This is to certify that the
thesis entitled

EDGE EFFECTS ON AVIAN AND MAMMALIAN
NEST PREDATOR RELATIVE ABUNDANCE AND
SPECIES RICHNESS

presented by

MICHELLE L. SMITH

has been accepted towards fulfillment
of the requirements for the

M.S. degree in ZOOLOGY

Catherine Lindell
Major Professor's Signature

3-15-04
Date

DATE DUE	DATE DUE	DATE DUE

EDGE EFFECTS ON AVIAN AND MAMMALIAN NEST PREDATOR RELATIVE ABUNDANCE AND SPECIES RICHNESS

By

Michelle L. Smith

A THESIS

Submitted to
Michigan State University
in partial fulfillment of the requirements
for the degree of

MASTER OF SCIENCE

Department of Zoology

2004

ABSTRACT

EDGE EFFECTS ON AVIAN AND MAMMLALIAN NEST PREDATOR RELATIVE ABUNDANCE AND SPECIES RICHNESS

By

Michelle L. Smith

Few studies have systematically examined patterns of species richness and relative abundance of nest predators, although numerous studies invoke these patterns as potential causes for decreased nesting success of forest songbirds. Species richness and relative abundance of avian and mammalian nest predators were compared between forest edge and interior in two landscapes, Allegan State Game Area and Fort Custer Training Center (FCTC), in southwest Michigan. One avian predator, the American crow (*Corvus brachyrhynchos*), was significantly more abundant at the edge than the forest interior at FCTC. Other predators, specifically blue jays (*Cyanocitta cristata*) and mammals, did not demonstrate an edge response. However, abundance of these groups was highly variable across years. Abundance of brown-headed cowbirds (*Molothrus ater*), a brood parasite, was greater at edges in FCTC in 2003. Total species richness for both landscapes did not differ between the edge and interior. Remote camera systems captured three predation events, two involving red squirrels (*Tamiasciurus hudsonicus*) and one, a blue jay. The high variability in response indicates a need to assess patterns of predator distributions through long-term studies and at different spatial scales.

ACKNOWLEDGEMENTS

Funding for this project was provided by the Department of Military and Veterans Affairs, Michigan State University Graduate School, Department of Zoology, Ecology, Evolutionary Biology, and Behavior Program, and Kellogg Biological Station. Permissions to conduct research at field sites were granted from Allegan State Game Area and Environmental Office of the Department of Military and Veterans Affairs. Permits to conduct wildlife studies were acquired from the All-University Committee on Animal Use and Care (#05/03-061-00) and the Michigan Department of Natural Resources (#1508). Sherman live-traps for small mammal trapping were borrowed from The Nature Conservancy and Henry Campa III.

I thank my graduate committee members, Catherine Lindell, Henry Campa III, and Barb Lundrigan, for their valuable input and suggestions on this project. Logistical support for statistical analyses was provided by Fernando Cardoso. Phil Myers and Scott Winterstein provided valuable insights regarding the design and implementation of the mammal surveys. Bert Skillen provided spatial data used in GIS analyses, and Tracy Aichele provided valuable assistance for those analyses.

I thank my field assistants, Michelle Speicher, Amy Roosenberg, Sarah Stoll, and Brian Wasielewski, for their hard work and dedication in the field. I also thank the Kalamazoo Nature Center, especially Ray Adams, Mark Miller, and John Brenneman, for their continued cooperation on this project. Special thanks are extended to fellow graduate students, Amy Majchrzak, Rosa Moscarella, and Bert Skillen, for their support and friendship.

Special thanks are extended to my family, especially Frank and Cheryl Smith, for their support. I am forever grateful for the love, patience, and support of Rob Schuster.

TABLE OF CONTENTS

LIST OF TABLES

LIST OF FIGURES

Chapter 1

INTRODUCTION

Habitat edges have been implicated as a potential cause for the recent decline of songbird populations. Few studies have systematically examined the impacts of edges on nest predators and brood-parasites, although this is commonly cited as a possible explanation for decreased nesting success of many songbirds. In this background chapter, I will provide a brief review of literature on habitat edges and discuss the impacts on populations of forest songbirds. In addition, I will specifically review literature on nest predator and brood-parasite response to fragmentation and habitat edges. I will then briefly outline the research question, as well as the objectives of my research. My aim is to provide background information regarding the impact of habitat edges on nest predator and brood-parasite populations, and explicate the importance of this information for the conservation management of songbird populations.

