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I am submitting herewith a thesis written by Troy Lee Ettel entitled "Predation of Artificial Nests in Grassland/Shrubland Fragments in Western Tennessee." I have examined the final 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.

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PREDATION OF ARTIFICIAL NESTS IN GRASSLAND/SHRUBLAND HABITATS IN WESTERN TENNESSEE

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ABSTRACT

Grassland and early successional plant communities formerly comprised important components of the southeastern landscape. Many early successional communities have disappeared in the absence of maintenance mechanisms to preserve their existence in the landscape. Today, avian species of grassland and early successional communities are experiencing dramatic declines throughout their range (Askins 1993). Declines in the southeastern United States may be largely related to habitat loss.

I focused on two habitats representative of grassland/shrubland successional stages that currently exist throughout the Southeast, old fields and open woodlands. Eight study plots of approximately 1.89 ha each were selected, 4 in each habitat type, on Ames Plantation near Grand Junction, TN. Avian communities were censused on the study plots to identify species presence and relative abundance. Artificial nests were used to gather information on relative predation trends within the old-field and open-woodland habitats. Four study plots were randomly selected as experimental plots (2 field and 2 forest plots) on which rodent populations were reduced to determine the effects of rodents on artificial nest success.

Avian communities mostly consisted of mixed-habitat species, capable of inhabiting several different habitat types (red-winged blackbird, *Agelaius phoeniceus*, indigo bunting, *Passerina cyanea*, and eastern towhee, *Pipilo erythrophthalmus*). Several species characterized as early successional habitat specialists (yellow-breasted chat, *Icteria*

virens, northern bobwhite, *Colinus virginianus,* and field sparrow, *Spizella pusilla*) were present on the study area. Grassland specialists were rare (one nesting pair of eastern meadowlark, *Sturnella magna*, in two years) despite available habitat and may result from the high area-sensitivity of most grassland birds in conjunction with the relatively small size of available habitat patches on the study area.

Kill-trapping of rodents decreased rodent populations below pre-treatment levels on experimental plots ($P = 0.001$). Trapped areas experienced lower artificial nest predation by rodents and greater nest success than untrapped plots (nests containing finch eggs, $P = 0.002$, nests containing bobwhite eggs, $P = 0.001$). Predation by non-rodents remained relatively stable between treatments, despite rodent removal. Therefore, there appeared to be no compensatory response to rodent removal in terms of increased predation on artificial nests by non-rodents.

Rodents were responsible for the majority of predation, predating more nests than all other sources combined (64.8% of all nests predated). The most abundant rodents on the study area were the white-footed mouse (*Peromyscus leucopus*) and the hispid cotton rat (*Sigmodon hispidus*). Predation of artificial nests by medium-sized mammals (striped skunk, *Mephitis mephitis*, opossum, *Didelphis virginiana*, and raccoon, *Procyon lotor*) was infrequent, accounting for approximately 7.0% of all predation.

Predation on nests containing finch eggs occurred with greater frequency than predation on nests containing bobwhite eggs ($P = 0.001$). Because rodents were the predominate nest predators, this result supports the hypothesis that small-mouthed predators may be less capable of biting into larger eggs such as northern bobwhite

(Reistma et al. 1990, Roper 1992, Haskell 1995*a,b*). The use of large eggs such as northern bobwhite in artificial nest experiments may reduce the likelihood of predation by small rodents.

Predation rates differed with habitat type, as forested plots experienced much greater predation than field plots (P= 0.001). Nests placed within dense vegetation were less likely to be predated than nests placed in sparser vegetation (P = 0.006). Nests placed in grass were less likely to be predated than nests placed in woody vegetation (P = 0.012) or forbs (P = 0.022).

Land managers may succeed in sustaining populations of early successional nesting birds by addressing nesting requirements with habitat manipulations. Habitat improvements that increase quality and composition of nesting cover may reduce predation rates. Future research needs include development of management strategies that benefit the variety of avian species using grassland and early successional habitats.

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INTRODUCTION

Reported declines of songbirds in the eastern United States have received much attention over the past decade. Most studies have focused on forest migrants, and have partially attributed population declines to increasing amounts of nest predation and parasitism resulting from fragmentation of the eastern deciduous forest (Brittingham and Temple 1983, Wilcove 1985, Robinson et al. 1995). Recently, research interest has shifted to focus more on birds of early successional habitats. Whereas population trends for forest migrants are somewhat inconsistent across regions and forest types, trends for early successional species are far more dramatic and distinct. Relative abundance of grassland birds has declined as much as 85-90% since the mid-1960's (Robbins et al. 1986, Heckert 1991). Presently, many habitats that formerly fulfilled nesting requirements of early successional species have disappeared from the landscape (Bollinger et al. 1990).

Grasslands are often associated with the American West, manifested in Great Plains and midwestern tallgrass prairies. However, in the southeastern United States, grassland habitats once extended from Arkansas, through Kentucky to Virginia and southward to Texas and Florida (Deselm 1994*a*). These grasslands were floristically related to the midwestern tallgrass prairie and covered an estimated 38,000 km^2 in the southeastern United States (Deselm and Murdock 1993). This estimate is almost exclusively confined to an extensive network of upland sites and does not take into

account southern pine savannas or the salt marshes of the Coastal Plain. The upland grasslands, termed "barrens" by early European explorers, existed as small to large islands in forested landscapes (Deselm and Murdock 1993). Barrens were described as grassy openings as well as low density woodland, thickets, savanna, and open woodlands with a grass understory, thus providing a wide range of early successional habitat (Deselm 1989). Barrens were probably maintained by a combination of natural and anthropogenic disturbances including periodic drought (frequently occurring throughout many areas of the Southeast), grazing-browsing-trampling by large herbivores, and both human- and lightning-caused fire (Deselm 1994*a*).

Tennessee lies between two major grassland areas, the large Kentucky Barrens which extended just across the border into Tennessee and the Black Belt of Alabama and Mississippi. Over 120 barren sites have been located in Tennessee since 1954, even though evidence of most Tennessee barrens probably disappeared from the landscape long before this time period (Deselm 1994*b*). The arrival of white settlers in Tennessee in the late 1700's and early 1800's brought about the elimination of large, wild mammalian herbivores such as elk and bison (scientific names for all mammals mentioned in the text can be found in Appendix A). Barrens vegetation, however, flourished and expanded its range as settlers cleared the forests and maintained early successional habitat with regular burning (Deselm 1989). During this early period, forests were burned in spring to enhance new forage growth for free-ranging livestock (Killebrew 1897). Open burning was outlawed in 1917 but open grazing persisted until 1947 when fencing of livestock was required by law (Tenn. Public Acts 1917, 1947). In the absence of fire and grazing by

large herbivores, the presence of barrens and barrens vegetation on the landscape decreased through natural succession. Many barrens succeeded into forested habitat while others were converted to agricultural lands, including pasture and cropland (Deselm and Murdock 1993). Initially, livestock grazing may have helped maintain barrens, however, following World War II, pasturage was widely planted in fescue (*Festuca pratensis*), an exotic, highly competitive cool-season grass that eventually replaced most native barren vegetation (Deselm 1994*a*).

Declines in avian species of early successional habitats may mirror habitat loss. Between 1966 and 1991, results from the Breeding Bird Survey for eastern North America indicated that 16 species of open and wooded grassland and 12 species of shrubland habitats experienced decreasing population trends (Askins 1993). Following the loss of natural grassland and shrubland habitat, small farms provided islands of habitat for early successional birds. In the early 1900's, typical farmland provided unimproved pastures consisting largely of native vegetation, grassy field borders, drainage ditches, and road rights-of-way as well as brushy hedgerows. Prior to the 1950's, little change in bird populations was detected in some regions (Warner 1994). By the 1960's, mechanization and the development of synthetic fertilizers brought intensified farming practices. As production of corn and soybeans increased and patterns of ownership changed from smaller to larger farms, grassy field borders and hedgerows were removed to create larger fields (Askins 1993, Warner 1994). Hayfields initially provided refuge for many birds. However in the past 50 years, hay harvesting regimes have undergone changes to increase yields. Cultivars of several haycrops have been developed to allow earlier and more

frequent cutting. Hayfields are now cut up to two weeks earlier in the season and up to 4-5 times per year (Warner and Etter 1989). This has resulted in increased mortality of the nests and young of grassland birds nesting in hayfields (Bollinger et al. 1990).

Declines of grassland and early successional birds east of the Mississippi River have been significant and habitat loss has been dramatic as a result of changes in land-use (Askins 1993). Research is critically needed to assess the habitat requirements of remaining avian populations. Specific questions that must be addressed include: 1) what type of grassland/early successional habitats currently exist throughout the Southeast; 2) what bird species are utilizing these habitats; and 3) are existing habitats adequately fulfilling the needs of nesting birds in terms of cover, structure, composition, and size? By identifying avian communities and relative abundance of species, management actions can be prescribed and the needs of priority species addressed.

The effects of nest predation and the size of existing habitat patches may differentially affect avian species groups. Several avian guilds with differing life histories overlap in habitats of the southeastern United States. These include ground-nesting birds, shrub-nesting birds, grassland specialists, shrubland specialists, and species of mixed habitats. A comparison of predation effects and habitat use would reveal information on predator communities threatening these avian species groups and facilitate creation of habitat management strategies that would be beneficial to avian conservation goals.

OBJECTIVES

This thesis presents results from a field study of grassland/shrubland birds on Ames Plantation, Grand Junction, Tennessee. I focused on factors influencing nest predation, to help address the adequacy of early successional habitat to breeding birds. My goals were to identify what avian species were present and gather information on both the intensity and source of avian nest predation within existing early successional habitats. Specifically my objectives were:

1) Identify avian species using grassland/shrubland habitats on Ames Plantation.
2) Determine if nest success can be increased empirically by reducing a predator population.
3) Identify the suite of nest predators potentially affecting nest success.
4) Determine habitat parameters related to the incidence of nest predation.

Objective 1 will be addressed in Chapter 1. Objectives 2 and 3 will be addressed in Chapter 2 and objectives 3 and 4 will be discussed in Chapter 3 of this thesis.

STUDY AREA

All field work was conducted on Ames Plantation, a 7527-ha University of Tennessee Agricultural Experiment Station located in Fayette and Hardeman counties in southwestern Tennessee (Figure 1.1). Ames Plantation lies approximately 70 km east of Memphis, TN and 16 km north of the Tennessee-Mississippi state line. The terrain is mostly rolling, with an average altitude of 122 m above sea level. Ames Plantation is located within the watershed of the North Fork of the Wolf River and many streams

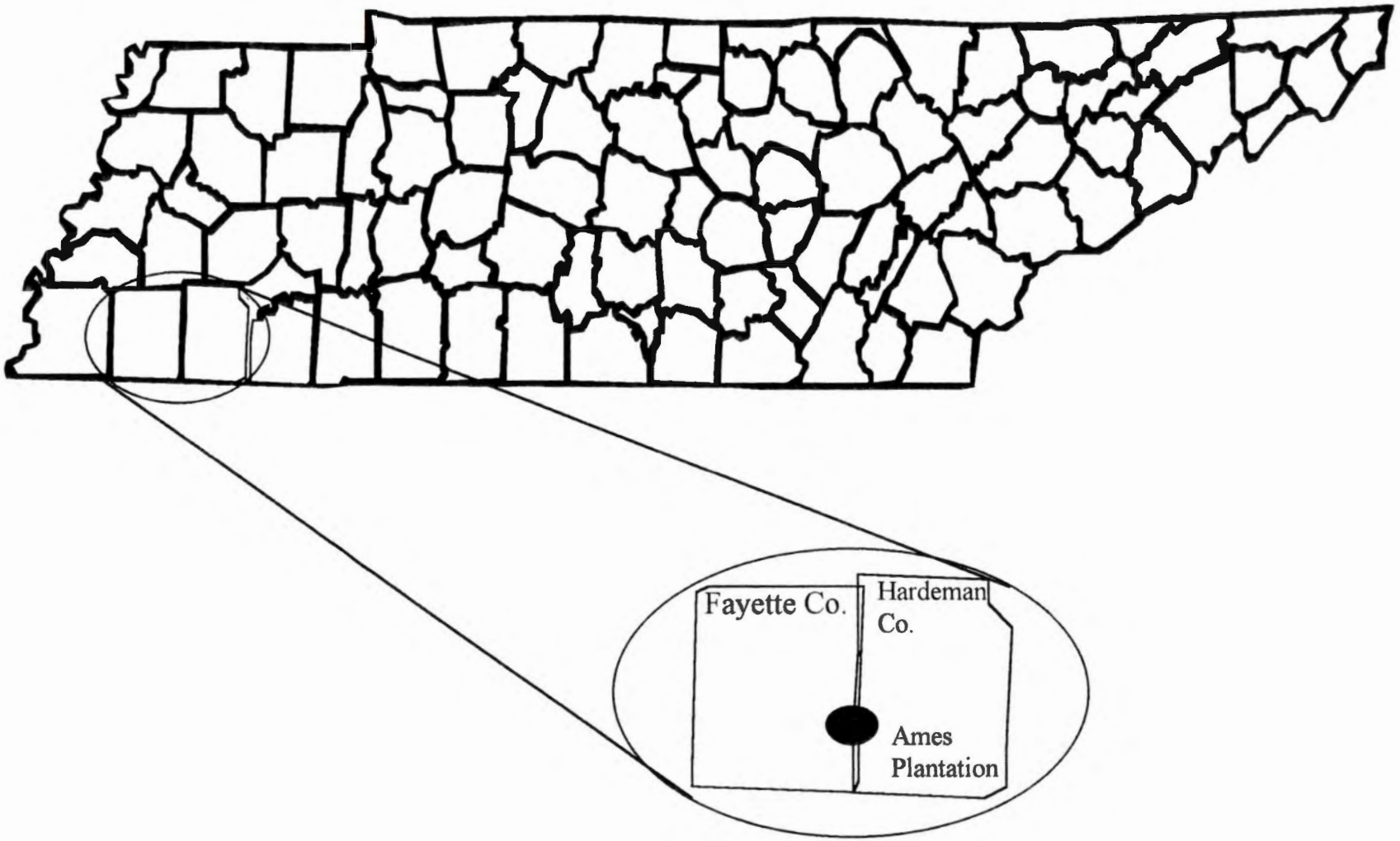

Figure 1.1. Location of University of Tennessee's Ames Planation Experiment Station, Grand Junction, Tennessee.

dissect the landscape, a large portion of the smaller ones consisting of intermittent waterways (Flowers 1964). Ames Plantation lies within the East Gulf Coastal Plain physiographic region, an area characterized by acidic soils in loess of varying thickness interspersed with sandy Coastal Plain material (Flowers 1964). Annual precipitation averages approximately 134.9 cm and average temperatures range between a minimum of 18-21° C to a maximum of 32° C in summer and 0-23° C in winter (Flowers 1964).

