same listening stations as the summer surveys;

however, to avoid hearing the same covey at 2 consecutive stops, every other station was surveyed (30 stations total). Each survey was conducted between one-half hour before sunrise until sunrise. Surveys consisted of a listener playing a recording of a covey call and listening for 5 minutes. Observers recorded the number of coveys heard and the approximate azimuth to the covey.

A digital landcover data layer was developed in ArcView GIS (Environmental Systems Research Institute (ESRI), Redlands, California, USA) using 7.5-minute digital orthophoto quarter quadrangles (DOQQs) obtained from Weyerhaeuser Company. Land cover classes were digitized in ArcView, then classified by stand age, structure, and management history. Application of Arsenal occurred during September 2001 and October 2002, and a growing season fire (April) was applied to all stands treated with Arsenal in 2003. No stands received prescribed fire in 2002, and several stands were treated with only prescribed fire in April 2003. This resulted in different landscape characteristics between years. Habitats were classified as early successional (planted fields, road edges, rights-of-way, and 0-5 year old pine plantations), 6-15 year old pine plantations, 15-29 year old pine plantations, herbicide (no-burn) treated stands (2002 only), herbicide+burn stands (2003 only), unburned mature mixed pine-hardwood stands, burned mature pine-hardwood stands (2003 only), and bottomland hardwoods (Table 2.1).

Buffers were created around each listening station at 200, 400, and 800 meter radii for each year to determine landscape characteristics associated with each station at multiple spatial scales (Figure 2.1). The proportion of each habitat located within buffers was calculated for each station. Landscape metrics were calculated within each buffer

and year using PatchAnalyst extension (Rempel and Carr 2003) in ArcView. Changing landscape characteristics and land management practices resulting from the application of Arsenal and prescribed fire required models to be examined annually. Associations at the landscape scale were examined to determine habitats important to bobwhites on JBWMA. Results of the landscape models were used, along with knowledge of habitats deemed important to bobwhites, to reduce variables for class-level analyses. Early successional plant communities, for example, have been shown to be crucial in the reproduction and survival of bobwhites (Landers and Mueller 1986, Burger 2002). Habitat variables classified as early successional were considered for class level examinations regardless of their significance at the landscape scale, due to the role these communities play in bobwhite population processes.

Stepwise logistic regression was initially used to assess landscape characteristics most closely associated with male and covey locations (Table 2.2). Default significance levels ($\alpha=0.05$) of entry and retention in SAS (SAS Institute Inc. 1999) were used to build models. To measure goodness of fit for each model, a Hosmer-Lemeshow statistic was calculated (Hosmer and Lemeshow 1989) using the LACKFIT option in SAS (SAS Institute Inc. 1999). Also, a rescaled, generalized R^2 provided by the RSQ option in SAS (Allison 1999) was used to examine the predictive power of each model. The generalized R^2 is based on the likelihood ratio chi-square for testing the null hypothesis that all of the coefficients are zero (Cox and Snell 1989).

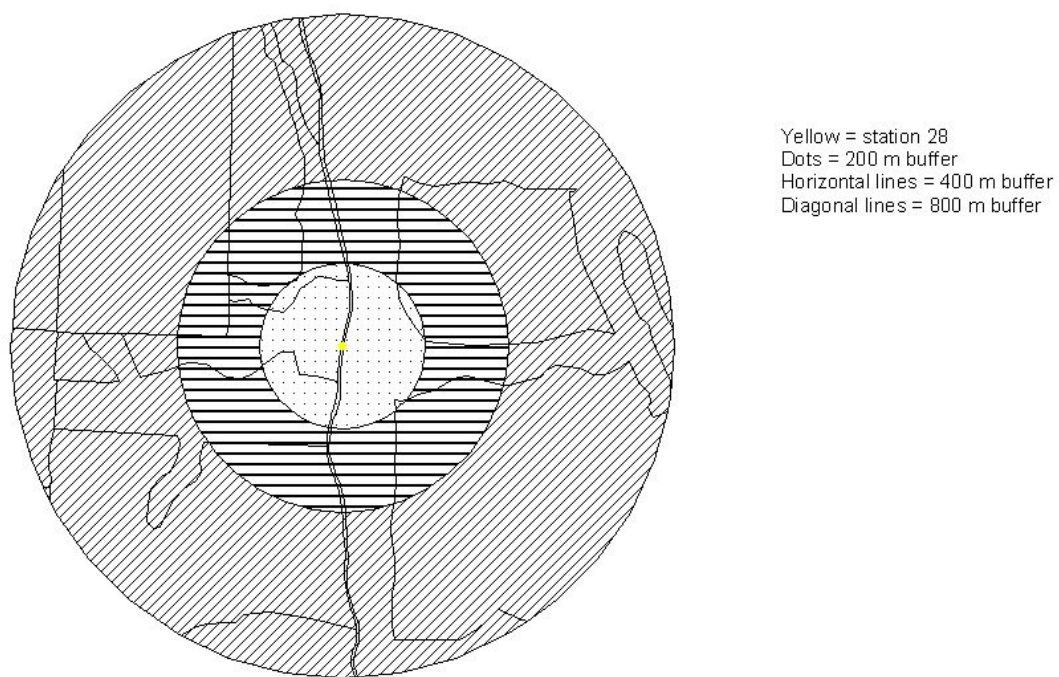


Figure 2.1. 200, 400, and 800 m buffers created around listening station # 28 on Jackson-Bienville Wildlife Management Area, 2002 and 2003.

Table 2.1. Proportion of the landscape composed of each habitat type on Jackson-Bienville Wildlife Management Area, 2002 and 2003.

<i>Habitat</i>	<i>2002</i>	<i>2003</i>
Bottomland Hardwoods	0.24	0.24
Early Successional	0.21	0.21
6-15 year old pine stands	0.03	0.03
16-29 year old pine stands	0.35	0.35
Herbicide	0.01	0.01
Burned	0.00	0.03
Mature Mixed Pine-Hardwoods	0.16	0.13

Table 2.2. Candidate variables used to determine landscape characteristics associated with bobwhite abundance and distribution on Jackson-Bienville Wildlife Management Area during 2002-2003.

<i>Parameter</i>	<i>Definition</i>
% Bottomland Hardwood	Proportion of Landscape in Bottomland Hardwoods
% Early Successional	Proportion of Landscape in Early Successional habitat
% 6-15 year old pine stands	Proportion of Landscape in 6-15 year old pine stands
% 16-29 year old pine stands	Proportion of Landscape in 16-29 year old pine stands
% Herbicide	Proportion of Landscape treated with Herbicide
% Burned	Proportion of Landscape treated with prescribed fire
% Mature Mixed Pine-Hardwoods	Proportion of Landscape in Mature Mixed Pine-Hardwoods
Number of Patches	Number of patches per buffer
Mean Patch Size	Average patch size within buffers
Median Patch Size	Median patch size in buffers
Patch Size Standard Deviation	Standard deviation of patch areas
Patch Size Coefficient of Variance	Coefficient of variation of patch size
Total Edge	Total amount of edge in each buffer
Edge Density	Amount of Edge relative to landscape area
Mean Patch Edge	Average amount of edge per patch
Mean Perimeter to Area Ratio	Sum of perimeter/area ratio divided by number of patches
Mean Shape Index	Measure of shape complexity
Mean Patch Fractal Dimension	Measure of shape complexity
Area Weighted Mean Patch Fractal Dimension	Measure of shape complexity weighted by individual patches
Area Weighted Mean Shape Index	Measure of shape complexity weighted by individual patches

Results

Male Distribution

Over both years, 266 males were recorded (2002=135, 2003=131). Males were detected between 29 April and 30 June each year (Figure 2.2). Mean number of males heard per station was 2.25 (SE=0.18) in 2002 and 2.18(SE=0.31) in 2003. Males were heard at 26 and 24 stations during 2002 and 2003, respectively.