Background

Land use conversion is prevalent in the Unites States, particularly in the Midwest region (Robinson et al. 1995). One result of this practice has been a dramatic increase in fragmentation. Fragmentation causes physical changes in landscapes, such as decreased effective habitat area and increased habitat edges (Marini et al. 1995, Bowers et al. 1996). These changes can alter ecological processes and impact community composition and structure (Oehler and Litvaitis 1996, Manson et al. 1999). Fragmentation also increases landscape heterogeneity, diversity and complexity, potentially leading to more complex biological communities and interactions (Fagan 1999). Despite the extent and potential implications of forest fragmentation, there is little consensus on the impacts of forest fragmentation on biological communities (Donovan et al. 1997, Fahrig 2003). Studies that examine changes in biological communities in landscapes that differ in spatial configuration, or arrangement, are necessary to assess the impacts of fragmentation (Fahrig 1999).

Edges are defined as transitions between two separate vegetation types marked by an abrupt change in structural complexity (Murcia 1995). The general types of forest edges, as defined by Kremsater and Bunnell (1999) include natural (e.g. topographic changes), permanent (e.g. forests adjacent to lakes), permanent anthropogenic (forests adjacent to residential development), natural successional (e.g. edges created by fires) and anthropogenic successional (e.g. edges created by clearcuts). This transitional area of an edge, also known as an ecotone, can impact the associated communities and ecosystems both directly and indirectly. Direct effects of habitat edges include changes in air temperature, wind velocity, humidity, and radiation (Kremsater and Bunnell 1999). These abiotic changes in the microclimate can impact species distribution and abundance through differential tolerance and survival at edges (Murcia 1995, Kremsater and Bunnell 1999). Wildlife species often demonstrate different responses to edges, ranging from preference to avoidance (Murcia 1995) and therefore edge effects are often species-specific (Manson et al. 1999). Among the direct abiotic and biotic affects, there are also indirect edge effects, such as increased species interactions (Murcia 1995, Morin 1999). Species interactions (e.g. pollination, predation, parasitism) may be either beneficial or detrimental. Consequently, edges vary in the impact on species as either a negative, positive, or non-existent effect. A majority of the recent literature has focused on the negative effects of habitat edges (Murcia 1995), although it is still debated as to whether or not edge effects exist (see Paton 1994, Lahti 2001). A general consensus is that edge effects exist for 'some species, in some landscapes, some of the time' (Dijak and Thompson 2000).

Despite the great amount of literature on the topic, there are numerous misconceptions and erroneous assumptions associated with edge effects. The four most commonly cited misconceptions include 1) similarity of edge response for related species; 2) changes in biological communities occurring at fixed distances from habitat edge; 3) comparable edge response for a species at various edge types; 4) consistent edge response for a given taxon among different geographic regions (Sisk and Haddad 2003). However, research has demonstrated that edge effects cannot be generalized by taxa (e.g. Chalfoun et al. 2002*a*), distance (e.g. Dale et al. 2000), edge type (e.g. Bayne and Hobson 1998), or geographic location (e.g. Marzluff and Restani 1999). Since research indicates the lack of a pattern of organismal responses to edges, it is therefore necessary to determine the causes underlying this phenomenon of 'edge effects' to assess the impact of edges on wildlife communities.

To understand the potential impact of habitat edges, it is necessary to critically examine proposed mechanisms, or causes, underlying edge effects. Several general mechanisms of edge effects have been proposed (Fagan 1999), including:

1) Differential movement of organisms at habitat edges, for example, edges may act as a filter or barrier for some species;
2) Increased mortality of organisms at habitat edges;
3) Differential reproductive subsidies at habitat edges, for example, increased food resources at edges; and
4) Increased species interactions at habitat edges, for example, increased pollination, predation or parasitism at edges.

While these four hypotheses have been addressed in the literature to varying degrees, there has been a substantial amount of research on the changes in biological interactions, in particular, nest predation and parasitism of songbird populations.