Ames Plantation is the site of the annual National Championship Field Trial for bird dogs. Approximately 2000 ha are split into two courses for the field trial. The field trial venue is actively managed for visibility and to provide habitat for northern bobwhite (see Appendix A for a list of scientific names for all birds mentioned in the text) through maximum interspersion of several different habitats. Forested habitats (both pine and hardwood), open fields, and cultivated fields and strips, are interspersed throughout the field trial courses in patches of varying size and configuration. Habitats are not contiguous, but patch isolation is not extreme due to the interconnected nature of the field trial grounds. Habitat patches are often found in close proximity (varying from adjacent to several hundred meters) to other habitat patches of similar vegetative composition and structure.

Cultivated fields are mostly rowcropped in soybeans and corn and forested areas are managed for timber production and include loblolly pine plantations (scientific names for all trees and other woody vegetation can be found in Appendix B). Open fields consist of both fallow fields (fields plowed but not planted in the current or previous year) and

old fields (fields consisting of a mixture of grasses, forbs, and shrubs maintained by some form of periodic disturbance). The field trial courses are burned on a three-year rotation and wildlife food plots are planted primarily along field and wood edges and through the centers of larger patches to provide feeding areas and brood habitat for bobwhite. Food plots are usually planted in soybean, corn, milo or millet or are left unplanted to provide brood habitat and dusting areas for bobwhite. Bushhogging is conducted on the courses throughout the spring and summer months to maintain field trial routes. Additionally, bushhogging of portions of the field trial courses is conducted in late fall and early winter to increase visibility for the field trial.

Two major types of grassland habitat exist on the field trial courses. In many areas, open fields provide grassland habitats comprised of a mixture of grasses and forbs with scattered patches of hardwood brush and shrubs (Figure 1.2). Between 1965 and 1969 much cropland formerly planted in cotton was converted to loblolly pine to help halt erosion and provide a future source of income. Thinning and burning in portions of these forests has resulted in understocked, even-aged stands with relatively open canopies, principally dominated by loblolly pine and red oak species. These openings allow development of an understory of hardwood brush and shrubs interspersed with grasses, providing a second type of grassland habitat available on the field trial courses (Figure 1.2). I will refer to this habitat type as "open woodlands."

I selected 8 study plots from the field trial courses, including 4 of each habitat type (Figure 1.3). All plots measured approximately 1.89 ha. My plots were situated in some of the larger habitat patches available on the field trial grounds. I was unable to measure

A.)

B.)

Figure 1.2. Open pine/hardwood forest (a), old-field (b), study areas, Ames Plantation, Grand Junction, Tennessee, 1996-1997.

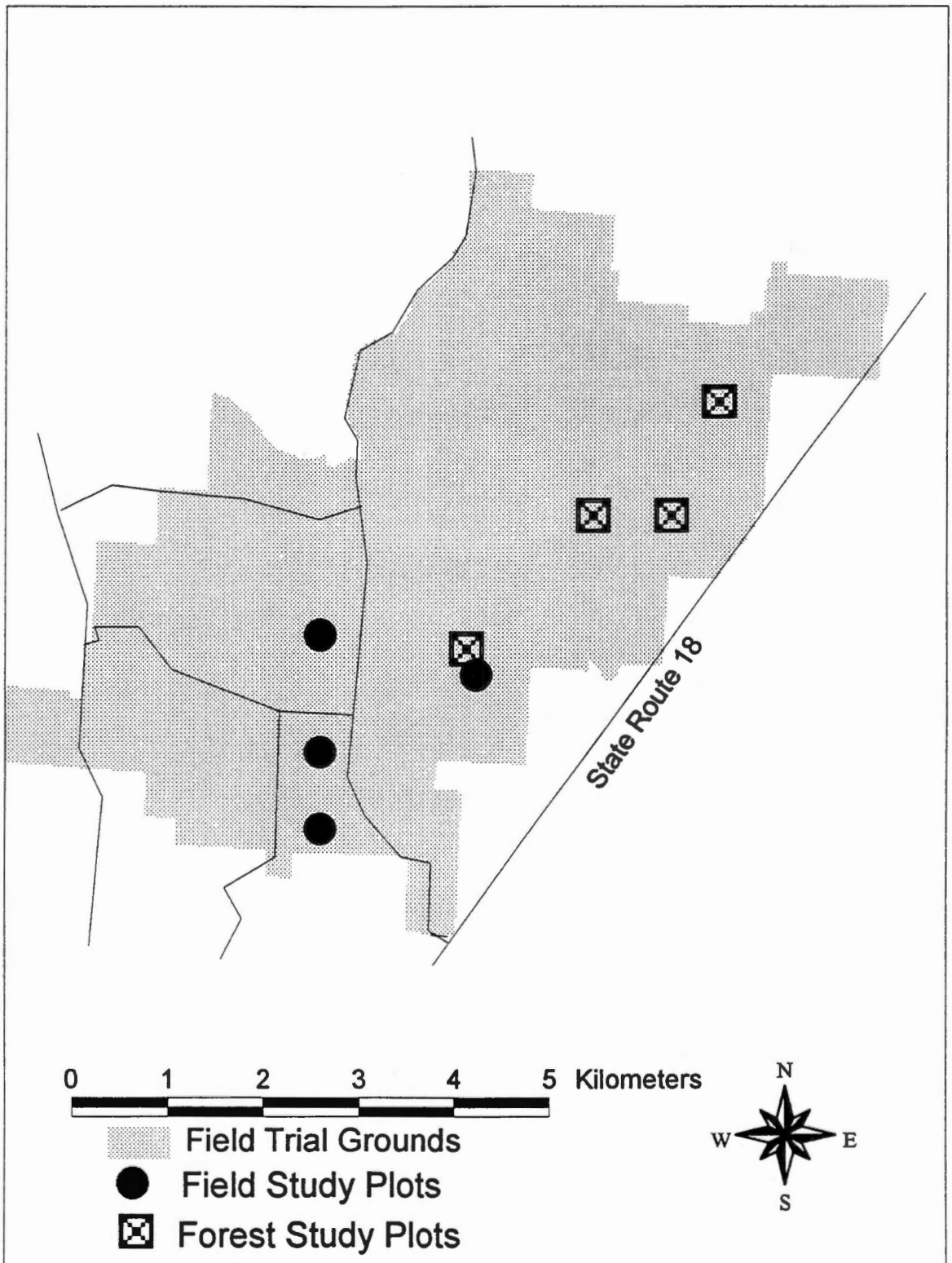

Figure 1.3. Location of forest and field plots, Ames Plantation, Grand Junction, Tennessee, 1996-1997.

directly the effects of habitat fragmentation because medium to large grassland patches were lacking. The greatest distance within any field or forest plot to habitat edge was only 75 m. Open fields containing the field study plots ranged from 1.89 ha. to 5 ha. Forest plots were adjacent to other woodlots or were linked by narrow corridors to other forested strips but were not large tracts of forest. The plot size of 1.89 ha was retained to preserve size consistency among plots. A grid was laid out on each plot using orange flagging to mark grid intersections. Each intersection was separated from the next adjacent intersection by a distance of 30 m, hence each grid cell measured 30 x 30 m (0.09 ha).

A general habitat description was conducted on each plot to characterize vegetation. A random sample of 10 points was selected from among the grid intersections available on each plot. Dominant vegetation was identified and ranked according to abundance at each sampled point. For forested plots, a prism was used to identify trees within the sampling area and dbh was measured on each sampled tree to estimate species composition and basal area of the plot.

Field plots consisted of a mixture of grasses and forbs, with occasional patches of shrubs and hardwood brush. All field plots could be classified as old fields or approaching old-field status, as all had been removed from cultivation for ≥ 3 years. Fields were burned on a three-year rotation, and were often bushhogged in late winter to improve visibility during the National Championship field trial. Field plots were dominated by forbs with grasses scattered throughout. Common forbs included species of goldenrod, ragweed, and lespedeza (Appendix B contains scientific names for all herbacious species

collected or recorded). Common grasses included several native species (*Andropogon* spp.) as well as exotic species such as Johnson grass and fescue.

Forested plots were relatively open-canopied stands dominated by loblolly pine and red oak. The first year following a burn, grasses were prominent on forested plots. By the second year post-burn, hardwood brush and shrubs typically dominated the sites with grasses much more restricted in distribution (Appendix B contains scientific names for all trees and woody plants collected or recorded). Forest plots varied in their relative amounts of open canopy with estimated basal areas ranging between 12.6 - 18.1 m^2/ha.

Chapter 1: Community composition of avifauna nesting in grassland/shrubland habitats

INTRODUCTION

Most avian species exhibit preferences for specific habitat conditions. This may vary from extreme habitat specialization, e.g. Kirtland's warbler, which nests only in a small region in Michigan where it is restricted to a specific seral stage of regenerating jack pine, to habitat generalization, e.g. the red-winged blackbird, which is commonly classified as a wetland or marsh bird but is also one of the most abundant breeding birds of tallgrass prairie (Ricklefs 1990, Herkert 1991).

Many avian species are particularly sensitive to habitat patch size. Groups of birds that nest in grassland, forest, and shrub habitats have all been found to demonstrate area-sensitivity. For example, upland sandpipers generally will not nest in grassland tracts less than 30-50 ha (Herkert 1991, Vickery et al. 1994). Several studies have shown cerulean warblers to be associated with large, contiguous forest tracts (Robbins et al. 1989). In shrubland/early successional habitat, golden-winged warblers usually select habitat >10 ha for nesting (Askins 1993). Species with such highly developed area requirements would be particularly sensitive to landscapes comprised of small habitat patches.

In addition to high-sensitivity to patch size, many species are sensitive to habitat structure and composition. Henslow's sparrow is particularly sensitive to vegetative structure, requiring greater vegetation height, density, and litter depth than most grassland species (Wiens 1969). The grasshopper sparrow, another grassland species, is most abundant in low, sparse vegetation (Swanson 1996). Yellow-breasted chats are extreme specialists of shrubland habitats, preferentially selecting areas with dense shrub cover and avoiding areas with a developed canopy or predominating grass cover (Askins 1993).

Because of the specific habitat preferences of many birds, the size and vegetative structure of available habitat patches may influence species presence. Large grassland preserves can be managed to provide a mosaic of patches of different successional stages to meet the needs of several nesting species (Askins 1993). In Tennessee, grassland and early successional habitats consist of relatively small habitat patches within landscapes dominated by agriculture and other extractive land uses such as forestry and surface mining. Avian species exhibiting highly developed area-sensitivity may be absent from areas consisting of complexes of small habitat patches. However, small grassland patches can provide important habitat for mixed-habitat species, whose nesting requirements are not associated with a single habitat type, but may be satisfied in several different habitats (Askins 1993).

I censused old-field and open-woodland habitats on Ames Plantation to record the composition of the avian community. Despite the area-sensitivity of many grassland birds, I focused upon small patches of grassland habitat, which are more representative of existing grassland habitats throughout the state of Tennessee. My specific objective was

to determine which species were present within these habitats and, particularly, which breeding birds were utilizing the habitats.

METHODS

I used a spot-mapping technique to identify avian species nesting in old-field and open-woodland habitats on Ames Plantation (Kendeigh 1944). The spot-mapping technique was chosen because of the small size of habitat patches and because it is generally acknowledged as the most accurate avian census technique (Robbins 1978). Spot-mapping was conducted from 11 May - 29 June in both 1996 and 1997, which falls within the period when breeding birds are most active in the Southeast (Robbins 1978).

Ten spot-map censuses were conducted each year for each of the 8 study plots. Spot-mapping was conducted within 3 hours after sunrise and the order of plots censused was alternated among days (Robbins 1981*a*). Spot-mapping consisted of a systematic walk along the established grids on each plot. Starting and ending points were rotated between censuses on a particular plot. All birds seen or heard within the plot were recorded by noting each bird's approximate location on a map of the plot (Kendeigh 1944). Whenever possible, species, sex, and activity of each bird were recorded. Spot-mapping was not conducted during conditions of heavy winds or rain because of reduced visual and auditory detectability of birds during weather extremes (Robbins 1981*b*).

All spot-maps from each plot were compiled and territories mapped for individual males of all species detected on the plots. A total number of territories was determined for each species. To attain a density estimate, the total number of territories for each

species was divided by the total area censused for each habitat type. Abundance was not extrapolated to number of territories per 40 ha as suggested by National Audubon Society guidelines because of the small size of the plots (1.89 ha).

RESULTS

Thirty-four species of birds were detected within censused forest plots and 18 species were detected within field plots. Fifteen species of birds established breeding territories within or across portions of the forest plots. These included 8 ground- or shrub-nesting species and 7 species of mid-story or canopy nesters (Table 1.1). Ten species established breeding territories including portions of field study plots, including 9 ground- or shrub-nesting species and one mid-story/canopy nester. Indigo buntings, red-winged blackbirds, field sparrows, and to a lesser extent common yellowthroats were the most abundant birds of field habitats. Eastern towhees and indigo buntings were the most abundant species of open woodlands, with yellow-breasted chats, northern cardinals, Kentucky warblers, and common yellowthroats present to a lesser extent. Midstory/canopy nesters were comprised mostly of eastern wood pewees and summer tanagers.

Several wide-ranging species were detected during spot-mapping, including woodpeckers (red-headed woodpecker, red-bellied woodpecker, pileated woodpecker, downy woodpecker, and hairy woodpecker), blue jay, American crow, wild turkey, and northern bobwhite. Although frequently detected, the wide-ranging habits of these species prevented abundance estimation using the spot-mapping method (Robbins 1978).

Table 1.1. Total number of avian territories per ha of available habitat on forest and field census plots, Ames Plantation, Grand Junction, Tennessee 1996-1997[1].

	1996		1997	
	Forest (7.56 ha)	Field (7.74 ha)	Forest (7.56 ha)	Field (7.83 ha)
Ground- and shrub-nesters				
BLGR	0.000	0.065	0.000	0.170
CAWR	0.000	0.000	0.132	0.000
COYE	0.154	0.323	0.220	0.585
EAME	0.000	0.065	0.000	0.000
EATO	1.212	0.000	0.739	0.000
FISP	0.066	0.672	0.132	0.843
INBU	1.124	0.915	1.014	0.989
KEWA	0.265	0.000	0.132	0.000
NOCA	0.099	0.000	0.364	0.021
RWBL	0.000	1.066	0.000	0.383
YBCH	0.231	0.000	0.320	0.000
Mid-story and canopy nesters				
AMGO	0.000	0.099	0.132	0.000
BGGN	0.176	0.000	0.265	0.000
EAKI	0.000	0.000	0.099	0.085
EWPE	0.331	0.000	0.496	0.000
GCFL	0.000	0.000	0.143	0.000
OROR	0.000	0.183	0.000	0.085
REVI	0.265	0.000	0.000	0.000
SUTA	0.441	0.000	0.364	0.000

[1]Common and scientific names and 4-letter codes for avian species in Appendix A.