2002

Landscape Models

For all models, the data adequately fit the logistic distribution ($P>0.199$). At the 200 m scale, 5 parameters were retained in the model: an intercept term ($\beta=289.5$, SE=100.4, $\chi^2_1=8.312$, $P<0.001$), proportion of the landscape in 16-29 year old pine stands ($\beta=-14.245$, SE=6.474, $\chi^2_1=4.842$, $P<0.01$), edge density ($\beta=4.705$, SE=1.578, $\chi^2_1=8.892$, $P<0.001$), number of patches ($\beta=-1.162$, SE=0.568, $\chi^2_1=4.183$, $P=0.04$), and area weighted mean patch fractal dimension ($\beta=-254.1$, SE=87.156, $\chi^2_1=8.496$, $P<0.001$). This suggests that males were associated with areas having greater edge density, but also with areas having a lesser proportion of the landscape in 16-29 year old pine plantations, fewer number of patches, and reduced shape complexity of patches (Table 2.3). The model correctly classified 76.9% of stations where males were heard and 76.5% of stations where males were not heard. The model provided a generalized R^2 of 0.63.

At the 400 m scale, 3 parameters were retained: an intercept term ($\beta=-6.455$, SE=1.853, $\chi^2_1=12.132$, $P<0.001$), proportion of the landscape classified as early successional ($\beta=5.114$, SE=1.673, $\chi^2_1=9.346$, $P<0.001$), and edge density ($\beta=1.048$, SE=0.351, $\chi^2_1=8.917$, $P<0.001$). This indicates males were detected in areas with a

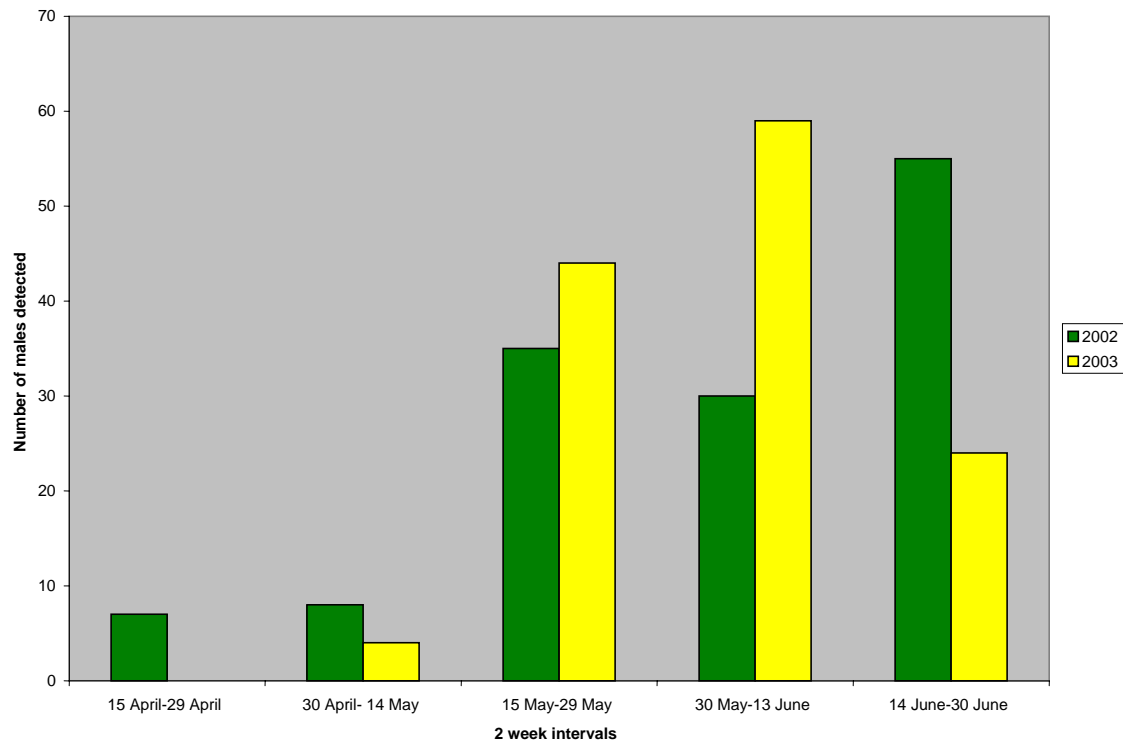


Figure 2.2. Distribution of calling male bobwhites at 2 week intervals on Jackson-Bienville Wildlife Management Area during 2002-2003.

greater proportion of the landscape in early successional habitats and greater edge density (Table 2.3). The model correctly classified 65.4% of stations where males were heard and 67.6% of stations where males were not heard, and provided a generalized R^2 of 0.45.

At the 800 m scale, 4 parameters were retained: an intercept term ($\beta=-10.320$, $SE=2.979$, $\chi^2_1=12.001$, $P<0.001$), proportion of the landscape classified as early successional ($\beta=10.849$, $SE=3.601$, $\chi^2_1=9.055$, $P<0.001$), proportion of the landscape in 16-29 year old pine plantations ($\beta=-12.507$, $SE=5.062$, $\chi^2_1=6.104$, $P=0.001$), and edge density ($\beta=2.361$, $SE=0.729$, $\chi^2_1=10.489$, $P<0.001$). This indicates males were detected in areas with greater proportion of the landscape in early succession and greater edge density, but in areas with a lesser proportion of the landscape in 16-29 year old pine plantations (Table 2.3). The model correctly classified 80.8% of stations where males were heard and 82.4% of stations where they were not, and produced a generalized R^2 of 0.70.

Class Level Models

For all models, the data adequately fit the logistic distribution ($P>0.31$). At the 200 m scale, 3 variables were retained: an intercept term ($\beta=-2.508$, $SE=0.720$, $\chi^2_1=12.128$, $P<0.001$), number of patches in early successional habitat ($\beta=1.172$, $SE=0.304$, $\chi^2_1=14.840$, $P<0.001$), and area weighted mean patch fractal dimension of 16-29 year old pine plantations ($\beta=-1.712$, $SE=0.811$, $\chi^2_1=4.458$, $P=0.035$). Males were detected in areas with more patches in early successional habitat, and reduced shape complexity (also a function of area) of 16-29 year old pine stands (Table 2.4). The model

correctly classified 69.2% of stations where males were heard and 79.4% of stations where males were not heard, and produced a generalized R^2 of 0.54.

At the 400 m scale, 3 variables were retained: an intercept term ($\beta=-4.317$, $SE=1.217$, $\chi^2_1=12.572$, $P<0.001$), number of patches in early successional habitat ($\beta=1.634$, $SE=0.435$, $\chi^2_1=14.116$, $P<0.001$), and area weighted mean patch fractal dimension of 16-29 year old pine plantations ($\beta=-2.153$, $SE=0.833$, $\chi^2_1=6.681$, $P=0.009$). Males were heard in areas with more patches in early successional habitat, and reduced shape complexity of 16-29 year old pine stands (Table 2.4). The model correctly classified 84.6% of stations where males were heard and 88.2% of stations where males were not heard, and produced a generalized R^2 of 0.73.