A recent focus in research has been the impact of fragmentation and habitat edges on songbird populations. Many songbird populations have suffered dramatic declines in recent decades (Böhning-Gaese et al. 1993). These population declines have been attributed to effects associated with increased levels of fragmentation and subsequent increase in habitat edges (Robinson et al. 1995). Nesting success is of particular interest, as this factor is directly related to population viability (Martin 1987). Numerous studies provide evidence that nest predation and parasitism rates increase with greater extent of fragmentation (see Lahti 2001). Furthermore, there is evidence to suggest that nest predation and parasitism levels increase with proximity to habitat edges (see Paton 1994). However, these effects are not always observed. For example, edge effects are rarely observed in the western U.S., although they are commonly reported in the Midwest (Marzluff and Restani 1999).

The negative impacts of forest edges are often attributed to increased nest predation and brood parasitism levels as compared to forest interior (Murcia 1995, Heske et al. 2001). Several hypotheses have been proposed to explain increased levels of nest predation and parasitism including increased abundance, density, activity or species richness of nest predators and brood-parasites (Chalfoun et al. 2002*b*). It has been suggested that nest predators and brood-parasites are more abundant or active at edges (see Lahti 2001). Evidence for differential abundance or activity at edges has been found for various taxonomic groups, including deer mice (*Peromyscus maniculatus*, Bayne and Hobson 1998, Kristan et al. 2003), raccoons (*Procyon lotor*, Dijak and Thompson 2000), brown-headed cowbirds (*Molothrus ater*, Coker and Capen 1995, Evans and Gates 1997) blue jays (*Cyanocitta cristata*, Robinson and Robison 1999, Chalfoun et al. 2002*b*), and crows (Andrén 1992). There are still relatively few studies that specifically examine patterns of nest predator and brood-parasite abundance, activity or density. Furthermore, most studies focus on a single taxonomic group, therefore few studies have examined species richness at edges (Chalfoun et al. 2002*a*).

Edge and landscape effects on nest predation levels are ultimately a function of nest predator ecology (Lahti 2001). Consequently, documenting patterns of nest predator abundance, distribution, density, and species richness would improve our understanding of differential nest predation rates (Chalfoun et al. 2002*a*). However, there has been little research regarding the effects of edges and landscapes on nest predators (Lahti 2001).

Edge effects, historically generalized across taxa, are now considered to be species-specific (Sisk and Haddad 2003). Many recent studies have demonstrated that edge effects vary among different taxa, in that some species respond to edges, as avoidance or affinity to edges, whereas other species are not affected. Many nest predator species are considered to be generalists and capable of using edges. For example, Dijak and Thompson (2000) determined that raccoons were more abundant in agricultural edges than in forest interior. Studies of deer mice indicated significantly higher abundance at forest-farm boundaries as compared to forest interior (Bayne and Hobson 1998). Similar observations were recorded for the white-footed mouse in a forested landscape in the Northeast U.S. (DeGraaf et al. 1999). Corvid predators, specifically crows and blue jays, are often cited as being edge species. For example, Andrén (1992) determined that crows were more abundant in forest-agricultural patches compared to forest interior. Blue jays were more abundant in both clearcut and forest edges than in forest interior (Robinson and Robinson 1999, Chalfoun et al. 2002*b*). Brood-parasites are also often associated with edges. The brown-headed cowbird (*Molothrus ater*), a brood-parasite, was more abundant in forest openings than continuous forest (Coker and Capen 1995, Evans and Gates 1997). As edge effects are species-specific, these effects also vary with different landscape types.

Landscape context is described as the habitat composition of the surrounding landscape (Lindenmayer et al. 1999). Few studies have systematically examined the influence of landscape context on nest predator and brood-parasite ecology, although this has often been cited as an important factor related to nest predation levels (Lahti 2001). Oehler and Litvaitis (1996) assessed the numerical response of medium-sized predators, such as raccoons and skunks, in relation to habitat edges and diversity in three landscapes. While raccoons did not respond to habitat edges, abundance was positively correlated with increased landscape heterogeneity (Oehler and Litvaitis 1996). Bayne and Hobson (1997) compared landscapes fragmented by silviculture and agriculture with a contiguous forest landscape. It was determined that increased predation of ground nests in the agricultural landscape was associated with increased abundance of the red squirrel and increased diversity of predators in the landscape (Bayne and Hobson 1997). Tewksbury et al. (1998) determined that parasitism was most strongly correlated with extent of human development in the landscape, as well as host abundance. Blue jays were found to be more abundant in forest fragmented by logging in comparison with uncut forest (Robinson and Robinson 1999). A meta-analysis by Chalfoun et al. (2002*a*) indicated that nest predators most often (41.2%) responded positively to edges in landscapes dominated by agriculture. Positive responses of predators through increased activity, abundance, or species richness were less common (12.9%) in forest landscapes and non-existent (0%) in grasslands (Chalfoun et al. 2002*a*). This pattern has been attributed to differential resources associated with these habitats, in that agriculture may provide additional food and hosts, whereas clear-cut or grassland habitats provide relatively few resources. A recent review provided support for this idea, indicating that edge effects were more likely to be significant in agricultural landscapes than in other landscapes (Lahti 2001).