Northern bobwhite, particularly, were frequently recorded on both forest and field plots. Several bobwhite nests were found within or near field study plots and at least 2 were found within forest plots, although many were inactive and empty when found. Therefore, bobwhite would rank as a relatively common nesting bird, particularly in field habitat. Additionally, wild turkey were seen on the study areas and wild turkey nests were found within plots of both habitat types. These birds should be included as breeding birds, though no discreet territories were mapped.

Several nests of ground- and shrub-nesting birds were found incidentally on census plots. These were marked and monitored to determine if the nest failed or fledged (Table 1.2). In all, nests of 10 species of ground- or shrub-nesting birds were found on the study area. Additionally, several nests were found of mid-story/canopy species on the census plots. These included, red-bellied woodpecker, blue jay, summer tanager, red-eyed vireo, blue-gray gnatcatcher, and eastern wood-pewee.

DISCUSSION

Leopold's Law of Interspersion (Leopold 1933) states that areas where simultaneous access is available to more than one environmental type will contain greater densities of game. In discussing edge-effects, Leopold (1933) noted the importance of edges to many different species, "An acre of fencerow or hedge, consisting, so to speak, entirely of edges, usually has more game (and songbirds also) than many acres of unbroken woods, or wheat, or corn." Gates and Gysel (1978) reported greater densities of nesting avian species within field-forest ecotones, with the overall species complex

Table 1.2. Nests found in open-woodland and old-field habitat, summer field season 1996 and 1997, Ames Plantation, Grand Junction, Tennessee.

Species	Number nests	Habitat	Number successful	Number parasitized	Number predated	Lost other causes
Blue grosbeak	3	field	0	1	2	1
Brown thrasher	1	woods	0	0	0	1
Common yellowthroat	2	field/woods	1	1	1	0
Eastern meadowlark	1	field	0	0	1	0
Eastern towhee	7	woods	1	3	6	0
Eastern wild turkey[1]	3	field/woods	-	-	-	-
Field sparrow	9	field/woods	1	1	7	1
Indigo bunting	10	field/woods	2	2	7	1
Northern bobwhite	3	field/woods	0	0	3	0
Red-winged blackbird	4	field	0	0	2	2
Yellow-breasted chat	1	woods	0	0	1	0
Totals	44		4	8	33	6

[1] Data on wild turkey nest success were not collected

dominated by species of mixed habitat. This positive edge-effect reaches maximal effectiveness when different habitats are interspersed in varying sizes and configurations throughout an area.

The Ames study area is centered around the National Championship field trial course. This course has been managed to provide maximum interspersion of habitats, particularly agricultural fields, old fields, and scattered woodlots, for the benefit of northern bobwhite. Therefore, species capable of inhabiting several habitat types including open woodlands, shrubland, old field, and field-forest edges would be expected to dominate the avian community of the field trial courses.

The most abundant species censused within the study area were indigo bunting, field sparrow, and red-winged blackbird. Habitat requirements for each of these species can be met in several habitat types. Red-winged blackbirds may be the most numerous North American landbird, nesting in great densities in wetland habitats, tallgrass prairie, and riparian habitats (Ehrlich et al. 1988, Herkert 1991). Similarly, indigo buntings and field sparrows inhabit various habitat types including forest edge and clearings, open woodlands, grassland, old fields, and shrubland habitats (Ehrlich et al. 1988). Eastern towhees were the second most abundant species on forested plots. Although eastern towhees were not found in old fields, they nest in a variety of other habitat types including shrubland, young forest, and forest edge (Askins 1993).

Several avian species characterized as early successional habitat specialists were detected on the study area (yellow-breasted chat, field sparrow, northern bobwhite) (Askins 1993). However, grassland specialists were rare despite the existence of nesting

records for several species, e.g. grasshopper sparrow, Bachman's sparrow, and dickcissel, within Fayette County, Tennessee (Nicholson 1997). Conditions exist on the study area that may be attractive to some species of grassland birds (i.e. weedy fields with mixed grasses for dickcissels and open pine woods with dense understory vegetation for Bachman's sparrow)(Ehrlich et al. 1988, Swanson 1996). However, the relatively small patches of grassland habitat on the study area may explain the absence of more area-sensitive species. The number of breeding birds found in grassland habitats is strongly related to patch size (Herkert 1991). Eastern meadowlarks, fairly common in some cattle pastures surrounding the study area, were rare within the study area, with only one eastern meadowlark territory established in 2 years on field study plots. Grasshopper sparrows have been reported as an area-sensitive grassland species, and are not usually found in grassland patches < 10 ha (Herkert 1991). Eastern meadowlarks are considered only moderately sensitive to patch size, but nesting usually does not occur on sites < 5 ha (Swanson 1996). On the field trial course, no contiguous grassland patch exceeded 5 ha, and may account for the rarity of grassland specialists.

Chapter 2: Effects of experimental reduction of rodent populations on success of artificial nests.

INTRODUCTION

Nest predation is thought to be the most important agent of mortality for many species of songbirds (Martin 1989). In fragmented landscapes, local songbird populations may be limited by a combination of nest predation and parasitism by brown-headed cowbirds (Brittingham and Temple 1983, Wilcove 1985, Robinson et al. 1995). The idea that nest predation has a negative correlation with the distance to habitat edge has been widely discussed and investigated (Paton 1994). Many studies have reported this relationship and the conclusion has been widely accepted that in a fragmented landscape, where the proportion of edge habitat is particularly high, nesting passerines will experience greater nest predation than in less fragmented landscapes (Robinson et al. 1995).

In a review of real and artificial nest studies, Paton (1994) concluded that most studies indicate a negative correlation between the distance of a nest to edge and the probability of predation. However, Paton (1994) noted that one particular area remaining unstudied is how the relative density of predators affects nest predation. Many researchers involved with both natural and artificial nest studies agree that data are still lacking on identification of predator assemblages and densities and their relationships with

nest success (Picman 1988, Martin 1989, Paton 1994, Fenske-Crawford and Niemi 1997) Studies of natural and artificial nests have revealed patterns of predation, but in most situations must depend on circumstantial evidence to speculate upon predator species identity and the relative importance of individual predator species (Gates and Gysel 1978, Gottfried and Thompson 1978, Yahner and Voytko 1989).

Leimgruber et al. (1994) suggested the use of automatic cameras would help solve this problem and indeed, cameras have provided valuable information in nest predator identification. However, various factors, including expense and an inordinate number of camera hours per predation event, severely limit most camera studies. Picman and Schriml (1994) conducted one of the only published camera studies with large sample sizes of predated nests and reported raccoons and skunks as the primary nest predators. However, Leimgruber et al. (1994) concluded that camera study techniques may be selectively biased towards large and medium-sized mammals. More timid species, such as small rodents, may be undersampled because of their fear of the camera flash or apparatus. As a result of these problems, information on relative importance of individual predator species and predator density have not been acquired and therefore impacts of individual predator species on nest success remains largely speculative.

Although most researchers using artificial nests recognize that artificial nest predation rates do not directly measure true predation rates on natural nests, they assume predation rates reflect the relative pattern of predation on real nests (Haskell 1995*a*). However, most artificial nest studies have typically used eggs of chicken, northern bobwhite, or Japanese quail to model passerine nests (Wilcove 1985, Yahner and Mahan

1996). These eggs are much larger than eggs of typical passerines and may fail to sample the entire predator assemblage by eliminating small-mouthed predators, thus failing to represent even relative patterns of predation (Reitsma et al. 1990, Roper 1992, Haskell 1995*a*,*b*).

Without accurate information about predator identity as well as relative importance of predator species, it is difficult to develop management strategies to provide adequate nesting habitats for breeding birds. To address these problems, I designed an artificial nest study that allowed sampling of the entire predator community by using two different sizes of eggs: society finch and northern bobwhite eggs. Specifically my objectives were: 1) determine the effect of egg size on artificial nest predation, 2) determine if rodents were significant nest predators, and 3) determine if rodent removal increased artificial nest success.

METHODS

Small Mammal Density

I trapped small mammals on all plots at the initiation and conclusion of all other field work in both 1996 and 1997 to attain a starting and ending index of population density. Small mammals were trapped using Sherman live-traps baited with peanut butter and rolled oats. A total of 16 traps was spaced systematically at alternate grid intersections on each plot, with a minimum distance of 60 m between traps. Traps were checked each morning for a 10-day period except for spring 1996 when the initial trapping period was conducted for 6-7 days. Each animal captured was identified to species and

released at the capture site. A density index of catch-per-unit effort was utilized to compare starting and ending small mammal densities (Caughley 1977).

Small Mammal Removal

In 1996, 2 study plots of each habitat type (4 plots total) were randomly selected as experimental plots. Two plots in each habitat type (4 plots total) also served as controls. Control and experimental groups were switched in 1997. Following live-trapping in 1996, a total of 22 Victor® rat traps were systematically spread throughout the experimental plots in an attempt to decrease rodent populations. Within the study area, traps were placed at each interior grid intersection, whereas traps were alternately spaced on the exterior grid points. In 1997, to further intensify the effect of small rodent removal, trapping effort was increased by adding 10 traps per plot and extending the trapping period by 9 days. Traps were placed on every grid intersection for a total of 32 traps per plot. Snap-traps were baited with peanut butter and checked daily throughout the artificial nest trials. Species was recorded for all animals caught and carcasses were collected and discarded away from the study plots to prevent the attraction of other predators. Pre-treatment and post-treatment density indices were compared between control and experimental plots to determine the effect of trapping.

Artificial Nest Trials

Artificial nests were placed throughout all study plots in 1996 and 1997, immediately following small mammal live-trapping, in conjunction with placement of snap-traps on treatment areas. Three artificial nest trials were conducted each year with 10 days separating trials. Artificial nest trials were conducted from 22 May - 23 July 1996

and 15 May - 9 July 1997. An early spring marked by warm temperatures and above-average rainfall prompted the earlier start in 1997.

Artificial nests consisted of a small, round, open-cup nest (11-cm diameter, depth ≤ 2.5 cm) constructed of woven grass, purchased from a hobby shop. Two different egg sizes were utilized in the experiment. One-half of the artificial nest sample contained eggs of northern bobwhite purchased from a commercial game farm. The other half of the artificial nest sample utilized an egg of a society finch obtained from a commercial aviculturist. Society finch eggs are much smaller than bobwhite eggs but are similar in size to eggs of many small passerines (Table 2.1). This approach allowed testing for differential predation of artificial nests related to egg size.

Each nest contained one real egg and one artificial egg. Real eggs were paired with artificial eggs to provide a nutritional reward to nest predators, thereby avoiding predator aversion to artificial nests. Artificial eggs were modeled after either a bobwhite or finch egg to allow identification of nest predators from dentition impressions (Møller 1989, Nour et al. 1993, Haskell 1995*a*). Artificial eggs were made of Plastalina®, a soft, malleable, whitish modeling clay. Artificial bobwhite eggs were made from plaster molds while finch eggs, because of their small size, were made by hand. All nests and eggs, both clay and real, were rinsed before use to remove human scent. Rubber gloves were worn during handling of nests and eggs, and rubber boots were worn during nest placement and nest monitoring to avoid transfer of human scent.

Nests were placed in each plot in pairs, 1 nest with bobwhite eggs and the other with finch, with 10 pairs placed per plot per trial. Nest pairs were placed systematically

Table 2.1. Egg sizes of 10 avian species found in old-field and open-woodland habitat in western Tennessee as compared to eggs of northern bobwhite, Japanese quail, and society finch.

Bird species	Egg size (mm)
Northern bobwhite[1]	30.0 x 24.7
Japanese quail	33.0 x 23.0
Society finch[2]	16.0 x 11.4
Blue grosbeak	22.0 x 16.8
Common yellowthroat	17.5 x 13.3
Dickcissel	20.8 x 15.7
Eastern meadowlark	27.7 x 20.3
Eastern towhee	23.1 x 17.0
Field sparrow	17.9 x 13.5
Grasshopper sparrow	18.6 x 14.4
Indigo bunting	18.7 x 13.7
Kentucky warbler	18.6 x 14.3
Yellow-breasted chat	21.9 x 16.9

[1]From a sample of 30 game farm eggs, measured summer 1996.
[2]From a sample of 30 aviculturist eggs, measured summer 1996.

Sources: Harrison, H.H. 1975. A field guide to the birds' nests east of the Mississippi River. Houghton Mifflin Company. New York. 257pp.

DeGraaf, R.M., and T.J. Maier. 1996. Effect of egg size on predation by white-footed mice. Wilson Bull. 108:535-539.

(Picman 1988), in alternate grid cells within an approximate 3-m radius of the cell center, with at least 3 m separating the two nests within a pair.

Artificial nests were placed on the ground within tufts of grass when available or alternatively at the base of a shrub or bush or other vegetation. Nesting birds in the area included both ground- and shrub-nesters. In placing only ground nests, I followed the recommendation of Martin (1987) who suggested nest height consistency within an artificial nest study. Restriction to ground nests also allowed direct comparisons between egg sizes. Nests were monitored every 3 days for a 12-day period, corresponding to the mean incubation period for most grassland/shrub passerines nesting on the study area (Reitsma et al. 1990). A 3-day monitoring period was selected because of the need to collect predated clay eggs.

An individual nest was considered predated when an egg was missing or when detectable damage had been done to either the real or artificial egg (Møller 1989, Nour et al. 1993). Detectable damage included a broken or cracked real egg and/or an artificial egg containing dentition impressions.

Identification of Predators

Predators of nests were identified primarily from dentition marks left on clay eggs in predated nests. Nest condition was also noted and used as a secondary identifier. Reference eggs were made by imprinting a set of clay eggs with dentition marks created using skulls of all potential mammalian predators on the study area. These reference eggs were then compared with eggs from predated nests and the nest predator identified.

DATA ANALYSIS

Density Index

I estimated small mammal density by calculating a catch-per-unit effort daily for each study plot by dividing the number of captures by the number of live-traps set (16).

Density Index = number of captures / total number of traps set (16)

All species of small mammals were pooled because we wished to determine the effect of the entire small mammal community rather than individual species. Only short-tailed shrews were censured from the data because they have not been reported as nest predators in the literature (George et al. 1986). The density index was recalculated with sprung traps censured from the calculation of the density index by subtracting the number of sprung traps on a plot per day from the total number of traps (16).