At the 800 m scale, 3 variables were retained: an intercept term ($\beta=-5.783$, $SE=1.902$, $\chi^2_1=9.251$, $P=0.002$), edge density of early successional habitats ($\beta=0.082$, $SE=0.024$, $\chi^2_1=11.354$, $P<0.001$), and edge density of 16-29 year old pine plantations ($\beta=-0.064$, $SE=0.025$, $\chi^2_1=6.492$, $P=0.01$). Males were detected in areas with greater edge density in early successional habitats and less edge density in 16-29 year old pine plantations (Table 2.4). The model correctly classified 80.8% of stations where males were heard and 82.4% of stations where males were not heard, and produced a generalized R^2 of 0.67.

2003

Landscape Models

For each model, the data adequately fit the logistic distribution ($P>0.40$). At the 200 m scale, 5 parameters were retained in the model: an intercept term ($\beta=-4.744$, $SE=1.304$, $\chi^2_1=13.225$, $P<0.001$), proportion of the landscape in early successional

habitats ($\beta=4.384$, $SE=1.542$, $\chi^2_1=8.087$, $P<0.001$), proportion of the landscape in unburned mature mixed pine hardwoods ($\beta=5.500$, $SE=2.035$, $\chi^2_1=7.308$, $P<0.001$), number of patches ($\beta=0.501$, $SE=0.175$, $\chi^2_1=8.159$, $P<0.001$), and mean perimeter to area ratio ($\beta=-0.001$, $SE=0.041$, $\chi^2_1=4.179$, $P=0.04$). Males were associated with landscapes having a greater proportion of early successional habitats and unburned mature mixed pine hardwoods, and greater number of patches. However, males were detected in landscapes with a smaller mean perimeter to area (i.e. less total edge) ratio (Table 2.3). The model correctly classified 70.8% of stations where males were heard and 80.6% of stations where males were not detected, with a generalized R^2 of 0.46.

At the 400 m scale, 4 parameters were retained: an intercept term ($\beta=58.047$, $SE=23.354$, $\chi^2_1=6.178$, $P=0.01$), proportion of the landscape in early successional habitats ($\beta=5.439$, $SE=1.944$, $\chi^2_1=7.827$, $P<0.001$), number of patches ($\beta=0.648$, $SE=0.194$, $\chi^2_1=11.218$, $P<0.001$), and mean patch fractal dimension ($\beta=-51.043$, $SE=19.230$, $\chi^2_1=7.045$, $P<0.001$). Males were heard in areas with a greater proportion of the landscape in early succession and a greater number of patches, but in areas with reduced shape complexity (Table 2.3). The model correctly classified 83.3% stations where males were heard and 77.8% of stations where males were not heard with a generalized R^2 of 0.54.

At the 800 m scale, 4 parameters were retained: an intercept term ($\beta=-3.756$, $SE=1.920$, $\chi^2_1=3.825$, $P=0.05$), proportion of the landscape in bottomland hardwoods ($\beta=-7.001$, $SE=3.121$, $\chi^2_1=5.932$, $P=0.01$) and 16-29 year old pine stands ($\beta=-4.839$, $SE=2.286$, $\chi^2_1=4.481$, $P=0.03$), and edge density ($\beta=1.332$, $SE=0.529$, $\chi^2_1=6.334$, $P=0.01$). Males were heard in areas with less proportion of the landscape in bottomland

hardwoods and 16-29 year old pine stands, while in areas with greater edge density (Table 2.3). The model correctly classified 66.7% of stations where males were heard and 69.4% of stations where males were not, with a generalized R^2 of 0.43.

Class Level Models

For the 200 m and 400 m models scales, the data adequately fit the logistic regression ($P>0.56$). At the 200 m scale, an intercept term ($\beta=-2.120$, $SE=0.611$, $\chi^2_1=12.021$, $P<0.001$), and number of patches in early successional habitats ($\beta=0.810$, $SE=0.256$, $\chi^2_1=10.024$, $P=0.001$) were retained. Males were detected in areas with greater number of patches in early successional habitats (Table 2.4). The model correctly classified 58.3% of stations where males were heard and 83.3% of stations where males were not heard, with a generalized R^2 of 0.29.

At the 400 m scale, an intercept term ($\beta=-3.483$, $SE=0.959$, $\chi^2_1=13.195$, $P<0.001$), edge density of early successional habitat ($\beta=0.027$, $SE=0.008$, $\chi^2_1=10.316$, $P<0.001$), and mean perimeter to area ratio of unburned mature mixed pine hardwoods ($\beta=0.003$, $SE=0.001$, $\chi^2_1=4.330$, $P=0.037$) were retained. Males were detected in areas with greater edge density of early successional habitats and higher mean perimeter to area ratios (i.e. more edge) of unburned mature mixed pine hardwoods (Table 3.4). The model correctly classified 79.2% of stations where males were heard and 75.0% of stations where males were not heard, and produced a generalized R^2 of 0.32.

At the 800 m scale, the data did not fit the logistic distribution ($P=0.04$), and the model should be interpreted with this forethought. An intercept term ($\beta=-1.356$,

Table 2.3. Landscape-level characteristics (mean \pm SE) associated with male bobwhite distribution and abundance at multiple spatial scales on Jackson Bienville Wildlife Management Area, 2002-2003 (NS denotes variable was not significant at that spatial scale).

Year	Parameter	200 m		400 m		800 m	
		Heard	Not Heard	Heard	Not Heard	Heard	Not Heard
2002	ES ^a	NS	NS	37.0(4.84)	16.0(3.24)	34.0(2.25)	17.0(1.96)
	16-29 year old pine stands ^b	1.0(2.50)	20.0(10.90)	NS	NS	3.0(0.36)	15.0(3.61)
	Number of Patches	6.77(2.15)	4.97(2.38)	NS	NS	NS	NS
	AWMPD ^c	1.24(0.01)	1.23(0.01)	NS	NS	NS	NS
	ED ^d	7.59(1.62)	6.23(1.62)	5.02(0.97)	4.09(1.00)	3.90(0.66)	3.20(0.64)
2003	ES ^a	29.0(6.25)	16.0(4.41)	31.0(5.29)	15.0(2.89)	NS	NS
	16-29 year old stands ^b	NS	NS	NS	NS	5.0(1.00)	14.0(3.24)
	BH ^e	NS	NS	NS	NS	8.0(0.49)	19.0(2.56)
	UNB ^f	21.0(4.84)	9.0(3.61)	NS	NS	NS	NS
	Number of Patches	6.83(2.14)	5.02(2.38)	11.17(3.33)	8.0(2.95)	NS	NS
	ED ^d	NS	NS	NS	NS	3.87(0.50)	3.26(0.42)
	MPAR ^g	2161.12(3425.63)	2094.94(5135.43)	NS	NS	NS	NS
	MPFD ^h	NS	NS	1.28(0.001)	1.29(0.001)	NS	NS

^a Proportion of landscape classified as early successional.

^b Proportion of landscape classified as 16-29 year old pine plantations.

^c Area weighted mean patch fractal dimension (Approaches 1 for simple shapes and 2 for complex shapes) (McGaril and Marks 1995).

^d Edge density (Amount of edge (m) relative to total land area (ha)).

^e Proportion of landscape in bottomland hardwoods.

^f Proportion of landscape in unburned mature mixed pine hardwoods.