The extent of fragmentation within a landscape also has an impact on wildlife populations. Small mammals often vary in abundance or density in response to fragmentation. For example, Yahner (1992) found that white-footed mice and southern red-backed voles were more prevalent in areas with relatively more fragmentation, although this was not the case for the masked shrew (*Sorex cinereus*). Abundance of small mammals also varied by year, where variation was attributed to changes in microenvironmental conditions (Yahner 1992). Generalist predators, specifically medium-sized mammals and corvids, are often associated with increased fragmentation. This idea was supported by the research of Oehler and Litvaitis (1996) that indicated greater abundance of raccoons and wild canids with increased extent of habitat fragmentation. In a study of corvids, Andén (1992) found that predation by crows increased relative to the amount of agriculture habitat in the landscape. Furthermore, abundance of the key predator (hooded crow, *Corvus corone*) was greatest in a heterogeneous forest-agriculture landscape (Andrén 1992). In addition, parasitism by brown-headed cowbirds, a brood-parasite, decreased with increasing forest cover; however, extent of human development and host abundance were better predictors of parasitism rates (Tewksbury et al. 1998). While landscape fragmentation and configuration are considered to be important factors affecting abundance and species richness of predator assemblages, few studies have examined these factors at the landscape scale (Fahrig 1999). Documenting species richness and abundance of communities in landscapes that vary spatially is a necessary component of fragmentation research (Fahrig 1999).

Knowledge of nest predator response to edge and landscape configuration is important to understanding the causes underlying differential predation rates on breeding birds. However, few studies have systematically examined patterns of nest predator abundance or species richness with regard to edges and landscape configuration.

Statement of Problem

Population declines of some songbirds have been attributed to increased forest fragmentation and amount of edge (e.g. Robinson et al. 1995). Nesting success is ultimately related to the ecology of nest predators and brood-parasites (Chalfoun et al. 2002*a*). Therefore, understanding patterns of nest predator and brood-parasite abundance and species richness is essential to understand effects of fragmentation and edges on breeding birds.

Objectives

The purpose of this research was to address knowledge gaps on nest predator and brood-parasite response to fragmentation and edges. Species richness and relative abundance of avian and mammalian species were compared between two fragmented landscapes, and relative abundance was compared between forest edge and interior as an indication of edge response. This research specifically addressed the hypotheses that 1) relative abundance of nest predator and brood-parasite species is greater at edges as compared to forest interior, and 2) nest predator species richness is greater at edges than in the forest interior.

Chapter 2

Edge Effects on Avian and Mammalian Nest Predator Relative Abundance and Species Richness

Abstract

Few studies have systematically examined patterns of species richness and relative abundance of nest predators, although numerous studies invoke these patterns as potential causes for decreased nesting success of forest songbirds. Species richness and relative abundance of avian and mammalian nest predators were compared between forest edge and interior in two landscapes, Allegan State Game Area and Fort Custer Training Center (FCTC), in southwest Michigan. One avian predator, the American crow (*Corvus brachyrhynchos*), was significantly more abundant at the edge than the forest interior at FCTC. Other predators, specifically blue jays (*Cyanocitta cristata*) and mammals, did not demonstrate an edge response. However, abundance of these groups was highly variable across years. Abundance of brown-headed cowbirds (*Molothrus ater*), a brood parasite, was greater at edges in FCTC in 2003. Total species richness for both landscapes did not differ between the edge and interior. Remote camera systems captured three predation events, two involving red squirrels (*Tamiasciurus hudsonicus*) and one, a blue jay. The high variability in response indicates a need to assess patterns of predator distributions through long-term studies and at different spatial scales.