Density Index = number captures / 16 - number sprung traps

A closed trap which contained no animal was classified as a sprung trap.

Caughley (1977) suggested that if capture indices exceed 0.20, capture data may be better treated as a frequency rather than as an absolute density. Forested plots typically exceeded this limit whereas field plots did not. Therefore, data were also calculated as a frequency index. The frequency index was calculated using the equation:

$$\textbf{Frequency Index} = (1 - f) = e^{-x}$$

where f = the frequency of capture per trap and x = the estimated density of captures per trap. Results of the frequency index and absolute density index calculations were compared.

I used analysis of variance to test for effects of time period (beginning or ending trap period), treatment (rodent removal or control), year (1996 or 1997), and habitat (field or forest) on the density of small mammals (Rencher 1995). Calculations of least square means were utilized to aid in interpretation. All analyses were conducted using PROC MIXED (SAS 1997).

Nest Predation

The fate of each artificial nest was classified into a variable, nest fate, with 3 levels: successful (unpredated throughout the 12-day trial), predated by rodents, or predated by another source. The number of rodent predations was likely underestimated because the level containing predation by "other" sources includes the category of unknown predator, some of which were possibly rodent-caused. Setting nest fate as the dependent variable, a generalized logit model was fitted with explanatory variables egg size (finch or bobwhite), habitat (field or woods), trial (1, 2, 3), treatment (trapped or untrapped), and year (1996, 1997) (Agresti 1990). A likelihood ratio test was run to determine goodness of fit.

Individual variable and interaction significance was ascertained using P-values for chi-squares calculated by the logit model and presented in an analysis of variance table. A frequency table was utilized to aid in interpretation of the ANOVA table. All calculations of the logit model were conducted using PROC CATMOD and interpretation was aided by a frequency table created by PROC FREQ (SAS 1997).

RESULTS

Small Mammal Density

Changes in small mammal densities occurred between treatments (Table 2.2) and habitats (Table 2.3). Comparison of analyses using both the frequency index and density index revealed some slight changes in P-values but no difference in overall interpretation (Table 2.4). Therefore, following Caughley's (1977) suggestion, all comparisons of rodent densities in the interpretation of results have been made using probability values from the frequency index analysis. The test for main effects revealed several significant variables (Table 2.4). Results from calculation of differences in least square means are presented to aid interpretation (Table 2.5). Densities of small mammals differed between 1996 and 1997, with greater densities in 1997 ($P = 0.006$). The interaction between year and time was marginally significant because greater densities of small mammals existed in both trapping periods in 1997 versus 1996 ($P = 0.053$).

Forested plots had much greater small mammal densities than field plots ($P \leq 0.001$). A significant interaction existed between trapping period and habitat and their relationship with small mammal densities ($P \leq 0.001$). Densities in field habitat increased with time, as final densities exceeded starting densities ($P \leq 0.020$). Densities on forested plots decreased with time, as final densities decreased from starting levels ($P \leq 0.001$).

Treatment was a significant main effect in the density index analysis ($P = 0.001$). The P-value was greater in the frequency index analysis, but approached significance ($P = 0.097$). The interaction of treatment with time was significant in both analyses, revealing differential effects of treatment on small mammal density between time periods ($P \leq$

Table 2.2. Mean number of small mammals captured per day per trap calculated as an absolute density and frequency for initial and final trapping periods on control and experimental plots, Ames Plantation, Grand Junction, Tennessee, 1996-1997.

	1996		1997	
Treatment	Initial trap period Mean (SE)	Final trap period Mean (SE)	Initial trap period Mean (SE)	Final trap period Mean (SE)
DENSITY	*n=27*	*n=40*	*n=40*	*n=40*
Control	0.1738 (0.0314)	0.2065 (0.0209)	0.3359 (0.0359)	0.3200 (0.0269)
Experimental	0.2180 (0.0369)	0.1170 (0.0173)	0.2588 (0.0400)	0.1651 (0.0129)
FREQUENCY	*n=27*	*n=40*	*n=40*	*n=40*
Control	0.2115 (0.0409)	0.2462 (0.0284)	0.4951 (0.0707)	0.4185 (0.0419)
Experimental	0.2819 (0.0559)	0.1325 (0.0207)	0.3786 (0.0695)	0.1851 (0.0154)

Table 2.3. Mean number of small mammals captured per day per trap calculated as an absolute density and frequency for initial and final trapping periods on field and forest plots, Ames Plantation, Grand Junction, Tennessee, 1996-1997.

	1996		1997	
Habitat	Initial trap period Mean (SE)	Final trap period Mean (SE)	Initial trap period Mean (SE)	Final trap period Mean (SE)
DENSITY	*n=26 (F), n=28 (W)*	*n=40*	*n=40*	*n=40*
Field	0.0622 (0.0105)	0.0860 (0.0136)	0.0946 (0.0115)	0.1784 (0.0151)
Forest	0.3201 (0.0305)	0.2375 (0.0189)	0.5001 (0.0313)	0.3068 (0.0275)
FREQUENCY	*n=26 (F), n=28 (W)*	*n=40*	*n=40*	*n=40*
Field	0.0658 (0.0116)	0.0945 (0.0156)	0.1026 (0.0128)	0.2034 (0.0190)
Forest	0.4147 (0.0475)	0.2841 (0.0264)	0.7711 (0.0641)	0.4002 (0.0428)

Table 2.4. Capture frequency index and density index results from analysis of variance for small mammal trapping data, Ames Plantation, Grand Junction, Tennessee, 1996-1997.

Explanatory variables	Numerator degrees of freedom	Denominator degrees of freedom	F-value	Pr > F
DENSITY INDEX				
year	1	273	9.91	**0.0018**
time	1	273	9.61	**0.0021**
time*year	1	273	1.01	0.3152
habitat	1	11	65.03	**0.0001**
habitat*time	1	273	58.29	**0.0001**
treatment	1	11	5.75	**0.0354**
habitat*treatment*time	2	273	5.38	**0.0051**
treatment*time	1	273	14.56	**0.0002**
FREQUENCY INDEX				
year	1	273	7.75	**0.0058**
time	1	273	16.88	**0.0001**
time*year	1	273	3.76	**0.0535**
habitat	1	11	41.62	**0.0001**
habitat*time	1	273	56.97	**0.0001**
treatment	1	11	3.29	0.0972
habitat*treatment*time	2	273	4.22	**0.0157**
treatment*time	1	273	10.58	**0.0013**

Table 2.5. Differences between least square means for trap frequencies across year, treatment, habitat, and trapping period combinations, Ames Plantation, Grand Junction, Tennessee 1996-1997.

Effect	Habitat (hab)	Treatment (trmt)	Trap period (time)	Year	Habitat (hab)	Treatment (trmt)	Trap period (time)	Year	Difference	Std. Error	DF	t	Pr > |t|
year				1996				1997	0.1558	0.0560	273	2.78	**0.006**
time			initial				final		-0.0917	0.0223	273	-4.11	**0.001**
time*year			initial	1996			initial	1997	0.1991	0.0611	273	3.26	**0.001**
time*year			initial	1996			final	1996	-0.0484	0.0332	273	-1.46	0.146
time*year			initial	1996			final	1997	0.0640	0.0611	273	1.05	0.296
time*year			initial	1997			final	1996	-0.2475	0.0594	273	-4.17	**0.001**
time*year			initial	1997			final	1997	-0.1350	0.0298	273	-4.53	**0.001**
time*year			final	1996			final	1997	0.1125	0.0594	273	1.89	0.059
treatment		control				experimental			-0.1013	0.0559	11	-1.81	0.097
habitat	field				forest				0.3607	0.0559	11	6.45	**0.001**
trmt*time		control	initial			control	final		-0.0196	0.0315	273	-0.62	0.535
trmt*time		control	initial			experimental	initial		-0.0292	0.0610	273	-0.48	0.632
trmt*time		control	initial			experimental	final		-0.1931	0.0602	273	-3.21	**0.001**
trmt*time		control	final			experimental	initial		-0.0096	0.0602	273	-0.16	0.873
trmt*time		control	final			experimental	final		-0.1735	0.0594	273	-2.92	**0.004**
trmt*time		experimental	initial			experimental	final		-0.1639	0.0315	273	-5.21	**0.001**
hab*time	field		initial		field		final		0.0757	0.0317	273	2.39	**0.017**
hab*time	field		initial		forest		initial		0.5281	0.0609	273	8.67	**0.001**
hab*time	field		initial		forest		final		0.2689	0.0602	273	4.46	**0.001**
hab*time	field		final		forest		initial		0.4524	0.0601	273	7.53	**0.001**
hab*time	field		final		forest		final		0.1932	0.0594	273	3.26	**0.001**
hab*time	forest		initial		forest		final		-0.2592	0.0313	273	-8.28	**0.001**
hab*trmt*time	field	control	initial		field	control	final		0.0900	0.0447	273	2.02	**0.045**
hab*trmt*time	field	control	initial		field	experimental	initial		-0.0210	0.0864	273	-0.24	0.808
hab*trmt*time	field	control	initial		field	experimental	final		0.0404	0.0852	273	0.47	0.636
hab*trmt*time	field	control	initial		forest	control	initial		0.5364	0.0862	273	6.22	**0.001**
hab*trmt*time	field	control	initial		forest	control	final		0.4072	0.0852	273	4.78	**0.001**
hab*trmt*time	field	control	initial		forest	experimental	initial		0.4989	0.0862	273	5.79	**0.001**
hab*trmt*time	field	control	initial		forest	experimental	final		0.1097	0.0852	273	1.29	0.199

Table 2.5. (Continued).

Effect	Habitat (hab)	Treatment (trmt)	Trap period (time)	Year	Habitat (hab)	Treatment (trmt)	Trap period (time)	Year	Difference	Std. Error	DF	t	Pr > |t|
hab*trmt*time	field	control	final		field	experimental	initial		-0.1110	0.0852	273	-1.30	0.194
hab*trmt*time	field	control	final		field	experimental	final		-0.0496	0.0839	273	-0.59	0.555
hab*trmt*time	field	control	final		forest	control	initial		0.4463	0.0850	273	5.25	**0.001**
hab*trmt*time	field	control	final		forest	control	final		0.3171	0.0839	273	3.78	**0.001**
hab*trmt*time	field	control	final		forest	experimental	initial		0.4089	0.0850	273	4.81	**0.001**
hab*trmt*time	field	control	final		forest	experimental	final		0.0197	0.0839	273	0.23	0.815
hab*trmt*time	field	experimental	initial		field	experimental	final		0.0614	0.0447	273	1.37	0.170
hab*trmt*time	field	experimental	initial		forest	control	initial		0.5573	0.0862	273	6.47	**0.001**
hab*trmt*time	field	experimental	initial		forest	control	final		0.4281	0.0852	273	5.02	**0.001**
hab*trmt*time	field	experimental	initial		forest	experimental	initial		0.5200	0.0862	273	6.03	**0.001**
hab*trmt*time	field	experimental	initial		forest	experimental	final		0.1307	0.0852	273	1.53	0.126
hab*trmt*time	field	experimental	final		forest	control	initial		0.4959	0.0850	273	5.84	**0.001**
hab*trmt*time	field	experimental	final		forest	control	final		0.3667	0.0839	273	4.37	**0.001**
hab*trmt*time	field	experimental	final		forest	experimental	initial		0.4585	0.0850	273	5.40	**0.001**
hab*trmt*time	field	experimental	final		forest	experimental	final		0.0693	0.0839	273	0.83	0.410
hab*trmt*time	forest	control	initial		forest	control	final		-0.1292	0.0442	273	-2.92	**0.004**
hab*trmt*time	forest	control	initial		forest	experimental	initial		-0.0374	0.0859	273	-0.44	0.664
hab*trmt*time	forest	control	initial		forest	experimental	final		-0.4266	0.0850	273	-5.02	**0.001**
hab*trmt*time	forest	control	final		forest	experimental	initial		0.0918	0.0850	273	1.08	0.281
hab*trmt*time	forest	control	final		forest	experimental	final		-0.2974	0.0839	273	-3.54	**0.001**
hab*trmt*time	forest	experimental	initial		forest	experimental	final		-0.3892	0.0442	273	-8.80	**0.001**

0.001). There was no difference in initial densities between control and experimental plots ($P = 0.632$), however final densities did differ, with control plot densities greater than treatment plot densities ($P = 0.004$). Starting and ending densities on control plots did not differ ($P = 0.535$). Starting and ending densities on experimental plots differed, with ending densities around one half of starting levels ($P = 0.001$).

A significant interaction between treatment, time, and habitat demonstrated a change in the effect of treatment over time between field and forest plots ($P = 0.016$). There were no differences between starting densities on control and experimental plots in the fields ($P = 0.808$). Ending densities did not differ between control field plots and experimental field plots ($P = 0.555$), resulting from apparent strong seasonal increases in small mammal populations in the field. There was a significant difference between ending densities of control and experimental forested plots, with experimental plot densities being much lower ($P < 0.001$). This was a change from initial densities between control and experimental forest plots, which showed no significant differences ($P = 0.664$). Densities in control plots in field habitats increased from initial to ending trap period ($P = 0.045$). Densities of control forested plots decreased from initial to concluding trap periods ($P = 0.004$). Densities on experimental field plots increased slightly as well, but this increase was not significant ($P = 0.171$). Densities on experimental forest plots did decrease from initial to final trap periods ($P < 0.001$).

Rodent Removal

A total of 269 small mammals representing 7 species was removed from the 4 experimental plots in 1996. An intensified trapping effort resulted in more captures in

1997 with 518 individuals representing 9 species removed from 4 experimental plots (Table 2.6). White-footed mice and hispid cotton rats were the most commonly caught animals. Most white-footed mice were captured in the forest or field-forest ecotone whereas cotton rats dominated captures in the field.

Nest Predation

I was unable to locate 8 nests at the end of all trials for both years out of a sample of 960 nests. These nests were censured from the data. Therefore the artificial nest sample contained 952 nests, of which 65.4 % were predated. Rodents predated 42.4% of available nests, representing 64.8% of all predation. Rodent predators may have consisted of any rodent species captured on the area, but were likely dominated by 2 species, cotton rats and white-footed mice, whose abundance appeared much greater than any other species (Table 2.6). Rodent predation could be split into 2 categories, medium-sized rodents and small rodents. Only 2 species of medium-sized rodents were captured on the study area, cotton rats and marsh rice rats. Cotton rats were abundant or common in all fields and were second in total captures in all habitats. Marsh rice rats were much more infrequent and usually restricted to moist areas of fields or hedgerows. Therefore, predation by medium-sized rodents can likely be attributed to cotton rats. Several species of small rodents were captured on the study area. Of these, the white-footed mouse was the most frequently captured. Assuming equal preference among the small rodent species for avian eggs, white-footed mice were likely responsible for the majority of predation by small rodents. Small and medium-sized rodents have been clumped together in a single category entitled rodents throughout this analysis, however they are referred to separately

Table 2.6. Number of small mammals captured by species in forested and field habitats, summer 1996 and 1997, Ames Plantation, Grand Junction, Tennessee.