^g Mean perimeter to area ratio (m:ha).

^h Mean patch fractal dimension (Measure of shape complexity, approaches 1 for simple shapes and 2 for complex shapes) (McGaril and Marks 1995).

SE=0.552, $\chi^2_1=6.043$, $P=0.014$) and number of patches in early successional habitats ($\beta=0.166$, SE=0.083, $\chi^2_1=4.051$, $P=0.041$) were retained. Males were detected in areas with a greater number of patches of early successional habitat (Table 3.4). The model correctly classified 70.8% of stations where males were heard and 69.4% of stations where males were not heard, with a generalized R^2 of 0.09.

Covey Distribution

2002

Sixteen coveys were detected during the fall of 2002. No habitat parameters were retained in any landscape-scale models at any spatial scale ($P<0.05$). For class-level analysis at the 200 m and 400 m scale, the data did not fit the logistic distribution adequately ($P<0.05$), and neither model retained any parameters. At the 800 m scale, the data adequately fit the logistic distribution ($P=0.75$).

Two variables were retained in the model: an intercept term ($\beta=-0.756$, SE=0.405, $\chi^2_1=3.48$, $P=0.06$), and area weighted mean patch fractal dimension associated with 16-29 year old pine stands ($\beta=-1.212$, SE=0.589, $\chi^2_1=4.232$, $P=0.039$). Area weighted mean patch fractal dimension associated with 16-29 year old pine stands where coveys were heard was 0.31(SE=0.32), while 0.74(SE=0.37) where coveys were not heard. Coveys were detected in areas with less shape complexity of 16-29 year old pine stands. The model correctly classified 75.0% of stations where coveys were detected and 60.4% of stations where coveys were not detected, with a generalized R^2 of 0.12.

Table 2.4. Class level characteristics (mean \pm SE) associated with male bobwhite distribution and abundance at multiple scales on Jackson Bienville Wildlife Management Area, 2002-2003 (NS denotes variable was not significant).

Year	Parameter	<u>200</u>		<u>400</u>		<u>800</u>	
		Heard	Not Heard	Heard	Not Heard	Heard	Not Heard
2002	NP ES ^a	3.34(2.47)	1.47(0.98)	4.88(5.14)	1.91(1.35)	NS	NS
	AWMPFD	0.09(0.12)	0.40(0.35)	0.24(0.26)	0.58(0.39)	NS	NS
	16-29 ^b						
	ED ES ^c	NS	NS	NS	NS	99.03(628.00)	56.82(795.80)
	ED 16-29 ^d	NS	NS	NS	NS		
2003	NP ES	2.91(2.86)	1.52(0.99)	NS	NS	6.71(11.69)	4.83(10.89)
	ED ES	NS	NS	126.36(3550.06)	73.07(1374.49)	NS	NS
	MPAR ^e	NS	NS	219.63(326.65)	68.72(171.86)	NS	NS
	UNB						

^a Number of patches in early successional habitat.

^b Area weighted mean patch fractal dimension of 16-29 year old pine stands (Approaches 1 for simple shapes and 2 for complex shapes) (McGaril and Marks 1995).

^c Edge density of early successional habitat (m/ha).

^d Edge density of 16-29 year old pine stands (m/ha).

^e Mean perimeter to area ratio (m:ha) of unburned mature mixed pine hardwoods.

2003

Eleven coveys were detected in fall 2003. No habitat parameters were retained in any landscape, or class-level models at any spatial scale, and the data did not fit the logistic distribution ($P < 0.05$).

Discussion

In southern Louisiana, Bell et al. (1985) found bobwhites in clearcuts 63% of the time, whereas clearcuts only accounted for 23% of the study area. Stoddard and Komarek (1941) found that populations of bobwhites could be maintained on heavily wooded lands if $\geq 25\%$ of the land comprised of openings or small fields. My findings support these earlier studies, in that early successional habitats were important in predicting the presence of bobwhites across multiple spatial scales. Buffers in my study only averaged 5.8% in early successional habitats; however, the percentage of landscapes where males were heard within significant buffers averaged 32.75% in early succession across all scales.

Edge density was positively associated with occurrence of males, and my study was consistent with other studies in Louisiana. Bell et al. (1985) found that bobwhites were found within 50 m of some edge 53% of the time, and Best (1983) noted a positive relationship with bobwhites and fencerow habitats. Schairer et al. (1999) also found that edge with early successional habitats appeared beneficial to bobwhites. Although edge plays a critical role in managing landscapes for bobwhite populations (Rosene 1969), there were conflicting associations with edge in some models in this study. The number of patches within each buffer was significant at only 2 buffer distances. While it was positively associated with male distribution at 400 m in 2003, it was negatively

associated with 200 m buffers in 2002. An increase in the number of patches will create more edge habitat, but there may be a point where edge is redundant or even detrimental to bobwhite populations (Guthery and Bingham 1992, Guthery 1997), offering an explanation for the contradiction.

At the landscape-level, a negative association with 16-29 year old pine stands and male occurrence was a common theme. This association was likely related to poor habitat quality in these stands. These stands had high canopy closure, reducing the amount of light reaching the understory, and understory vegetation is sparse, or dominated by woody species. Similarly, Schairer et al. (1999) found that closed canopy forests were negatively associated with bobwhite populations.

Bissonette (2003) pointed out that arbitrary choices of scale are not sufficient if progress is to be made in relating organisms to landscape patterns. A basic understanding of the natural history of a species seems to be a prerequisite to choosing the proper scale (Bowers and Dooley 1999, Bissonette 2003). However, Levin (1992) states no single landscape mechanism explains all pattern of organism distribution on all scales. Schairer et al. (1999) used 800 m buffers in the Piedmont region of Virginia, with habitats consisting of conifers, mixed hardwoods, and agricultural lands. This was similar to Roseberry and Sudkamp's (1998) landscapes (900 m radius) evaluated in Illinois. Roseberry (1982) and Davis (1979) used estimates of 400 m and 800 m, respectively, as the range at which bobwhites could be detected. DeMaso et al. (1992) used a 700 m radius for indexing bobwhite density, based on the threshold of human audibility. Although these estimates are at the maximum range of human threshold, wind velocity and vegetation biomass may affect detectability rates of bobwhites. Roseberry and

Sudkamp (1998) examined a variety of landscapes in Illinois with differing proportions of agriculture and forested habitats, while DeMaso et al. (1992) estimated covey density in the prairies and plains of Texas. These landscapes are different from the intensively managed pine forests of the southeast. High basal area pine stands on JBWMA may influence the detectability of bobwhites. Since the power of a bobwhite's call has not been measured (DeMaso et al. 1992), I used several spatial scales in determining landscape characteristics associated with bobwhite populations. By using 800 m radius buffers, I tried to include landscapes surrounding potential bobwhite locations; however, the spacing of listening stations at 0.8 km apart caused overlap of most station's at this scale. There was no overlap at the 200 and 400 m spatial scales, resulting in each stations associated landscapes being independent. These stations also had the potential to incorporate habitats actually used by males in the spring.

Although herbicide application has the potential to benefit bobwhites (Jones and Chamberlain 2004), habitats treated with Arsenal and subsequently burned were not found important at predicting bobwhite occurrence. This may have resulted from the small amount of area treated. Only 63 ha were treated with Arsenal in 2002 and 79 ha in 2003. These applications were localized to a small proportion (1.0%) of the entire landscape, and only 8 listening stations had this habitat incorporated into them. Although bobwhites were detected at all of these stations, determining their importance to bobwhites was impossible. Future efforts may consider applying Arsenal across more of the landscape.