Introduction

Fragmentation, the isolation of viable habitat patches, decreases effective habitat area relative to edge habitat (Marini et al. 1995, Fahrig 2003). Forest fragmentation has been implicated as a potential cause of population declines of passerines in North America (e.g. Robinson et al. 1995). This is a particular concern in the midwestern U.S., as fragmentation is prevalent and nest predation levels are relatively high in this region (Robinson et al. 1995, Donovan et al. 1997). Nest predation is a primary cause of nest failure of songbirds (Martin 1993), and therefore a key factor in population viability (Marini et al. 1995). Numerous studies have demonstrated an association between fragmentation and increased nest predation in the Midwest (e.g. Gates and Gysel 1978, Andrén and Angelstam 1988, Robinson et al. 1995). Specifically, increased levels of landscape fragmentation negatively impact nesting success for many forest songbirds, with higher nest predation in more fragmented landscapes (Wilcove 1985, Donovan et al. 1997, Harley and Hunter 1998, but see Paton 1994, Lahti 2001). Previous research indicated that nest predators are influenced by landscape fragmentation, however relatively few studies examine the impact of fragmentation levels on nest predators (Chalfoun et al. 2002*a*).

Edge habitat, the interface of adjacent land cover types, is a direct result of fragmentation. Ecological consequences of increased edges include abiotic and biotic changes, such as changes in wind disturbance or dispersal ability (Murcia 1995). Consequently, the creation of edges may also change community dynamics and biological interactions (Fagan et al. 1999), such as predator-prey dynamics (Mahan and Yahner 1999). Edges often have different effects on associated communities, although recent research has focused on negative effects (Murcia 1995). In particular, forest songbirds are negatively affected by amount and proximity of edges (Hoover et al. 1995). For example, King et al. (1998*a*) found that nest predation increased with proximity to clearcut edges. Nest predation is also greater in relatively small forest tracts in comparison with larger patches, as smaller forest tracts have a greater proportion of edge habitat (Hoover et al. 1995). However, recent reviews of edge effects on nest predation indicated mixed results, suggesting that edge effects are not consistent (Paton 1994, Hartley and Hunter 1998, Lahti 2001). Nest predation levels are ultimately a function of nest predator ecology; therefore, understanding edge effects on nest predator communities may prove to be important in establishing the relationship of edges and associated effects on breeding birds (Chalfoun et al. 2002*a*).

The pattern of elevated nest predation at forest edges has been attributed to increased abundance of nest predators at edges (Lahti 2001). While many studies have supported this hypothesis, other studies have found no differences in nest predator abundance (Paton 1994). For example, Dijak and Thompson (2000) determined that raccoons (*Procyon lotor*) were more abundant at forest edges within close proximity (less than 25 m) to agricultural fields compared to the forest interior (greater than 100 m). However, Chalfoun et al. (2002*b*) did not detect any significant differences in the relative abundance of raccoons in a similar landscape and geographic region. King et al. (1998*b*) found no difference in abundance of blue jays (*Cyanocitta cristata*) at a forest clear-cut edge as compared to the forest interior. In contrast, other studies have detected greater abundances of a crow (Andrén 1992) and other avian predators (Chalfoun et al. 2002*b*) at forest edges. Few studies have systematically examined nest predator ecology, and most focus on a single response variable in a single taxon (Chalfoun et al. 2002*a*). Consequently, most studies have not examined species richness, although greater species richness is one potential explanation for increased predation at edges (Chalfoun et al. 2002*b*). Studies that simultaneously examine avian and mammalian predators are necessary to determine causes underlying differential depredation of songbird nests, as birds and mammals are important predators of nests (Bollinger and Peak 1995) and often demonstrate differing patterns in relative abundance (Nour et al. 1993).