	Field		Forest		
Species	1996	1997	1996	1997	Total
House mouse	0	21	0	0	21
Hispid cotton rat	23	129	0	18	170
White-footed mouse	38	18	147	208	411
Golden mouse	1	0	7	3	11
Short-tailed shrew	14	64	24	21	123
Marsh rice rat	4	20	0	5	29
Woodland vole	2	4	6	5	17
Fulvous harvest mouse	3	2	0	0	5
Southern flying squirrel	0	0	0	7	7
Totals	85	258	184	260	794

on occasion for descriptive purposes.

Egg size was originally included in the full model and differences existed in predation between the 2 sizes of eggs ($P \leq 0.001$) (Table 2.7). However, no biologically meaningful, significant interactions with egg size were detected. Therefore, to facilitate interpretation, data were reanalyzed in a reduced model by egg size. All 4- and most 3-way interactions were removed due to lack of significance (Table 2.8).

Finch Eggs

Of 475 nests containing finch eggs, 73.1% were predated. Rodents predated 45.2% of available finch egg nests, representing 61.6% of all predations. Real finch eggs were typically consumed in rodent predations of artificial nests. Small rodents predated 168 nests with finch eggs. In 118 of the 168 nests (70.2%) predated by small rodents, the finch egg was consumed and shell fragments were left in or around the nest. Another 21 predations (12.5%) by small rodents resulted in the total disappearance of the real finch egg without any trace remaining. In 29 (12.5%) instances, small rodents disturbed nests (i.e. bit into the clay egg) but the real finch egg remained unharmed.

Artificial nests with finch eggs predated by cotton rats usually resulted in the consumption of the real finch egg. Cotton rats predated 47 nests and left egg fragments in the nest in 23 cases (48.9%). Another 19 nests (40.4%) predated by cotton rats resulted in the total disappearance of the real finch egg. In 5 (10.6%) finch-egg nests, cotton rats disturbed the nest (i.e. bit into clay egg) but did not consume the real finch egg.

Predation of finch eggs differed between 1996 and 1997 ($P = 0.022$). Predation from sources other than rodents decreased in 1997 (33.3% in 1996 versus 22.6% in 1997)

Table 2.7. Maximum-likelihood analysis of variance table for full model with nest fate as the response variable, Ames Plantation, Grand Junction, Tennessee.

Explanatory variable	Degrees of freedom	Chi-square	P-value
intercept	2	0.01	0.9974
size	2	11.87	**0.0026**
year	2	0.00	0.9995
size*year	2	2.24	0.3255
treatment	2	1.72	0.4227
size*treatment	2	0.50	0.7802
year*treatment	2	3.58	0.1670
size*year*treatment	2	2.10	0.3493
habitat	2	0.00	0.9988
size*habitat	2	1.32	0.5164
year*habitat	2	0.00	0.9994
size*year*habitat	2	0.49	0.7835
treatment*habitat	2	0.04	0.9780
size*treatment*habitat	2	1.75	0.4172
year*treatment*habitat	2	6.32	**0.0425**
size*year*treatment*habitat	2	0.88	0.6427
trial	4	2.90	0.5746
size*trial	4	4.98	0.2895
year*trial	4	0.04	0.9998
size*year*trial	4	0.81	0.9372
treatment*trial	4	2.41	0.6607
size*treatment*trial	4	3.31	0.5068
year*treatment*trial	4	2.66	0.6154
size*year*treatment*trial	4	4.73	0.3159
habitat*trial	4	5.43	0.2461
size*habitat*trial	4	1.99	0.7380
year*habitat*trial	4	1.49	0.8292
size*year*habitat*trial	4	1.19	0.8804
treatment*habitat*trial	4	2.09	0.7198
size*treatment*habitat*trial	4	6.38	0.1726
year*treatment*habitat*trial	4	3.63	0.4578
likelihood ratio	4	1.01	0.9090

Table 2.8. Maximum-likelihood analysis of variance table for reduced model analyzed by egg size with nest fate as the response variable for artificial nests, Ames Plantation, Grand Junction, Tennessee.

Explanatory variable	Degrees of freedom	Chi-square	P-value
FINCH EGGS			
intercept	2	30.83	0.0001
year	2	7.65	**0.0219**
treatment	2	12.04	**0.0024**
year*treatment	2	1.54	0.4619
trial	4	6.30	0.1781
habitat	2	6.53	**0.0382**
trial	4	6.30	0.1781
year*trial	4	4.30	0.3674
treatment*trial	4	2.62	0.6225
year*treatment*trial	4	10.49	**0.0329**
habitat*trial	4	30.42	**0.0001**
likelihood ratio	18	20.33	0.3147
BOBWHITE EGGS			
intercept	2	0.00	0.9985
year	2	6.40	**0.0407**
treatment	2	27.95	**0.0001**
year*treatment	2	1.50	0.4722
trial	4	9.94	**0.0415**
year*trial	4	0.45	0.9785
treatment*trial	4	1.98	0.7391
year*treatment*trial	4	4.81	0.3072
habitat	2	1.97	0.3738
year*habitat	2	2.28	0.3206
treatment*habitat	2	1.36	0.5055
year*treatment*habitat	2	1.65	0.4376
trial*habitat	4	27.48	**0.0001**
year*trial*habitat	4	2.01	0.7344
treatment*trial*habitat	4	3.24	0.1839
likelihood ratio	4	3.24	0.5190

while rodent predation increased (43.0% in 1996 versus 47.3% in 1997). This result corresponded with greater rodent densities recorded in 1997. The number of successful nests (nests where neither clay nor real eggs were damaged) also increased in 1997 (23.6% in 1996 versus 30.1% in 1997) as a result of the decrease in predation by other predators in 1997.

Removal of small mammals had a significant effect upon nest fate ($P = 0.002$). Rodent predation decreased substantially from control to experimental areas (53.1% vs. 37.8%, respectively). The number of successful nests increased on experimental plots as compared with control plots (33.2% vs. 20.1%, respectively).

Habitat type of the study plot also was related to the likelihood of predation of nests with finch eggs ($P = 0.038$). Predation by other predators was similar in field and forest habitats (28.4% vs. 27.4%, respectively). Rodent predation was initially greater in forested plots than in fields (68.7% vs. 30.4%, respectively) and the overall number of successful nests was also lower on forested plots than in fields (22.4% vs. 30.9%, respectively). The effect of habitat changed with trial ($P \leq 0.001$). Rodent predation was initially much higher in forested habitats then decreased steadily with time through the trials (68.7%, 43.2%, and 38.7% for Trials 1, 2, and 3, respectively). Rodent predation in field habitat started at a much lower level, but steadily increased with time through the trials (30.4%, 38.0%, and 53.8% for Trials 1, 2, and 3, respectively). Decreasing trends in rodent predation in the woods coupled with increasing trends in the fields resulted in a reversal of predation levels with greater rodent predation in fields than in woods after Trial

1. Paralleling these trends, the number of successful nests decreased steadily with time in the fields, while increasing in forested habitats.

A significant interaction was also found between year, treatment, and trial (P = 0.033). No biological significance could be determined for this statistically significant relationship.

Bobwhite Eggs

Of 476 nests containing bobwhite eggs, 57.8% were predated. Rodents predated 39.7% of available bobwhite egg nests, representing 68.7% of all predation. Rodents rarely damaged the real bobwhite egg, though frequently damaged the artificial egg. Small rodents predated 123 nests with bobwhite eggs, leaving the real egg unharmed 87% of the time. In 8.9% of small rodent predations, the real bobwhite egg disappeared, and therefore may not have been predated by a small rodent. Only 5 small rodent predations (4.1%) were accompanied by a damaged egg that appeared to be the work of a small rodent.

Real bobwhite eggs were also rarely damaged in cotton rat predations. Of 66 cotton rat predations, only 10 (15.2%) were accompanied by a damaged egg that appeared to be attributable to a cotton rat. In 53 cases (80.3%) of cotton rat predation of artificial bobwhite nests, cotton rats did not damage the real egg. In an additional 3 cases (4.5%) of cotton rat nest predation, the bobwhite egg disappeared and may not have been predated by this rodent.

Predation on bobwhite eggs differed between 1996 and 1997 (P = 0.041). Predation from sources other than rodents decreased slightly in 1997 (20.2% vs. 16.0%,

respectively) while rodent predation increased (34.0% vs. 45.4%, respectively). The number of successful bobwhite egg nests decreased slightly from 1996-1997 (45.8% vs. 38.7%, respectively).

Differences existed in response of nest fate to the explanatory variable treatment ($P \leq 0.001$). Predation by non-rodents decreased slightly from control to experimental plots (20.7% vs. 16.0%, respectively). However, rodent predation decreased sharply between untrapped and trapped plots (49.6% vs. 29.8%, respectively). Paralleling this trend, the number of successful nests was much greater on trapped plots than on untrapped plots (54.2% vs. 30.3%, respectively).

Differences were detected among nest fates across the 3 trials ($P = 0.041$). Predation by non-rodents increased slightly between Trials 1 and 2 and decreased in Trial 3 (20.1%, 24.8%, and 9.4% for Trials 1, 2, and 3, respectively). Predation by rodents initially decreased then stabilized or increased slightly (45.3%, 35.7%, and 38.1% for Trials 1, 2, and 3, respectively). However, the percent of successful nests increased throughout the trials, becoming greatest in the third trial (34.6%, 38.7%, and 52.5% for Trials 1, 2, and 3, respectively).

Nest fate did not differ by habitat type for bobwhite-egg nests ($P = 0.374$). However, there was a significant interaction between habitat and trial ($P \leq 0.001$). Rodent predation was initially much greater in forested habitats then decreased steadily with time through the trials (57.5%, 32.9%, and 28.7% for Trials 1, 2, and 3, respectively). Although rodent predation in field habitat started at a much lower level, it steadily increased with time through the trials (32.5%, 38.5%, and 47.5% for Trials 1, 2, and 3,

respectively). Decreasing trends in rodent predation in the woods coupled with increasing trends in the fields resulted in a reversal of predation levels, with greater rodent predation pressure in fields than in woods after Trial 1. Paralleling these trends, the number of successful nests decreased steadily with time in the fields, while increasing in the woods.

DISCUSSION

Artificial nests are commonly used in wildlife research to assess trends in avian nest predation. Artificial nests best mimic conditions found at a natural nest during the egg laying and incubation periods, bearing little resemblance to natural nests during the nestling and fledgling periods. Debate continues over whether results obtained through the use of artificial nests are good representations of predation on natural nests. Many researchers have found artificial nest predation to be similar to predation on natural nests within the same or similar habitats (Gottfried and Thompson 1978, Major 1990, Reitsma et al. 1990, Yahner and Delong 1992) whereas others report little or no correlation (Roper 1992, Guyn and Clark 1997).

The overall predation rate for artificial nests in this study was 65.4%, with a corresponding nest success of 34.6%. During the 2 years of the study, I collected information on 44 natural nests found within my study plots or similar habitats on the study area (Table 1.2). These nests experienced a success rate of only 12.2 %. Though the sample size was small, the results correspond with some of the lowest nest success percentages reported. In his review of 32 open-nesting passerines, Martin (1989) reported only 2 species (indigo bunting and Bell's vireo) with a lower nest success rate than 12.2%.

Best (1978) recorded a 10.2% success rate on field sparrow nests in similar habitats in Illinois and concluded recruitment was far below the level required to sustain the population. The percent of artificial nests that were not predated in my study (34.6%) is much higher than the success rate I recorded for natural nests on my study area. Therefore, it is likely that the artificial nest results underestimated actual predation rates, rather than over-estimated them.

"Rodents are significant nest predators"

The differences in predation rate between egg sizes clearly demonstrated a preference among predators for the smaller finch eggs. More finch-egg nests were predated than bobwhite-egg nests. This appears to be the result of a predator assemblage dominated by small-mouthed predators. Small rodents predated finch-egg nests more often than bobwhite egg nests (35.3% vs. 25.8%). Based on our capture index, white-footed mice were much more common than any other rodent species. White-footed mice, therefore, likely accounted for the majority of predation caused by small rodents. DeGraaf and Maier (1996) found that captive white-footed mice would not eat Japanese quail eggs but readily ate much smaller zebra finch eggs. In our study, small rodents were the most abundant and most common predators of nests. However, although rodents frequently predated artificial nests of both egg types, real finch eggs were usually consumed during nest predation by rodents whereas real bobwhite eggs almost never were. This is consistent with the observation that small-mouthed predators may be unable to eat large eggs (Reitsma et al. 1990, Roper 1992, Haskell 1995*a*,*b*, Ettel et al. 1998). Studies using automatic cameras have noted the frequent appearance of small rodents at artificial nests,

typically *Peromyscous* spp. or *Microtus* spp. (Leimgruber et al. 1994, Vander Haegen and DeGraaf 1996, Fenske-Crawford and Niemi 1997). These studies used quail eggs, however, and actual predation by small rodents was rarely observed. Scratch marks were frequently reported on the quail eggs, likely indicating failed attempts by rodents to consume eggs. My results with bobwhite eggs displayed a similar pattern. Small rodents predated 123 bobwhite-egg nests, often damaging the clay egg but rarely harming the real egg. In 107 (87%) of these small rodent predations, the real bobwhite egg was undamaged. On several occasions, scratch marks, similar to those reported in other studies, were noticed on bobwhite eggs presumably indicating failed attempts by small rodents to open bobwhite eggs. Only 5 (4.1%) small rodent predations of bobwhite-egg nests resulted in damage to the real egg.

Cotton rats were the second most common predator of artificial nests. Results from nests predated by cotton rats followed a pattern similar to nests predated by smaller rodents, with real finch eggs rarely left undamaged (10.6%) and real bobwhite eggs often not damaged (87.0%). Several texts have mentioned cotton rats and other rodents as predators of bobwhite nests, based on animal sign found at predated nests (Stoddard 1931, Simpson 1976). However, because rodents, including cotton rats and smaller rodents, rarely succeeded in damaging bobwhite eggs in this study, their ability to effectively consume bobwhite eggs is questionable. Rodents frequently visited artificial nest sites with bobwhite eggs, often severely damaging the nest structure and clay eggs and removing eggs from the nest bowl without damaging them. Therefore rodent

predation of bobwhite nests may frequently involve nest disturbance that may provoke the hen to abandon the nest, rather than actual consumption of the bobwhite egg.