Burger (2002) noted that early successional habitats are critical for bobwhite populations to survive. Class-level models in this study clearly supported this contention.

Also at the class level for each spatial scale, edge density or number of patches was positively related to the presence of bobwhites. Brennan (1991) and Burger (2002) also point out that once habitat is lost through natural succession, bobwhite populations suffer. Again, the class-level models tended to support this notion.

Landscape variables examined in this study were not able to predict occurrence of bobwhite coveys during fall. Since surveys were only conducted at each station once during the fall, detectability of coveys was low. Repeated measures of covey calling rates may alleviate this problem in future studies. DeMaso et al. (1992) used repeated line-transect methodology to index bobwhite density, and Michener et al. (2000) used hunter encounters to relate bobwhite covey activity to landscape features. Although this measure may be biased to higher quality landscapes, because of reliance on hunting parties that select habitat features generally associated with encountering bobwhites for hunting, it ensured multiple encounters with coveys. To alleviate this problem, future research efforts should use repeated surveys to increase the likelihood of detecting bobwhite coveys.

To address regional population questions for bobwhites, biologists and managers must recognize regional or landscape perspectives and realize that viable local populations are affected by interactions with surrounding populations (Fies et al. 2002). Empirical models provide a starting point for developing spatially explicit habitat models (Burger 2002). These models combine habitat models with population models that incorporate habitat-specific population parameter estimates. Burger (2002) noted that these estimates can help relate demographic parameters to relative habitat quality. Development of management plans on JBWMA may incorporate the models in this study

to promote habitats known to be associated positively with bobwhites. However, model validation could not be done due a lack of data from similar suitable landscapes with similar management practices. Therefore, caution must be taken when considering this data. Future research efforts in the region may provide more insight into bobwhite population processes. Roseberry and Sudkamp (1998) found that bobwhites would rarely be abundant in the absence of a suitable landscape matrix. Management at the landscape scale should incorporate all components required by bobwhites throughout their life. Landscape-scale models have the capacity to identify these components and concentrate their efforts in areas likely to support high bobwhite populations (Schairer et al. 1999). Most plantation acreage (55%) in the midsouth occurs in industrial forest lands (Rosson 1995), and a large percentage (55%) of that is in seedling and sapling size-class (Trani et al. 2001) therefore, pine plantations will likely constitute a substantial proportion of early successional habitats available to bobwhites (Burger 2002). Understanding the influence of habitat composition on bobwhite occurrence could allow the development of management strategies to maximize bobwhite abundance and wood fiber production. Forest managers could plan timber harvests and thinnings around surrounding landscape components capable of supporting bobwhite populations. Landscape models used in this study also can be used to monitor individual and population responses to management activities conducted in intensively managed pine systems by examining habitat characteristics associated with bobwhite occurrence. Future management plans aimed at maintaining an economic timber supply while simultaneously managing for bobwhites can use these spatial models to achieve both goals.

CHAPTER 3: VEGETATION AND ARTHROPOD RESPONSE TO BURNING, MOWING, AND APPLICATION OF SELECTIVE HERBICIDES

Arthropods play a crucial role in the development of bobwhite chicks during their first 2 weeks of life. Rosene (1969) pointed out that abundance and availability of arthropods may be an important factor in determining chick survival. Arthropods comprise >80% of a chick's diet during the early stages of life (Handley and Cottom 1931, Nestler et al. 1945, Hurst 1972, Jackson et al. 1987), and provide an essential source of amino acids, protein, water, and energy needed for growth and survival (Nestler et al. 1942, 1945). Insufficient dietary protein may suppress the immune system of chicks (Lochmiller et al. 1993) rendering them more susceptible to disease, and Hill (1985) and Potts (1986) noted associations between brood survival in gray partridges (*Perdix perdix*) and arthropod abundance.

The quality of brood-rearing habitat for bobwhites has been closely linked with arthropod abundance (Jackson et al. 1987, Madison et al. 1995). Habitat patches with relatively greater abundance of arthropods are considered better brood-rearing habitats than patches lower in arthropod abundance (Burger et al. 1993, DeVos and Mueller 1993, Parsons et al. 2000). Bobwhites require early successional areas for brood-rearing (Burger 2002); however, in the southeast, increasing acreage of monoculture pine plantations (Trani et al. 2001) coupled with an increase in total vegetation control in clearcuts has reduced the availability of these areas (Fies et al. 1992). In forested landscapes, early successional patches are created by timber thinning or clear-cutting and only exist for a short period of time (2-5 years), then are lost through natural succession (Burger 2002). Thus, maintaining early successional areas needed for brood-rearing requires disturbance. Although brood-rearing is associated with arthropod abundance,

manipulating vegetation structure and diversity can affect arthropod distribution and abundance (Southwood et al. 1979, Deulli et al. 1990).

Prescribed fire has been frequently used to promote early successional plant communities in managed pine forests. DeVos and Mueller (1993) noted that most of their principle brood-rearing habitats and high densities of arthropods were in fire maintained pine habitats with 50% bare ground, 50% brood-rearing cover, and approximately 40% overstory canopy coverage. Hurst (1972) pointed out that burned areas had a greater biomass of insects than unburned areas, and Speake and Sermons (1987) noted brood use of fire-managed pine woodlands. Unfortunately, the role and extent of prescribed fire in intensively managed pine ecosystems has been reduced (Brennan 1991). Therefore, alternatives or supplements to prescribed fire, such as selective herbicides, may provide a means of managing and/or improving brood-rearing habitats for bobwhites.

Palmer et al (2001) reported that imprinted chicks may provide a more biologically meaningful technique to estimate arthropod abundance than traditional techniques. Specifically, imprinted chicks were likely to sample arthropods in the physical space available to wild chicks, select arthropods more nutritionally suitable for wild chicks, and interact with environmental factors, such as vegetation structure, similarly to wild chicks. Therefore, using imprinted chicks may provide better estimates of arthropod response to manipulations aimed at bobwhite habitat improvement. Further, comparing this technique to availability of arthropods using conventional methods (i.e. sweep nets and pitfalls) may help provide a better estimate of arthropod use and availability.

My objective was to evaluate effects of different management activities used to manage brood-rearing habitats for bobwhites. Specifically, I evaluated effects of prescribed fire, fire in combination with herbicides, herbicides only, mowing, and no treatment on arthropods and vegetation. Arthropod use by imprinted chicks also was examined relative to management activities. Arthropod response was evaluated using 2 techniques. Imprinted chicks were used to determine use, and sweep nets in combination with pitfalls to determine availability. Vegetation response was measured in the context of structural variables known to be important to bobwhites.

Methods

Bobwhite chicks were human-imprinted following guidelines of Kimmel and Healy (1987) and Palmer et al. (2001). Imprinting was conducted immediately upon hatching by remaining close to chicks (within 0.5 m), covering the chicks with hands in a brooder containing a heat lamp, and hand-feeding chicks insects, for a minimum of 12 hours (Palmer et al. 2001). Chicks were exposed to a variety of habitats by allowing 20-30 chicks to forage for 30-60 minute periods twice a day in habitats similar to the treatment stands (i.e. herbicide, mowed, burned, and control stands not used for foraging trials).