The main objective of this study was to determine the relative abundance and species richness of avian and mammalian nest predators at edges in comparison to the forest interior in two landscapes in southwest Michigan during 2002 and 2003. This study tested the predictions that 1) nest predators would be relatively more abundant at forest edges, although it was expected that this would vary by taxonomic group; and 2) species richness of nest predator assemblages would be greater at forest edges. Total abundance of all nest predators was also compared between two forested landscapes, which provided an indication of spatial variability in the region. In addition, relative abundance of the brown-headed cowbird (*Molothrus ater*), a brood parasite, was assessed in 2003. Based on previous research (Coker and Capen 1995, Evans and Gates 1997), it was predicted that this species would also be more abundant at habitat edges. In 2003, I sought to identify local nest predators in one of the landscapes.

Methods

Study Area

Study areas were located in Allegan State Game Area (ASGA; Figure 1; 42° 32′ N, 85° 59′ W) and Fort Custer Training Center (FCTC; Figure 2; 42° 18′ N, 85° 19′ W) in southwest Michigan. Both areas consist of a heterogeneous matrix of forest and other land cover types, such as grasslands and wetlands. ASGA and FCTC differ in total area, approximately 20,000 ha and 3,000 ha respectively. Both areas are subject to management activities such as timber harvesting and vegetation management. However, potentially invasive activities, such as timber-harvesting in the vicinity of sampling sites, were suspended for the duration of this study.

Common tree species in the study areas included oak (*Quercus* spp.), maple (*Acer* spp.), aspen (*Populus* spp.), and pine (*Pinus* spp.). Understory largely consisted of sassafras (*Sassafras albidum*), dogwood (*Cornus* spp.), elderberry (*Sambucus canadensis*), spice bush (*Lindera benzoin*), and multiflora rose (*Rosa multiflora*).

Survey Design

Sampling was conducted in deciduous/mixed forest stands. The adjacent edge habitat, defined by Paton (1994) as an opening of three times the canopy height, consisted of forest adjacent to grasslands at least 3,600 m^2 in area. The physical appearance of the edge was considered to be a canopy dripline edge, or edges with an abrupt change in plant biomass and height with dense understory (Murcia 1995).

Random sampling points were selected in forested areas using a stratified scheme so that points were within two distance-from-edge classes: 1) distances less than 50 m

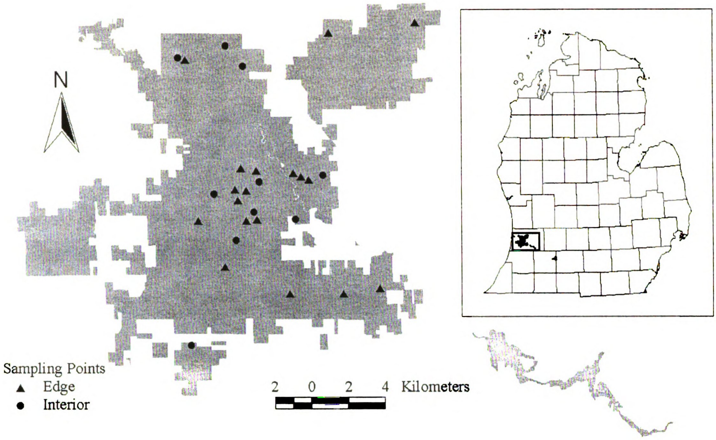

Figure 1. Sampling points in Allegan State Game Area (ASGA) located in Allegan County, MI. Locator map (inset) of ASGA in the lower peninsula of Michigan.

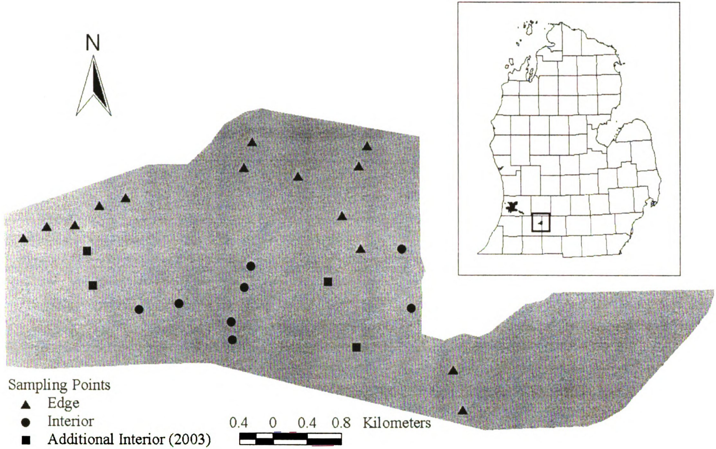

Figure 2. Sampling points in Fort Custer Training Center (FCTC) located in Kalamazoo and Calhoun Counties, MI. Locator map (inset) of FCTC in the lower peninsula of Michigan.

from the habitat boundary (edge); and 2) distances more than 200 m from a habitat boundary (interior; Figure 3). The zero-marker for distance classes was the base of the tree trunks at the edge boundary. The edge distance was considered to be the area in which edge effects i.e. biological changes in the community most likely occur; and the interior distance was considered to be the point at which edge effects diminish (Paton 1994).