Although mice have been observed attempting to predate quail eggs, no clear evidence demonstrating the frequency of rodent predation of passerine nests exists. Guillory (1987) reported predation of prothonotary warbler nestlings and adults by *Peromyscus* spp. Using artificial eggs, Rogers et al. (1997) reported results very similar to mine. Dentition marks found on artificial eggs placed in song sparrow nests suggested that mice (*Peromyscus* spp.) and shrews were the leading causes of egg loss in song sparrow nests. However, Rogers et al. (1997) were skeptical of these results, doubting that mice were common nest predators of song sparrow nests. I recorded actual and frequent predation of artificial nests by small rodents. Because of their ubiquitous habits, and numbers far exceeding any other local predator, *Peromyscus* and other species of small rodents, are potentially significant agents of nest loss for small passerines.

Rodents made up a very large component of the predator community, accounting for more predation for both sizes of eggs than all other sources combined. Rodent populations were greater in 1997, and subsequently predation by rodents increased in 1997 as compared to 1996. Rodents, thus, appeared to drive the overall pattern of predation on artificial nests. When rodent populations were reduced, the number of successful nests increased and the percentage of predated nests that rodents were accountable for decreased.

"Predation differs between old-field and open-woodland habitat."

Variations in predator access and movement between different habitats may affect

the patterns of nest predation on songbirds (Seitz and Zegers 1993). In my study, wooded habitats initially had greater rodent populations and subsequently experienced greater predation of artificial nests by rodents. Rodent community composition also differed in the forest where white-footed mice predominated, as compared to the field where cotton rats predominated. Several studies have found predation to increase near and within woody and riparian edges (Burger et al. 1994, Vander Haegen and DeGraaf 1996). The greater initial rodent populations in wooded areas in this study resulted in a better opportunity for the expression of treatment effects (rodent removal) in wooded habitats. Initially, the high rodent populations found in the woods resulted in large numbers of rodents trapped in that habitat. Therefore, after starting at very high levels, rodent predation on forested plots decreased with time.

Rodent populations in the fields were initially low and increased with time. Likewise, rodent predation in fields increased with time. Sections of all field plots had been partially mowed or burned prior to the start of trapping in both 1996 and 1997. Cotton rats were the most abundant rodent of field habitat. Goertz (1964) reported that cotton rats preferred and were typically found in areas of tall, dense vegetative growth, but would colonize sparsely vegetated areas once the vegetation had recovered. Portions of my field study plots were burned and bushhogged in both years. Cotton rats and other predators may have remained in the brushy forest-field edges or within the forest early in the season where varying levels of overhead vegetative structure afforded security from avian and mammalian predators. Fewer cotton rats were caught in the early trap period

than the final trap period in both years and may reflect these immigration patterns into the fields.

Far fewer cotton rats were caught in 1996 than in 1997 (23 vs. 147). I compared cotton rat captures between 1996 and 1997 by calculating a density index exclusively for cotton rats. Cotton rat mean daily densities (number of captures/total # traps) were 0.018 and 0.046 during the initial and final trapping periods, respectively, in 1997 as compared to mean daily densities of 0.003 and 0.029 in 1996. Goertz (1964) reported that cotton rat populations frequently cycle between years. Despite continued trapping on experimental field plots, no significant decrease of rodent populations was detected on field plots in either year. In 1997, rodent populations actually increased in trapped fields despite very high removals of cotton rats (129 individuals). Recruitment from reproduction or immigration of cotton rats into the field plots may have helped offset the effects of removal, particularly in 1997. Cotton rats dominated the rodent community in field habitats, but wooded habitats rarely contained cotton rats. I believe that the high levels of cotton rats in the fields and the inability to reduce those populations with trapping largely accounted for the observed differences in rodent predation rates between woods and field. Rodent predation increased in fields over time as the number of cotton rats in the fields increased, while rodent predation in the woods steadily decreased.

"Can nest success be increased by reducing the density of a nest predator?"

Reducing populations of predators to increase populations of a desired prey species is a concept that has received much attention in wildlife management (Stoddard 1931, Leopold 1933). Several studies indicate that removal of a primary predator will

result in increases in desired prey species. In a 20-year study of predator-prey populations in Alaska, wolf control resulted in corresponding increases in moose and caribou populations (Boertje et al. 1996). Predator reduction has long been a practice on preserves managed for upland game birds under the supposition that the reduction of predators of both nests and adults was related linearly to the total number of birds available to be hunted in the fall. Stoddard (1931) wrote that to properly manage for bobwhite, "Among the measures that must be adopted is the rational control of natural enemies." Increases of up to 75% in August gray partridge populations in Great Britain have been reported following predator control (Tapper et al. 1996). These studies focused mostly on removal of medium-sized mammals with large home ranges that are known to prey upon nests and sometimes adults as well.

Reitsma et al. (1990) believed smaller mammals, specifically red squirrels and chipmunks, to be the major causes of passerine nest loss on their study area. They developed an artificial nest study and trapped these rodents from the study site, to determine if nest success increased. No change in nest success was determined. However, a low number (23 and 26 from 2 plots) of individuals was removed. I removed 671 small mammals (excluding short-tailed shrews) during 1996 and 1997 on my experimental plots and determined that this treatment affected success of artificial nests. Fewer nests were predated by rodents on trapped plots than in untrapped plots.

Predator removal may be deemed impractical because of the great amount of labor involved and also the complicating factor of compensatory predation. The concept of compensatory predation assumes that although the population of one predator may be

reduced, predation may be temporarily decreased by that species but the net number of nests predated will remain constant. Other predators will compensate for the removal of the first predator and take more nests (Leimgruber et al. 1994). Reitsma et al. (1990) postulated that the combined effect of several other predators, e.g. blue jays, raccoons, and deer mice, may have resulted in sufficient nest predation to compensate for removal of red squirrels and chipmunks. Hensley and Fisher (1975) removed grey foxes in an area of upland poultry farms and found the result to be an irruption of weasels. If compensatory agents were at work in this study, reductions in rodents would have lead to reductions in rodent predation while the number of successful nests remained approximately stable. However, decreases in rodent predation coincided with increases in overall nest success on trapped plots. I witnessed no compensatory predation by other predators and conclude that compensatory predation is not always an automatic response to predator removal.

Chapter 3: Effects of habitat parameters and individual predator species on the survival of artificial nests.

INTRODUCTION

Birds may counter the risk of nest predation through careful selection of breeding territories and nest sites (Martin 1987). Females may select nest sites based on specific vegetative characteristics that ensure greater safety of the nest from predators (Ricklefs 1990). This has been demonstrated by studies of several polygamous species including red-winged blackbirds (Lenington 1980) and lark buntings (Pleszczynska 1978) where nest success was associated with the density of vegetative cover surrounding the nest.

Several artificial nest studies have found correlation between nest predation and various vegetative parameters at the nest site. Leimgruber et al. (1994) found artificial nest predation to be lower in dense ground cover. Jobin and Picman (1997) found increases in vegetation density and overhead concealment to be associated with greater artificial nest success in some marshes. Although artificial nest results may not be directly representative of predation on natural nests, studies on natural avian nests have indicated similar basic relationships between nest cover and nest success (Martin 1989).

One important element of any study of nest predation, whether based on natural or artificial nests, is the identification of predators. Identification of individual predator

species is largely absent in nest predation studies (Picman 1988, Martin 1989, Paton 1994, Fenske-Crawford and Niemi 1997). The breeding grounds of most avian species are shared with many species of potential nest predators. Different predators possess different foraging behaviors and search images and therefore nest site vegetative characteristics may affect predation risk in different ways. Density of nest cover, for example, may influence nest success where visual predators predominate, but may have less effect upon predators that rely predominately on olfactory cues. To better understand the relationships between habitat parameters and nest predation, primary predators must be identified. Although identification of predators has been attempted with several techniques, accurate identification of predators has seldom occurred and continues to be an important research need.

Objectives of this part of the study were to: 1) identify nest site habitat parameters associated with increased predation of artificial nests, 2) determine if negative distance to edge effects exist, and 3) identify the suite of predators predating artificial nests and document their relative importance.

METHODS

I placed 960 artificial nests in 2 different grassland/shrub habitats on Ames Plantation in western Tennessee from May-July, 1996-97. Nest site characteristics including vegetation measurements were recorded for each nest in an attempt to relate predation on artificial nests with habitat parameters. Clay eggs were used in artificial nests along with real eggs to allow identification of predators through dentition imprints

left in the clay eggs and to allow a comparison of the relative importance of predator species. For a complete description of artificial nests and experimental protocol see Methods, Chapter 2.

To maintain the field trial grounds and control encroachment of woody plants, burning is conducted on the study area on an approximate 3-year rotation. Burning is rarely uniform, but often patchy, as fires pass through some stretches while missing or incompletely burning others. Therefore, each nest site was designated into a management type, burned or unburned, based on whether the nest site had been burned within the current year.

The height and species composition of vegetation immediately surrounding each nest site was measured. The vegetative composition consisted of the vegetative cover into which the nest was originally placed. Vegetation was classified into 4 types: 1) forb (all non-woody plants except grasses), 2) grass, 3) litter (dead vegetative matter), or 4) woody vegetation (any plants containing a woody stem including tree seedlings, vines, and shrubs). The vegetative composition was recorded by ranking the vegetation type immediately surrounding each nest into 3 classes: 1) primary vegetation (the dominant vegetation type surrounding the nest), 2) secondary vegetation (the second-most abundant vegetation type), and 3) tertiary vegetation (any other prominent vegetation).

Two different types of habitat edge were recognized. The distance from edge habitats to each artificial nest was visually estimated using flagged gridlines as reference points. Primary edge was considered to be the nearest woody cover (woodlot or hedgerow) or (as in the case for open woodland) ecotone. Secondary edge represented

any other recognizable edge and was only considered if the secondary edge was closer to the nest than the primary edge. Secondary edge included anthropogenic disturbances to the habitat such as wildlife food plots, mowed/bushhogged strips, and seldom-used dirt roads.

I developed a nest detectability index to determine how vegetative density and nest visibility affected the success of artificial nests. A visual estimate was made from a point 1-m distant from the nest from each cardinal direction by recording the percentage of the nest visible at ground level from each point. The measurement was taken at the 1-m distance at ground level to represent the perspective of a terrestrial predator encountering the nest. Visual estimates taken at each nest were grouped into 1 of 4 detectability classes as follows: 1) 0-25% of the nest visible, 2) 26-50% of the nest visible, 3) 51-75% of the nest visible, and 4) 76-100% of the nest visible. For each nest, estimates were summed from all 4 directions and the mean detectability estimate was calculated. A second measure of detectability, overhead cover, was also estimated using this 1-4 scale from approximately 1 m directly above the nest. This variable was not averaged with the other cover measurements because it was an estimate of overhead cover rather than ground cover.

Determination of predator identity for each predated nest was accomplished primarily by identification of dentition imprints upon clay eggs. Nest condition was also noted and used as a secondary identifier. Predators were clumped into several classes based upon identification of the predator:

OPSM = Opossum.

RCCN = Raccoon.

SKUN = Striped skunk.

COTR = Medium-sized rodents smaller than a squirrel but larger than mice and voles. Only two species of medium-sized rodents occupied the study areas, based upon trapping results: cotton rats and marsh rice rats. Cotton rats were abundant on the study area whereas marsh rice rats were uncommon and restricted in their distribution. Therefore, I assumed the majority of predation within this class was caused by cotton rats.

SMMA = Small rodents including mice, voles, and southern flying squirrels. Several species of small rodents were captured on the study area. As listed in order of total numbers caught these included the white-footed mouse, house mouse, golden mouse, woodland vole, southern flying squirrel, and harvest mouse. All species were caught infrequently except white-footed mice. White-footed mice were the most abundant rodent of any size on the study area. The greatest densities of white-footed mice were in forested areas, but they were also caught in field edges. Because of their overwhelming abundance, I attributed predation within this class primarily to white-footed mice.

UKPD = Unknown predators who left some sign at the predated nest. This usually included real eggs only slightly damaged or slight, unknown marks on the clay egg. This category may include avian predators (whose beak could make the small holes found in many eggs), fire ants, or small rodents. It almost certainly excludes snakes, because of the appearance of shell or clay fragments, and medium-sized

mammals because of the delicacy of damage found on real eggs and slight, faint marks on clay eggs.

UKGN = Unknown predators who removed eggs without leaving any trace of their presence behind. In the literature this has usually been identified as snake (Best 1978) or avian predation (Yahner and Scott 1988).

UKMA = Unknown mammals determined from dentition imprints or from extreme damage or disruption to the nest. This includes all mammals previously mentioned including the medium-sized mammals and rodents.

Control of medium-sized mammalian predators has been conducted during several intervals over the past few decades on the study area. Most recently, active predator control began over the entire field trial grounds in 1994, and continued for a few weeks each winter throughout this study (Table 3.1).

STATISTICAL ANALYSIS

To explain patterns of predation, nest site characteristics were used as explanatory variables in a logistic regression model with nest fate as the response variable (Agresti 1990). All variables measured at each nest site were initially included in the full model and analyzed using PROC LOGISTIC (SAS 1997). These included: management type (burn or unburned), vegetation height (cm), horizontal nest cover (average of visibility index scaled from 1-4), overhead cover (overhead visibility estimate scaled from 1-4), distance to primary edge (m), distance to secondary edge (m), vegetative composition (forb, grass,

Table 3.1. Total animals removed during predator control on Ames Plantation, Grand Junction, Tennessee 1994-1997 and average home ranges and population densities of trapped species.

Species	Year				Home Range	Average Density	Estimate No. for Field Trial Course
	1994	1995	1996	1997	(ha)	(per 100 ha)	(approx. 2000 ha)
Opossum[1]	32	60	70	48	4.65	100 - 1800	2000 - 36,000
Skunk[2]	12	19	9	19	2.6 - 3.9	20 - 40	400 - 800
Raccoon[3]	2	8	7	9	49	2.1 - 20	42 - 400
Bobcat[1]	3	7	3	6	5957	0.07 - 3	1.5 - 60
Coyote[4]	2	2	3	3	800 - 4100	0.2 - 0.4	4 - 8
Grey fox[5]	0	1	1	4	75 - 653	1.2 - 2.1	24 - 42

Sources:

[1]Schwartz, C.W. and E.R. Schwartz. 1981. The wild mammals of Missouri. Univ. Missouri. Press, Columbia. 356 pp.