Foraging trials were conducted with chicks 10-12 days old in May-August annually. Arthropods were withheld from chicks 18 hours before foraging trials and all food was restricted for 2 hours before the trials to clear crops and gizzards of arthropod parts and encourage foraging during trials (Palmer et al. 2001). Five chicks per brood were released into habitat types (~ 1 ha) managed with herbicide only, herbicide in conjunction with fire, mechanical manipulations (mowing), prescribed fire only, and non-

managed (control) sites (6 plots per treatment) between 0900 hrs and 1500 hrs. One brood was allowed to forage near the edge of each habitat and one brood was foraged near the center of each habitat plot. Handlers released broods, then remained within 2-5 meters of chicks. Handlers remained stationary when possible to avoid arthropod disturbance. Chicks were allowed to forage for 30 minutes, euthanized, then frozen to allow crop examination in the lab. All research was conducted under Louisiana State University Institutional Animal Care and Use Protocol AE 02-07.

Crop contents were rinsed using ethyl alcohol and water into a petri dish with 1cm² square grids. Each grid was searched twice using a 30x-dissecting microscope to estimate foraging efficiency within each habitat. Arthropods considered to be important to bobwhite chicks included: Araneae, Coleoptera, Diptera, Hemiptera, Homoptera, Hymenoptera, Lepidoptera, and Orthoptera (Handley and Cottom 1931, Hurst 1972, Jackson et al. 1987, Manley et al. 1994, Maidens and Carroll 2002). All orders comprising <1.0% of orders by count collected were grouped into 1 variable called other.

Arthropod response to each treatment was measured using 38 cm-diameter sweep nets and pitfall traps to provide a measurement of availability of arthropods in each treatment plot. Sweep net samples were conducted immediately after foraging trials were concluded by sweeping 3, 5 m transects. Arthropods collected were immediately frozen for later identification in the lab. Pitfall traps were made of 400-ml, 3-corner plastic beakers used to hold 250-ml plastic beakers trimmed to fit inside the larger beaker, with 1, 45.72 x 10.16 cm (18 x 4 in) aluminum sheet metal drift fence run from each corner of the plastic beaker, and a 30.48 x 30.48 cm (12 x 12 in) piece of aluminum sheet metal folded over the beakers to serve as a rain guard (Hooper-Bui and Pranschke 2003). Traps

were placed in the ground with the lip of the beaker flush with the ground. Ethylene glycol was placed in each pitfall to trap arthropods. One pitfall was placed in the approximate center of each plot, and one near each edge of the plot 180° from the center pitfall. Pitfall traps were operated for 2 consecutive days immediately following foraging trials to evaluate availability of species not likely to be sampled in sweep nets. Pitfalls were emptied during the evening, closed to avoid nocturnal arthropods, then re-opened again the next day.

Arthropods from crop contents, sweep nets, and pitfalls were identified to order using diagnostic fragments including heads, antennae, cerci, pronota, mandibles, femora, tarsi, tibiae, wings, and body segments (Moreby 1988). Only one insect was counted for all body parts identified. A count and estimated weight (g) based on length and width was determined for each chick following previously established guidelines (Greenberg and McGrane 1996, Palmer et al. 2001, Maidens and Carroll 2002). Arthropod abundance was determined by sweep nets and pitfalls and compared to availability of arthropod abundance determined by chick foraging trials.

Vegetation structure was assessed to determine microhabitat characteristics within each plot. Vegetation surveys were conducted the same day as chick foraging trials within each plot. Starting point for each survey was determined by a random azimuth and number of paces from the center pitfall trap (approximate center of treatment stand). Vegetation composition (% coverage) of grasses, forbs, vines, woody species, ferns, debris, and bare ground (Greenfield et al. 2002) were determined from plot center and each of the cardinal directions using a 0.5 m² Daubenmire frame (Daubenmire 1959). Canopy closure was measured with a forest densiometer (Lemmon 1956) to determine

light penetration to ground level. Minimum, maximum, and average visual obstruction readings (VOR) were determined using a Robel pole (Robel et al. 1970) to estimate vegetation height and density. The number of trees and diameter breast height (dbh) was recorded within a 10 m radius of plot center to assess tree density.

I used a randomized complete block analysis of variance (ANOVA) design to test response of arthropods and vegetation to the different management practices. Annual models were conducted using SAS system for windows (SAS Institute Inc. 1999) because of the aforementioned differences in management practices between years. The first model tested response of vegetation, canopy coverage, and VOR among treatments by year. The second model examined arthropod use and availability among treatments by year. Tukey's Multiple Comparison was used on significant effects to compare among treatments.

Results

2002

Vegetation Response

Vegetation characteristics differed among treatments for percentage woody ($F_{2,10}=29.97$, $P<0.001$) and percentage debris ($F_{2,10}=13.25$, $P=0.001$, Table 3.1). Minimum ($F_{2,10}=40.45$, $P<0.001$), maximum ($F_{2,10}=60.09$, $P<0.001$), and average vegetation height ($F_{2,10}=43.18$, $P<0.001$) differed among treatments (Table 3.1).

Table 3.1. Mean vegetation structural characteristics of vegetative variables that differed among treatments used to manage brood-rearing habitats for bobwhites on Jackson Bienville Wildlife Management Area, 2002-2003.

Year	Variable	Treatment				
		Burn only	Control	Mowed	Herbicide (burn year 2)	Herbicide (burn year 1)
		Mean(SE)	Mean(SE)	Mean(SE)	Mean(SE)	Mean(SE)
2002	% woody	N/A	64.53(3.14)a ^a	12.50(1.73)b	15.40(1.72)b	N/A
	% debris	N/A	13.03(1.29)b	41.27(1.59)a	48.48(0.46)a	N/A
	Minimum vegetation height (m)	N/A	0.85(0.07)a	0.09(0.004)b	0.25(0.004)b	N/A
	Maximum vegetation height (m)	N/A	1.31(0.02)a	0.28(0.01)c	0.84(0.02)b	N/A
	Average vegetation height (m)	N/A	0.95(0.07)a	0.15(0.003)c	0.43(0.01)b	N/A
2003	% woody	32.96(5.46)ab	57.40(0.85)a	27.44(2.36)b	16.60(0.17)b	24.88(3.70)b
	% debris	13.92(0.41)a	23.80(0.37)ab	37.36(0.81)b	20.23(0.53)ab	32.56(0.83)b
	Percentage fern	24.96(2.13)a	1.30(0.21)b	0.00(0.00)b	13.00(0.54)ab	10.88(1.90)ab
	Minimum vegetation height (m)	0.26(0.007)b	0.78(0.27)a	0.14(0.03)b	0.07(0.001)b	0.11(0.01)b
	Maximum vegetation height (m)	0.83(0.02)b	1.40(0.01)a	0.59(0.04)bc	0.48(0.01)c	0.46(0.08)c
	Average vegetation height (m)	0.44(0.01)b	1.00(0.19)a	0.28(0.03)b	0.21(0.01)b	0.27(0.04)b

^a Means across rows followed by the same letter do not differ, Tukey's HSD (P>0.05).

Arthropod Response

For pitfall traps, there was no difference in mean number of arthropods among treatments ($F_{2,46}=1.57$, $P=0.219$), and mean number of arthropods captured in sweep nets did not differ among treatments ($F_{2,46}=1.59$, $P=0.164$). Mean arthropod abundance per chick also did not differ among treatments ($F_{2,171}=0.63$, $P=0.680$), nor did mean biomass of arthropods (g) per chick ($F_{2,171}=0.61$, $P=0.691$, Table 3.2). Mean biomass consumed per chick across all treatments was 0.0024 g (SE= 0.00005).