Sampling points were at least 200 m apart, which is sufficient to ensure independence of sampling points (Oehler and Litvaitis 1996, Dijak and Thompson 2000). Amount and proximity of other features, such as water bodies and roads, were standardized as much as possible. ASGA had 10 interior and 18 edge points, for a total of 28 sampling points (Figure 1). FCTC had 8 interior and 14 edge points in 2002, and an additional 4 interior points in 2003, for a total of 26 sampling points (Figure 2).

Survey Methods

Avian and mammalian nest predators and brood parasites were surveyed simultaneously from June through August of 2002, and May through August of 2003. Observers specifically avoided wearing bright colors during avian surveys (Riffell and Riffell 2002). All observers used unscented personal products and scent-eliminator products prior to field work and for the duration of the study. All surveys were conducted in clear weather.

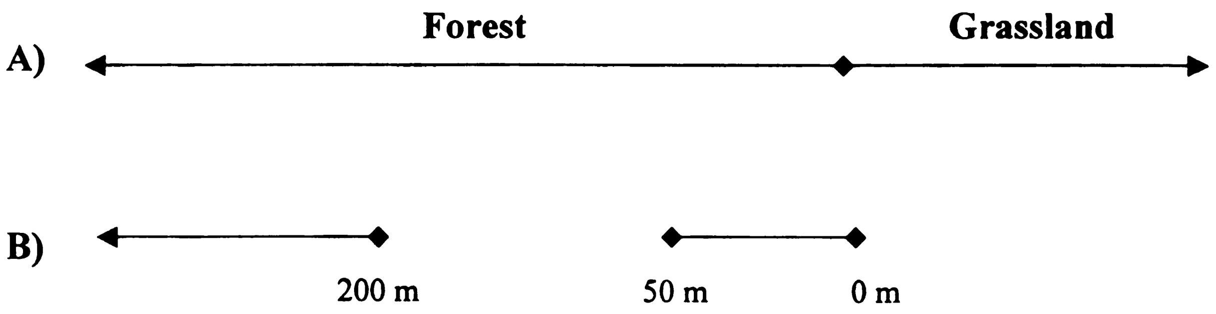

Figure 3. Schematic of the survey design for sampling points. A) Focal vegetation types of forest and grassland. B) Sampling points were located in forest stands and were categorized as either edge or interior. Edge points were within 50 m of the forest- grassland habitat boundary. Interior points were at least 200 m from any edge.

Point counts were used to provide the measure of relative abundance of the avian nest predators and brood parasites (Thompson et al. 1998). Fixed-radius (50 m) 10-minute point counts were used according to the protocol in Ralph et al. (1993). Point counts were conducted in weather with little or no wind, from sunrise to three hours after sunrise (Ralph et al. 1993). Each point was sampled at least three times throughout the field season in 2002, and six times in 2003.

Mammals were surveyed with track stations (Thompson et al. 1998), which were set up in close proximity to the established sampling points. Survey design and operation for aluminum track stations follows protocol of Justice (1961). Track plates (0.30 m x 0.45 m) were prepared with a modified chalk substrate as described by Drennan et al. (1998). Plates were left unbaited at sampling points for approximately 24 hours of exposure (Justice 1961). Animal tracks left in the substrate were preserved and cataloged. Tracks from track stations were identified using guides to mammal tracks (e.g. van Apeldoorn et al. 1993). Each point was sampled at least five times (except one point which was sampled only four times) in 2002, and eight times in 2003.