[2]Wade-Smith, J. and B.J. Verts. 1982. *Mephitis mephitis*. Mammalian Species 173:1-7.

[3]Lotze, J. and S. Anderson. 1979. *Procyon lotor*. Mammalian Species 119:1-8.

[4]Bekoff, M. 1977. *Canis latrans*. Mammalian Species. 79:1-9.

litter, woody vegetation) and several biologically plausible 2- and 3-way interaction terms. Distance to primary edge was assigned to one of seven 10-m classes starting with distance ≤ 10 m and ending with distance ≥ 60 m. Distance to edge was measured in distance classes rather than as a continuous variable to reduce error from visually estimating rather than actually measuring distances. Many nests did not have a measurement for secondary edge because the site lacked any other definable edge besides the primary edge. Therefore, secondary edge was analyzed as a categorical variable (0 = nest lacking secondary edge, 1 = nest possessing secondary edge). To test for differences in vegetative composition, dummy variables were created for the primary, secondary, and tertiary vegetation classes. Nests concealed within forbs, grass, or litter were compared against the woody vegetation class to detect differences in the likelihood of predation between the vegetation classes. The TEST statement in PROC LOGISTIC was used to detect differences amongst the first 3 classes (SAS 1997). The TEST statement allows comparison between and among dummy variables within the original analysis.

Several variables were also included in the full model because of their importance in the predation model created in Chapter 2. These included year (1996,1997), habitat (field or woods), trial, treatment (trapped or untrapped), and egg size (finch or bobwhite). Inclusion of these variables allowed creation of a global analysis measuring the significance of habitat variables in the presence of all other measured effects. A reduced model was created from the entire set of variables and interaction terms in the full model using the SCORE selection procedure in PROC LOGISTIC (SAS 1997). The SCORE selection fits all possible model combinations from the set of explanatory variables entered,

ranking the variable combinations by a score measuring goodness of fit. Because of the large number of variables and interaction terms involved, subsets of explanatory variables were entered into the SCORE selection process and variables from selected models from each analysis were combined. The Hosmer-Lemeshow Goodness of Fit Test was then used to test variable combinations for final model selection (Hosmer and Lemeshow 1989). Using the Hosmer-Lemeshow test, a P-value ≥ 0.05 suggests the rejection of the null hypothesis that the model does not fit the data.

Because nests were placed in pairs, I used a chi-squared test of independence to determine if the predation of 1 nest in a pair was related to the fate of the second (Ott 1993). Additionally, a chi-squared test was used to determine if the identity of the predator of 1 nest in a pair was independent of the identity of the predator of the second nest. In the comparison of predator identity, 6 predator classes (all except small mammals and cotton rats) were collapsed into a single class (others) because of the relatively few records in some classes. The chi-squared test was then used to compare fates of nest pairs with each nest classified as unpredated or 1 of the 3 remaining predator classes, small mammals, cotton rats, and other predators.

A Fisher's Exact Test was conducted to determine if the density of ground level vegetation (nest visibility) surrounding 1 nest in a pair was related to the density of the vegetation surrounding the second nest (Agresti 1990). A Fisher's Exact Test was also used to compare the primary vegetation classes around each nest in a pair to determine if vegetative composition differed between pairs.

RESULTS

Logistic Model

Twenty-one nests were censured from the analysis because of missing data (n = 939). According to the Hosmer-Lemeshow criteria, the final model adequately fit the data (P = 0.20) (Table 3.2).

There were several significant main effects in the final model. Habitat type (forest/field) was related to the likelihood of predation ($P \leq 0.001$). Nests placed in forested plots had a greater probability of being predated than those placed in field plots (69.6% as compared with 61.7% predated, respectively).

The trial period was also related to the probability of predation ($P \leq 0.001$). The likelihood of a nest being predated increased through the trials in the model. Habitat greatly influenced the effect of trial (habitat*trial interaction, $P \leq 0.001$). The likelihood of predation was greater in forest than field plots in Trial 1 (88.7% versus 51.2%). As the season progressed, however, predation decreased in forested plots dropping to 63.7% in Trial 2 and 56.2% in Trial 3. Field plots experienced the opposite effect, with predation increasing to 65.6% in Trial 2 and 68.2% in Trial 3.

The amount and type of nest cover was strongly related to the likelihood of predation on artificial nests. Nests that were more detectable at ground level were more likely to be predated than less detectable nests (P = 0.006). The majority of nests were well concealed, with 61.2% of all nests falling into the lowest detectability ranking (0-25% of the nest visible). However, a greater proportion of nests within the lowest detectability class escaped predation than would be expected by chance alone. Only 59.9% of nests in

Table 3.2. Reduced logistic regression model of predation on artificial nests including habitat explanatory variables from Ames Plantation, Grand Junction, Tennessee, 1996-1997.

Variable	DF	Parameter Estimate	Standard Error	Wald Chi-square	P -value
intercept	1	-2.98	1.02	8.53	**0.003**
cover	1	1.97	0.71	7.69	**0.006**
habitat	1	2.18	0.39	31.30	**0.001**
trial	1	0.87	0.17	25.72	**0.001**
type	1	0.12	0.42	0.09	0.770
treatment	1	1.03	0.92	1.26	0.263
overhead cover	1	0.28	0.07	15.52	**0.001**
vegetation height	1	0.01	0.01	3.00	0.083
distance to edge	1	0.08	0.16	0.23	0.632
habitat*trial	1	-0.74	0.10	54.45	**0.001**
cover*habitat	1	-0.55	0.26	4.40	**0.036**
treatment*cover	1	-1.35	0.66	4.22	**0.040**
forb cover*	1	-0.22	0.27	0.66	0.417
grass cover*	1	-0.64	0.25	6.34	**0.012**
litter*	1	-0.27	0.50	0.29	0.589
type*trial	1	-0.49	0.19	6.48	**0.011**
distance*cover	1	-0.08	0.12	0.53	0.465
treatment*distance	1	-0.40	0.21	3.74	**0.053**
trmt*dist*cover	1	0.27	0.15	0.07	0.554

*Dummy variables for primary vegetation classes.

the lowest detectability class were predated whereas 74.9% of all other nests were predated.

The amount of cover directly above the nest also was related to the likelihood of predation. Nests with greater overhead detectability had a greater chance of being predated than nests with more overhead concealment ($P \leq 0.001$). Nest predation increased slightly from nests with the lowest detectability ranking (61.3%) for overhead cover to nests with the greatest visibility (69.6%).

The type of vegetation around the nest also was related to the chance of a nest being predated. The type of vegetation varied with each individual nest site with grass frequently the primary nest cover (40.9% of nests), followed by forbs (34.4% of nests), woody vegetation (21.8% of nests), and litter (2.9% of nests). Nests placed within a grass-dominated site had a greater chance of avoiding predation than nests placed within woody vegetation ($P = 0.012$) or nests placed within forbs ($P = 0.022$). There was no difference detected between nests placed within grass and nests placed within litter ($P = 0.429$), although sample sizes for nests placed within litter were very small. Fifty-nine percent of nests placed in grass were predated compared with 62% in litter, 67% in forbs, and 75% in woody vegetation.

The effect of ground-level nest detectability was also influenced by habitat (cover*habitat interaction, $P = 0.036$). Nest detectability was more strongly related to the likelihood of predation for nests in fields than nests in forest. Field nests had better cover, on average, than nests placed in the woods (average detectability index of 1.07 as compared to 1.77). Field cover was more dense, with the majority of nests (80.0%)

falling within the lowest detectability ranking (0-25% of the nest visible). Nest cover in the forest was more sparse, with only 37.6% of nests falling within the lowest detectability ranking. Success of the least detectable artificial nests was greater than expected in both habitats, but proportionally greater in field than forest plots when compared to their availability. Artificial nests within the lowest detectability ranking accounted for 90.6% of unpredated nests in fields and only 47.9% of unpredated nests in forested habitats.

The relationship of cover around the nest to the likelihood of predation was also affected by treatment (cover*treatment interaction, P = 0.040). Nest cover was less important on experimental plots where the small mammal population was decreased. On control plots, 59.6% of all nests fell within the lowest detectability class whereas 62.9% of nests fell within the lowest detectability class on experimental plots. However, nests in the lowest detectability class accounted for a larger proportion of unpredated nests on control plots (76.5%) than experimental plots (68.8%).

The change in predation with trial was dependent upon the size of the egg (type*trial interaction, P = 0.011). Predation on finch eggs fluctuated very slightly from Trial 1 - Trial 3 (74.8 - 77.1%). Predation on bobwhite eggs declined substantially as the season progressed, from 65.4% in Trial 1 to 47.5% in Trial 3.

There was a marginal negative distance to edge effect when comparing treatments (treatment*distance to edge interaction, P = 0.053). There was no apparent distance to edge relationship on control plots. On experimental plots, however, the frequency of predation decreased in greater distance classes (Table 3.3). Nests < 50 m from the edge on control plots experienced a predation rate of 75.5% (n = 368) as compared to a

Table 3.3 Predation trends by distance classes for artificial nests on experimental plots, Ames Plantation, Grand Junction, Tennessee 1996-1997.

Distance class	Total number of nests	Percent of nests predated
0 - 9 m	53	66.0
10 - 19 m	83	57.8
20 - 29 m	96	53.1
30 - 39 m	85	57.6
40 - 49 m	68	66.2
50 - 59 m	38	47.4
≥ 60 m	52	42.3

predation rate of 72.5% for nests ≥ 50 m from the edge (n = 109). On experimental plots, however, nests < 50 m from the edge experienced a predation rate of 59.0% (n = 385) as compared to a predation rate of 44.4% for nests ≥ 50 m from the edge (n = 90).

Nest Pairs

Predation was not independent between the 2 nests of a pair (df = 1, $\chi^2 = 21.952$, P = 0.001). If 1 nest within a pair was predated, there was an increased probability that the second nest would be predated. If 1 nest within a pair went unpredated or was predated by a particular class of predator, it was likely that the second nest would follow the same fate (df = 9, $\chi^2 = 144.440$, P = 0.001). Nest detectability was not independent within a pair (Fisher's Exact Test, P ≤ 0.001). Nest detectability and therefore vegetative density tended to be similar between nests of a pair. The primary vegetative type of a nest was also not independent within a pair (Fisher's Exact Test, P ≤ 0.001). The vegetative composition was similar for both nests within a pair.

Predator Assemblage

Use of the clay eggs allowed 72.0% of predated nests to be assigned to specific predator classes (small mammal, cotton rat, skunk, opossum, raccoon). If the more general classification "unknown mammals" is included, 81.2% of predated nests could be assigned to a predator class. Rodents (both cotton rats and smaller rodents) accounted for the majority of predation (64.8%) whereas medium-sized mammals (skunks, opossums, raccoons) accounted for relatively little nest predation (7.0% of predation)(Table 3.4). Relative importance of individual predators can be compared with the condition of predated natural nests (Table 3.5) from my natural nest sample (Table 1.2).

Table 3.4. Relative importance of predators of artificial nests (n = 477 finch nests, n = 476 bobwhite nests) on Ames Plantation, Grand Junction, Tennessee 1996-97.

Predator	Finch Nests	% Predated Finch Nests	Bobwhite Nests	% Predated Bobwhite Nests	**Total**	% Total Predation	% All Nests Predated
Small rodents	168	48.1	123	44.7	291	46.6	30.6
Cotton rats	47	13.5	66	24.0	113	18.1	11.9
Unknown predators	60	17.2	16	5.8	76	12.2	8.0
Unknown mammals	31	8.9	27	9.8	58	9.3	6.1
Unknown gone	28	8.0	13	4.7	41	6.6	8.6
Skunk	8	2.1	11	4.0	19	3.0	2.0
Opossum	4	1.1	9	3.3	13	2.1	1.4
Raccoon	2	0.6	10	3.6	12	1.9	1.3
Unpredated	129	-	201	100	330	-	-
Total	477	100	476	57.8	952	100	69.9

Table 3.5. Conditions of natural nests found in old-field and open-woodland habitat after predation during summer field season 1996 and 1997, Ames Plantation, Grand Junction, Tennessee.

Species	Number predated nests	Egg(s) or fledglings removed - no predator sign	Egg shells present	Nest severely disturbed or destroyed	Nest slightly disturbed
Blue grosbeak	2	1	-	1	-
Common yellowthroat	1	-	-	-	1
Eastern meadowlark	1	1	-	-	-
Eastern towhee	6	5	-	-	1
Field sparrow	7	6	-	-	1
Indigo bunting	7	4	-	1	2
Northern bobwhite	3	2	1	-	-
Red-winged blackbird	2	1	1	-	-
Yellow-breasted chat	1	-	-	1	-
Totals	30	20	2	3	5

DISCUSSION

The degree of concealment of an individual bird nest may ultimately play an important role in the fate of the nest. Jobin and Picman (1997) found vegetation density and nest concealment good predictors of success of artificial passerine nests. Pleszczynska (1978) demonstrated empirically the effect of increasing nest cover with natural nests of lark buntings. She manipulated nest sites of female lark buntings by attaching plastic leaves directly above the nests to increase overhead nest cover and witnessed increases in nest success. Others have failed to detect differences between highly concealed artificial nests in dense vegetation, and less concealed artificial nests in more sparse vegetation (Major 1990, Esler and Grand 1993, Seitz and Zegers 1993).

Conflicting results between artificial and natural nest studies have prevented isolation of a consistent trend on the effect of vegetation density on nest predation. Martin (1989) conducted an extensive review of nesting studies for 32 species of passerines and concluded that the majority of studies found that predation decreased when dense foliage surrounded the nest, regardless of habitat. My results appear to support this conclusion. The amount and variety of available cover may have had a strong influence on survival of my artificial nests. Artificial nests with dense vegetation around the nest had a lower probability of being predated than nests with less dense cover. Open cup nests have been shown to experience greater levels of predation than domed nests that offer some overhead concealment of nest and eggs (Møller 1989, Linder and Bollinger 1995). I found artificial nests with less overhead cover were also more likely to be predated than nests with more overhead cover.

The search image and foraging strategy of a predator appear to be important in determining the effect of habitat parameters on nest predation (Martin 1987). If dense vegetative cover increases the likelihood that a nest will be successful, this strongly suggests that the predator(s) must be visually-oriented. Predators not dependent upon visual cues for location of prey would be expected to be less affected by the density of vegetation surrounding a nest. Best (1978) reported that cover had no influence on predation of field sparrow nests in Illinois where predation was mostly attributed to snakes. In my study the probability of a nest being predated increased as visibility of the nest increased. This suggests that highly visible nests were more easily detected by predators. Therefore the primary predator would be expected to be at least partially reliant upon vision.