2003

Vegetation Response

Vegetation characteristics differed among treatments for percentage woody ($F_{4,14}=5.74$, $P=0.007$), percentage debris ($F_{4,14}=6.31$, $P=0.004$), and percentage fern ($F_{4,14}=3.17$, $P=0.047$, Table 3.1). Minimum ($F_{4,14}=6.62$, $P=0.003$), maximum ($F_{4,14}=25.27$, $P<0.001$), and average vegetation height ($F_{4,14}=9.17$, $P<0.001$) differed among treatments (Table 3.1).

Arthropod Response

For pitfall traps mean number of arthropods differed among treatments ($F_{4,60}=2.69$, $P=0.039$), whereas mean number of arthropods captured in sweep nets did not ($F_{4,60}=0.92$, $P=0.459$, Table 3.2). Mean number of arthropods consumed per chick did not differ among treatments ($F_{4,221}=1.49$, $P=0.205$), nor did mean biomass of arthropods (g) consumed per chick ($F_{4,221}=1.59$, $P=0.178$). Mean biomass of arthropods consumed per chick across treatments was 0.015 g (SE=0.0009).

Discussion

The use of imprinted chicks requires accepting assumptions about the foraging behavior of bobwhite chicks (Palmer et al. 2001). The first assumption is that behavior of bobwhite chicks is innate. If this assumption proves to be false, then the use of imprinted chicks to estimate foraging indices is questionable (Palmer et al. 2001). The second assumption is that bobwhite chicks have an innate ability to select arthropods. Hurst (1972) stated that wild chicks should select the same types of arthropods as tame chicks. Palmer et al. (2001) found that foraging patterns and arthropod selection were similar between imprinted chicks and wild chicks. Thus, following the guidelines of Kimmel and Healy (1987) and Palmer et al. (2001), imprinted chicks were used to examine arthropod use by bobwhite broods on JBWMA. Imprinted chicks in this study selected similar types of arthropods as those assumed to be beneficial to bobwhites (Hurst 1972, Jackson et al. 1987, Welch 2000) (Table 3.3).

Hurst (1972) reported that insect abundance increases in areas after treatment with prescribed fire. However, a difference in arthropod consumption among treatments was not found in this study. Relatively few arthropods were consumed by chicks (mean/chick = 0.183-1.976) compared to other studies (mean/chick = 3-196) (Welch 2000, Palmer et al. 2001). Likewise, fewer arthropods were captured using sweep nets and pitfalls in my study (mean/sample = 2.722-20.483) compared to previous studies of arthropod abundance (mean/sample = 10.2-238.3) (Hurst 1972, Jackson et al. 1987, Welch 2000, Palmer et al. 2001, Maidens and Carroll 2002). This indicates that densely forested

Table 3.2. Mean number of arthropods per sample among treatments on Jackson Bienville Wildlife Management Area, 2002-2003.

Year	Treatment	Method		
		Chicks	Pitfalls	Sweep nets
		Mean(SE)	Mean(SE)	Mean(SE)
2002	Control	0.19(0.40)	20.56(9.64)	2.67(7.77)
	Mowed	0.25(0.43)	21.72(14.90)	5.06(3.53)
	Herbicide	0.18(0.25)	15.28(9.56)	2.72(5.51)
2003	Control	1.98(17.42)	14.08(18.87)	4.33(20.61)
	Mowed	1.34(7.49)	42.33(59.36)	3.60(3.83)
	Herbicide (burn year 2)	1.38(3.93)	16.08(6.97)	4.58(4.10)
	Herbicide (burn year 1)	1.12(3.04)	18.40(22.85)	2.33(2.02)
	Burn only	0.56(1.23)	9.60(6.01)	3.33(7.67)

Table 3.3. Percentage of arthropods selected by taxonomic order for foraging trials, pitfalls, and sweep nets on Jackson Bienville Wildlife Management Area, 2002-2003.

Order	Method		
	Chicks	Pitfalls	Sweep nets
Araneae	0.31	6.07	23.04
Coleoptera	0.92	0.29	2.53
Diptera	0.62	0.94	4.84
Hemiptera	2.48	0.12	0.46
Homoptera	0.00	27.72	12.90
Hymenoptera	94.73	57.93	50.92
Lepidoptera	0.00	0.04	0.00
Orthoptera	0.62	0.34	4.83
Other	0.32	6.55	0.48

settings may not have the arthropod availability and may not provide adequate amounts of arthropods needed by broods. Previous studies were set in old field habitats (Jackson et al. 1987), long-leaf pine ecosystems historically managed for bobwhites (Hurst 1972, Welch 2000), or agricultural landscapes (Palmer et al. 2001, Maidens and Carroll 2002). My findings suggest that intensively managed forest landscapes similar to JBWMA do not provide high quality brood-rearing habitats for bobwhites.

The difference in abundance of arthropods in pitfalls in 2003 may have been the result of the timing of treatments. During 2002, mowing was conducted on 15 May, whereas in 2003 mowing was conducted on 22 April. The earlier mowing in 2003 may have allowed greater arthropod response prior to sampling. Likewise, stands treated with prescribed fire only during 2003 were burned during the early portion of the growing season (April-May). Although sampling did not occur within these stands until vegetation had begun to recover, arthropods may not have had sufficient time to respond to treatment, explaining low invertebrate abundance in burned stands relative to other treatments. Swengel (2001) found that many groups of invertebrates decline immediately after fire. Anderson et al. (1989) captured greater numbers of arthropods on unburned sites than sites that had been burned during that year, but subsequently found more arthropods on burned sites. Jones (2003) noted substantial differences in arthropod abundances between stands treated with dormant season fire and herbicides.

There also remains uncertainty about the comparison of abundance of arthropod estimates among methods. Past research has shown high variability in estimates and it has been pointed out that some methods may overestimate arthropod abundance (Byerly et al. 1978, Schotzko and O'keeffe 1986, Mommertz et al. 1996). Sweep nets may bias

studies by sampling arthropods inaccessible to bobwhite chicks. Pitfall traps, on the other hand, may only capture arthropods incapable of scaling drift fencing. Future studies using imprinted chicks in comparison to pitfalls and sweep nets across treatments or habitat types may reveal a preferred method of sampling abundance of arthropods, unfortunately this study could not provide a meaningful answer to this question.

Also, Southwood (1978) noted that efficiency of sampling arthropods changes across differing habitats. Discordance among arthropod abundance and imprinted chicks may increase as changes in plant diversity and arthropods across habitat patches are included in comparisons (Palmer et al. 2001). Several attempts have been made to compare preferred foods with abundance of arthropods on game birds; however, much of this work occurred in 1 habitat type (Potts 1986, Sotherton 2000, Palmer et al. 2001). While habitat structure was very similar in this study across plots, management techniques resulted in different understory composition.

Previous studies have quantified the proportion of vegetation characteristics required by bobwhite chicks (DeVos and Mueller 1993, Burger et al. 1994, Taylor and Burger 2000). These estimates included a greater than average amount of broad-leaved herbaceous vegetation, grasses, and shrubs, with 19-61% bare ground, and 50% brood cover. While a reduction in the amount of woody vegetation and the height of vegetation improved post-treatment, no substantial improvement in the amount of grasses, forbs, or bare ground occurred for any treatment. Vegetation characteristics in this study may explain the lower abundances of arthropods found as compared to other studies.