Rodents, specifically white-footed mice and cotton rats, were the most important predators of artificial nests in this study. Within the rodent family *Muridae* (contains both white-footed mice and cotton rats), several species exhibit dependence upon vision for foraging. Hamsters showed decreased efficiency in foraging when lesions were made in the tectum of the midbrain, which controls whole-body orientation including visually dependent behavior. Blinded hamsters also failed to approach food presented to them (Finlay and Sengelaub 1981). Tactile and visual cues have been demonstrated to be more important to black rats than olfactory cues in responses to type and positioning of food (Cowan 1976). Like hamsters and black rats, white-footed mice and cotton rats may be reliant upon vision to forage efficiently. Visual cues are known to be important to white-footed mice in orientation and homing (Lackey et al. 1985), and may indicate some

reliance upon visual cues for foraging as well. The significance of the treatment-cover interaction demonstrated that artificial nests with increased cover were more likely to be successful on control plots where rodents were not trapped, as compared to experimental plots where rodents were trapped. This trend supports the hypothesis that dense vegetation may decrease the ability of mice and rats to detect prey items.

Medium-sized mammalian predators, i.e. striped skunks, raccoons, and sometimes opossums, have often been implicated as predators of artificial and natural avian nests (Picman 1988, Jobin and Picman 1997), but their effect on artificial nest predation in this study was low. Striped skunks accounted for 3% of overall predation, raccoons less than 2%, and opossums 2.1%. Where medium-sized mammals are significant predators, increased vegetative density and structural diversity may lead to decreased predation. Leimgruber et al. (1994) reported increased nest success with greater foliage density and herbaceous ground cover where raccoons and skunks were the primary predators. Using captive raccoons, Bowman and Harris (1980) demonstrated decreased predation of artificial nests with increasing structural heterogeneity. In my study, increases in vegetative density resulted in better nest concealment and may have presented a less detectable prey item for searching predators. This may decrease predation not only by small mammals, but medium-sized mammals as well.

Inverse relationships between vegetative cover and predation may exist in most habitats (Martin 1989), however the degree of expression of this inverse relationship may change with habitat type and with predator type. For example, Yahner and Wright (1985) and Yahner and Mahan (1996) found predation on artificial nests to be much lower in

younger clearcuts than older clearcuts or uncut stands. They concluded that the dense vegetation of young stands provided better nest concealment than less dense vegetation found in uncut or older clearcuts. I found predation overall to be greater in open woodlands than fields. I also found differences between the composition of vegetation associated with predated and unpredated nests. The amount of cover available to ground- and shrub-nesting birds was lower on forested plots than field plots. The composition of vegetation also differed between field and forest plots. Grass was more prevalent and more evenly scattered throughout field plots whereas grass was more restricted on forested plots, particularly the second year following a burn when it mostly existed in scattered clumps. The occurrence of grassy cover at nest sites on forested and field plots did not differ because I preferentially selected grass clumps as nest sites in an attempt to best mimic natural nest sites of ground-nesting birds. As a result, grass-dominated nest sites accounted for 39.7% of nests in fields and 42.0% of nests in the forest, despite decreased occurrence of grass in the forest. Reductions in the amount of available grass cover (as opposed to the frequency of occurrence of grass cover) coupled with increased nest detectability on forested sites may account for differences in predation observed between habitats.

Studies of both natural and artificial nest studies have reported negative distance to edge effects (Paton 1994). Originally, the hypothesis that nest predation increased with increasing proximity to edge was advanced because several species of nest predators including corvids and several species of medium-sized mammals have increased activity near ecotones (Gates and Gysel 1978, Wilcove 1985). Recent artificial nest studies that

have allowed the detection of small-mouthed predators indicate that where smaller animals are significant nest predators, distance to edge effects may not exist (Nour et al. 1993, Haskell 1995*a*). I detected no distance to edge effects on control plots, where rodents dominated the predator community. There are 2 possible explanations for this result: 1) because rodents are ubiquitous and distributed throughout the habitat rather than clumped along habitat edge, no edge effects exist and/or 2) my plots were found within habitat patches that were too small to allow detection of a distance to edge effect. Although I found no edge effect for control plots, experimental plots (where rodents were removed) demonstrated a negative distance to edge effect. Edge effects were most apparent at 50 m, with nests > 50 m receiving decreased predation over nests closer to the edge. Although my study sites were small (< 5 ha), this distance does agree with Paton (1994), who reported that edge effects were greatest < 50 m from an edge. My data suggest that the study plots may have been large enough to demonstrate edge effects but that these effects were only detectable when rodent populations were decreased.

Limited removal of medium-sized mammalian predators on the field trial courses at Ames Plantation may have influenced results. Removal of this class of predators could have 2 separate effects. Striped skunks, grey foxes, and coyotes consume significant quantities of several species of mice and rats and small rodents fill a smaller portion of the diets of bobcats, opossums, and raccoons (Schwartz and Schwartz 1981, Wade-Smith and Verts 1982). Therefore, the high rodent population and, subsequently, high rodent predation may have resulted from reduction of medium-sized mammalian predators.

Secondly, reductions in skunk, raccoon, and opossum populations could explain the low presence of these animals in our sample of predation.

In spite of these potential problems, I do not believe these biases significantly affected my results. Limited control of medium-sized predators during the winter on Ames Plantation likely had little effect on predator populations during the avian nesting season (May - August) and therefore probably had little or no overall effect upon my artificial nest predation data. Relatively large numbers of some species (particularly opossums and skunks) were removed over the 4 years of winter trapping. These can be compared with the total population numbers for several species of medium-sized mammals inhabiting the field trial course, which have been estimated from reported densities of these animals (Table 3.1). The total number of animals removed from year-to-year was relatively small when compared with the estimated populations on the field trial grounds. Tapper et al. (1996) found that predator populations recovered within several months after trapping ceased as replacement animals filled the roles of removed animals. On Ames Plantation, trapping is conducted for only a short period in the month of February. This leaves two months before the nesting season (late April - August) during which time natural recruitment and emigration may serve to replace trapped individuals.

CONCLUSIONS

In examining the avian composition, and the effects of rodent removal and habitat parameters on the success of artificial nests on Ames Plantation I have identified the following conclusions:

1) The avifauna of the field trial grounds consisted almost entirely of mixed-habitat species of grassland/shrubland habitats. Extremely area-sensitive grassland specialists such as grasshopper sparrow and moderately area-sensitive grassland specialists such as dickcissel and eastern meadowlark were absent, likely because patch sizes were small.
2) Thirty-six of 41 natural nests (88%) of 10 ground- or shrub-nesting birds were unsuccessful. On average, 65.8% of artificial nests were predated during a 12-day nest trial. Rodents were the predominant predators of artificial nests and may be significant predators of ground- and shrub-nesting passerines. Predation of artificial nests by medium-sized mammals appeared to be negligible.
3) Patterns and trends associated with artificial nest predation may differ between different egg sizes. Finch eggs were frequently consumed during predation of artificial nests by rodents whereas bobwhite eggs rarely were.
4) Predation of artificial nests decreased as rodent populations decreased. Compensatory predation by other classes of predators did not occur.
5) More easily detected artificial nests were more likely to be predated than nests that were better concealed.

6) Artificial nests placed within grassy cover were less likely to be predated than nests placed in forbs or woody cover.

7) Artificial nest predation did not increase near habitat edges in plots where no rodents were removed. A slight distance to edge effect was detected on plots where rodents were removed. This effect appeared strongest at distances below 50 m.

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LITERATURE CITED

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APPENDICES

Appendix A.

Scientific names for mammals and birds mentioned in the text.

Table A.1. Common and scientific names for all mammals mentioned in the text.

Common name	Scientific name
Bison	*Bison bison*
Bobcat	*Lynx rufus*
Coyote	*Canis latrans*
Deer mouse	*Peromyscus maniculatus*
Elk	*Cervus elaphus*
Fulvous harvest mouse	*Reithrodontomys fulvescens*
Golden mouse	*Ochrotomys nuttalli*
Grey fox	*Urocyon cinereoargenteus*
Hamster	*Mesocricetus spp.*
Hispid cotton rat	*Sigmodon hispidus*
House mouse	*Mus musculus*
Marsh rice rat	*Oryzomys palustris*
Opossum	*Didelphis virginiana*
Raccoon	*Procyon lotor*
Striped skunk	*Mephitis mephitis*
Short-tailed shrew	*Blarina brevicauda*
Southern flying squirrel	*Glaucomys volans*
White-footed mouse	*Peromyscus leucopus*
Woodland vole	*Microtus pinetorum*

Table A.2. Common names, scientific names and species' codes used to identify bird species mentioned in the text.

Common name	Scientific name	Species code
American crow	*Corvus brachyrhynchos*	AMCR
American goldfinch	*Carduelis tristis*	AMGO
Bachman's sparrow	*Aimophila aestivalis*	BACS
Bell's vireo	*Vireo bellii*	BEVI
Blue-gray gnatcatcher	*Polioptila caerulea*	BGGN
Blue grosbeak	*Guiraca caerulea*	BLGR
Blue jay	*Cyanocitta cristata*	BLJA
Brown thrasher	*Toxostoma rufum*	BRTH
Brown-head cowbird	*Molothrus ater*	BHCO
Carolina Wren	*Thryothorus ludovicianus*	CAWR
Common flicker	*Colaptes auratus*	NOFL
Common yellowthroat	*Geothlypis trichas*	COYE
Dickcissel	*Spiza americana*	DICK
Downy woodpecker	*Picoides pubescens*	DOWO
Eastern kingbird	*Tyrannus tyrannus*	EAKI
Eastern towhee	*Pipilo erthrophthalmus*	EATO
Eastern wood-pewee	*Contopus virens*	EWPE
Field sparrow	*Spizella pusilla*	FISP
Grasshopper sparrow	*Ammodramus savannarum*	GRSP
Gray partridge	*Perdix perdix*	GRPA
Great crested flycatcher	*Myiarchus crinitus*	GCFL
Hairy woodpecker	*Picoides villosus*	HAWO
Henslow's sparrow	*Ammodramus henslowii*	HESP
Indigo bunting	*Passerina cyanea*	INBU
Japanese quail	*Coturnix coturnix*	COCO
Kentucky warbler	*Oporornis formosus*	KEWA
Lark bunting	*Calamospiza melanocorys*	LARB
Northern bobwhite	*Colinus virginianus*	NOBO
Northern cardinal	*Cardinalis cardinalis*	NOCA
Orchard oriole	*Icterus spurius*	OROR
Pileated woodpecker	*Dryocopus pileatus*	PIWO
Red-bellied woodpecker	*Melanerpes carolinus*	RBWO
Red-eyed vireo	*Vireo olivaceus*	REVI
Red-headed woodpecker	*Melanerpes erthrocephalus*	RHWO
Red-winged blackbird	*Agelaius phoeniceus*	RWBL
Society finch	*Lonchura domestica*	SOFI
Song sparrow	*Melospiza melodia*	SOSP
Summer tanager	*Piranga rubra*	SUTA
Upland sandpiper	*Bartramia longicauda*	UPSA
Wild turkey	*Meleagris gallopavo*	WITU
Yellow-breasted chat	*Icteria virens*	YBCH
Zebra finch	*Peophila guttata*	ZEFI

Appendix B.

Woody and herbaceous vegetation collected and recorded from Ames Plantation, Grand Junction, Tennessee 1996-1997.

Table B.1. Tree, shrub, and vine species observed in open woodlands or field plots on Ames Plantation, Grand Junction, Tennessee 1996 and 1997.

Common Name	Scientific Name
Bicolor lespediza	*Lespediza bicolor*
Blackberry	*Rubus spp.*
Black cherry	*Prunus serotina*
Black gum	*Nyssa sylvatica*
Devil's walking stick	*Aralia spinosa*
Flowering dogwood	*Cornus florida*
Grape	*Vitis spp.*
Green ash	*Fraxinus pennsylvanica*
Hickory	*Carya spp.*
Japanese honeysuckle	*Lonicera japonica*
Loblolly pine	*Pinus taeda*
Northern red oak	*Quercus rubra*
Persimmon	*Diospyros virginiana*
Poison ivy	*Rhus radicans*
Redbud	*Cercis canadensis*
Sassafras	*Sassafras albidum*
Smilex	*Smilex spp.*
Southern red oak	*Quercus falcata*
Winged sumac	*Rhus copallina*
Sweet gum	*Liquidamber styraciflua*
Sycamore	*Plantanus occidentalis*
Trumpet creeper	*Campsis radicans*
Virginia creeper	*Parthenocissus quinquefolia*
White oak	*Quercus alba*
Wild strawberry	*Fragaria virginiana*
Willow oak	*Quercus phellos*
Winged elm	*Ulmus alata*
Yellow poplar	*Lirodendron tulipifera*

Table B.2. Herbaceous vegetation and grasses observed on field and open woodland plots Ames Plantation, Grand Junction, Tennessee 1996 and 1997.

Scientific Name	Scientific Name
Allium vineale	*Erianthus divaricàtus*
Ambrosia artemisifolia	*Euphorbia corallata*
Ambrosia trifida	*Festuca pretensis*
Andropogon gerardii	*Geranium carolinianum*
Andropogon elliotii	*Helianthus hirsutus*
Andropogon ternarius	*Juncus biflorus*
Andropogon virginicus	*Leptilon candénse*
Aristida spp.	*Lespedeza cunceata*
Arthraxon hispidus	*Lespedeza striata*
Bidens coronata	*Muhlenbergia schréberi*
Cassia fasciculata	*Panicum scoparium*
Cerastium spp.	*Passiflora incarnata*
Bromus racemosus	*Rudbeckia hirta*
Chasmanthium latifolia	*Solidago spp.*
Chenopodium album	*Sorghastrum nutans*
Chrysanthemum leucanthemum	*Sorgum halepense*
Cyperus esculentus	*Trifolium pratense*
Desmodium ciliare	*Xanthium pennsylvanicum*
Eragrostis spectábilis	
Erigeron annuus	

VITA

Troy Lee Ettel was born in 1973 and grew up in Georgetown, Indiana, a small town situated in the majestic hill country of southern Indiana. Wading the creeks and wandering through the fields and forests around his home he gained a deep appreciation and respect for the out-of-doors and the creatures within it. In 1991, Troy enrolled at Indiana University, intending to study the natural sciences. He graduated from Indiana University in 1995 with a B.S. in biology, and in the same year, enrolled at the University of Tennessee to pursue a masters degree in wildlife science.

Throughout his academic career, Troy has discovered the importance of integrating the lessons and values learned throughout life into a profession. He also has learned to recognize the importance of science in conservation and natural resource management. Troy hopes to apply the knowledge he has gained in a career as a conservation biologist.

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