Past research has shown that selective herbicides can be used to manage habitats for bobwhites (Guthery et al. 1987). Welch (2000) found an increase in the amount of

forb coverage with the application of Arsenal. Jones and Chamberlain (2004) found that a combination of Arsenal and prescribed fire improved habitat conditions for nesting and brood-rearing bobwhites. While some vegetative characteristics did improve with the application of Arsenal, along with other manipulations in this study, vegetation suggested as crucial to brood rearing habitat did not improve. Arthropod response also did not improve with the application of Arsenal; however, the short term nature of this project may not have been able to detect any real differences. Consistency in the timing of management activities, in the form of prescribed fire and mowing, may provide a better estimate of arthropod abundance over time. Since prescribed fire was not applied to any stands until 2003, arthropods may not have responded by the time of sampling.

Dunwiddie (1991) noted that arthropods were more abundant 2 growing seasons post burn than in control plots. Jones and Chamberlain (2004) found that during the second growing season post-treatment, herbicides were more effective than prescribed fire alone at improving vegetative characteristics and arthropod diversities for brood-rearing bobwhites. Further sampling efforts one-year post burn may reveal vegetation and arthropod response to be significant.

CHAPTER 4: CONCLUSIONS AND MANAGEMENT IMPLICATIONS

Managers in the southeast have historically relied on prescribed fire to manage for bobwhites. However, a reduced role of prescribed fire has led to a decline in bobwhite populations (Brennan 1991, Droege and Sauer 1990, Burger 2002). Bobwhites depend on early successional habitats for nesting, brood-rearing, and escape cover, and without disturbance these habitats are lost through natural succession. Other forms of manipulation have been suggested to help combat succession, including mowing, clear-cutting, thinning of forest stands, and the use of selective herbicides. However, direct effects of these methods on bobwhite populations have not been fully examined.

I found that bobwhites were closely associated with early successional habitats on the densely forested landscapes of JBWMA. Conversely, bobwhites also were negatively associated with 16-29 year old pine stands. These dense pine stands have high basal areas with little or no understory vegetation, offering little or no benefit to bobwhites. These findings occurred across multiple scales at the landscape and class-scale, indicating the importance of early successional habitats in predicting the occurrence and distribution of bobwhites.

On heavily forested landscapes, managers need to recognize that bobwhites may only be present on a portion of the landscape containing early successional habitats. Guthery (1997) noted that management practices aimed at increasing usable space should result in increased mean density of bobwhites. The usable space hypothesis, as proposed by Guthery (1997), states that in order for habitats to be fully usable they must be compatible with bobwhite's physical, behavioral, and physiological adaptations in a time-unlimited sense. In my study, landscape models indicate that usable space on JBWMA

consisted primarily of early successional habitats. Improving usable space for bobwhites can not be viewed on a patchwork scale, but must be viewed across the landscape. As bobwhite populations become more isolated they may become more vulnerable to demographic and stochastic processes (Burger 2002). Also, Roseberry (1993) noted that the viability of local populations depends on certain spatial and temporal characteristics of neighboring patches. Thus, in order to manage for bobwhite populations in forested systems, creation of usable space in the form of early successional habitat must be conducted across the landscape.

Selective herbicides, prescribed fire, and mowing were used in an attempt to create more early successional habitat on JBWMA. Of primary concern was creating suitable brood-rearing habitat. Through the use of foraging trials, pitfall traps, and sweep nets, I found that these manipulations in general did not improve habitat quality for brood-rearing. Each sampling technique used in this study found fewer arthropods deemed crucial to the growth and survival of bobwhite broods than previous studies (Welch 2000, Maidens and Carroll 2001, Palmer et al. 2001). Vegetation data noted improved habitat structure, but vegetative conditions fell short of previous estimates of brood-rearing needs (DeVos and Mueller 1993, Burger et al. 1994, Taylor and Burger 2000). I found a reduction in the amount of woody vegetation and vegetation height, but no improvement in the amount of grasses, forbs, and bare ground needed for brood-rearing (Burger et al. 1994).

I found that imprinted chick behavior and selection of arthropods was similar to wild chicks. Imprinted chicks in this study selected similar types of arthropods found by others (Hurst 1972, Welch 2000, Palmer et al. 2001); however, I found chicks did not

consume the quantity reported in previous studies. This reflects the inability of intensively managed forest landscapes to provide arthropods needed for bobwhite chicks to survive.

It should be noted that sampling of these habitats took place within weeks of manipulation by burning and mowing. Previous studies have shown an increase in number of arthropods 1 and 2 years post treatment (Anderson et al. 1989, Dunwiddie 1991, Jones and Chamberlain 2004). Thus, habitat enhancement that occurred in this study may show marked improvements in future years.

As pine-dominated forest landscapes continue to increase across the southeast (Trani et al. 2001, Burger et al. 2002), along with a reduction in the application of prescribed fire (Brennan 1991), bobwhite populations continue to decline. To combat this problem, human intervention with disturbance regimes are needed to restore plant communities beneficial to bobwhites (Burger 2002). Drastic improvements, however, may not be achieved in the short term. Improving habitat quality for bobwhites may require intensive management efforts applied over several seasons.

Selective herbicides and mowing can improve habitat quality for bobwhites; however, arthropods needed by broods did not respond to these management practices on JBWMA. Although herbicides and mowing did increase the structure of the landscape, the functional role of the landscape remains to be seen. Historically, this region provided suitable habitat for bobwhites, but intensive silviculture and natural succession has reduced the quality of the region. Researchers and managers need to continue to put forth the effort to find practical methods of improving habitat quality in densely forested landscapes. Consistent management regimes over several years should be used to

understand the role management practices on JBWMA has on bobwhite population processes. This project was the first 2 years of a 4 year project looking at the effects of restoring habitat quality for bobwhites using selective herbicides, mowing, and prescribed fire. Future efforts should continue to look at stand manipulated in this study to measure the long-term effects of these restoration efforts, as well as continue the efforts started by this study.

As this study has shown, early successional habitats are vitally important for bobwhites to survive in intensively managed forest landscapes. It may take a combination of several techniques to see vast improvements in habitat quality for bobwhites. Managers need to consider using clear-cuts, thinning of pine stands, mowing, and the application of selective herbicides to manage for bobwhites in pine dominated systems.

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VITA

Charles Lynn Kitts was born in Knoxville, Tennessee, 21 September 1973, to Luther J. and Thelma Lucille Kitts. He grew up in the mountains outside Knoxville, where he learned to hunt and fish from his older brothers Allen Kitts and Tim Kitts. He attended Clinton High School and upon completion of his high school equivalency degree, started attending Roane State Community College in nearby Oak Ridge, Tennessee. After 2 semesters he transferred to the University of Tennessee at Martin where he became interested in upland gamebird ecology. During the summer of 1998 he took an internship with Tall Timbers Research Station in Tallahassee, Florida, where his interest in northern bobwhite quail and longleaf ecosystems grew. He received his Bachelor of Science in natural resource management in 1999.

After graduation he took a technician position with the University of Georgia working with bobwhites in the piney woods of east Georgia. He also worked as a research technician for Auburn University. In 2001 he returned to the longleaf ecosystem by accepting a research technician position with the Joseph Jones Ecological Research Center in Newton, Georgia, where his interest in bobwhites continued. In January 2002 he was offered a graduate research position at Louisiana State University working with bobwhites under Dr. Michael Chamberlain. The degree of Master of Science in wildlife will be awarded during spring 2004.