In August of 2003, small mammals were also surveyed in two trapping sessions using Sherman live-traps baited with black-oil sunflower seeds and corn-oat-molasses mix. A total of four traps were set randomly placed in close proximity (within 5 m) to each sampling point overnight, checked and rebaited daily for each 3-day trapping session. Data recorded included condition of traps (closed, open), identification of individuals, and number of individual captures per sampling point. Traps that were closed and empty were not counted for sampling effort. Captured animals were marked by clipping a small patch (3 mm^2) of the guard hairs on the dorso-caudal end. Live trapping and track plate surveys were not conducted simultaneously.

Nest predators were identified by monitoring active nests of wood thrush (*Hylocichla mustelina*) with a remote camera system (TrailMaster® Passive Game Monitor 550). Nests were located by a collaborating research group (Kalamazoo Nature Center). Cameras were set up in close proximity to nests (within 2 m) shortly after nests were located. Of the 8 nests that were monitored, 6 were still in the egg stage and 2 were in the nestling stage. Cameras remained on nests until the nest finished, either because of predation or a successful fledging event. Nest predators were identified from photographs after predation events.

Vegetation composition and structure was characterized once at each sampling point. Approximately half of the points were surveyed in July of 2002, and all but one of the remainder were surveyed in July of 2003. Protocol was a modified version of the James and Shugart (1970) method of habitat characterization. Variables measured within 11.3 m radius plots included the number of trees by species by diameter breast height (DBH) classes (8-23, 23-38, and > 38 cm DBH) and canopy height. Variables measured within 5 m radius circular plots were: number of logs ($\geq$ 7.5 cm in diameter, and $\geq$ 1.0 m long), number of stumps ($\geq$ 7.5 cm in diameter, and $\geq$ 0.25 m tall), number of woody stems (> 0.5 m tall, < 8.0 cm DBH in size classes 0-2.5 cm and > 2.5 cm), number of fallen branches (> 0.5 m long), and litter depth. Fifty meter line intercepts (Canfield 1941) were used to quantify percent bare ground, grasses, leaf litter, saplings, shrub, logs, snags, and canopy cover.

Land cover data for landscape metrics were acquired from the southern Lower Peninsula land cover 2000 data set (DNR). Land cover categories were originally in an Anderson Level II classification, but were reclassified for this analysis to an Anderson Level I classification. This classification consisted of seven land cover categories including forest, agriculture, shrub, urban, water, wetland and barren. A 15 km radius area centered on each study site was used to describe each landscape. Patch Analyst 3.0 (Rempel and Carr 2003) was used to calculate landscape and class level metrics. Landscape metrics included number of patches, mean patch size (ha), median patch size (ha), total edge (m), edge density (m/ha), and mean patch edge (m/patch); class metrics included total (ha) and percentage of area by land cover category. Landscape metrics were used to provide an indication of extent of fragmentation, and class metrics were used to describe the landscape context as relative proportions of cover types.

Statistical Analyses

The number of detections was used as the measure of abundance, although this does not necessarily indicate number of individuals. Abundance data did not meet normality assumptions, and transformations did not improve normality. Therefore differences in abundance between locations (i.e. ASGA or FCTC) or distance-classes (i.e. edge or interior) were tested using a repeated measures generalized log-linear model (PROC GENMOD; SAS Institute, 1999). The Pearson scale was used in all models to minimize overdispersion. All models met convergence criteria, which validates assumptions of this test. Total abundance was compared between sites using the model of location (ASGA or FCTC), year and interaction. This test was based on the number of total detections from both avian point counts and mammal track plates. The number of detections by taxonomic group was used to compare edge response. This model included treatment (edge or interior), year, and interaction. Avian abundance was analyzed by species for both predators (American crow, *Corvus brachyrhnchos*; blue jay) and the brood parasite (brown-headed cowbird). The brown-headed cowbird data were analyzed only for 2003 due to low numbers of detections in 2002. For track plate surveys, small-mammals and medium-sized mammals were grouped by family. Small mammal families included Muridae, Soricidae, and Dipodidae. Medium-sized mammal family groups included Mustelidae, Sciuridae, Didelphidae, Procyonidae, Erethizontidae, Canidae and Felidae. Some mammal tracks were difficult to differentiate between families, particularly when only partial prints were obtained. In these instances, tracks were identified to group of small mammal (fore-print < 2cm in length) or medium-sized mammals (fore-print > 2cm in length) where possible. Tracks that were identifiable only to group comprised a large portion of the detections. The exclusion of this data would have dramatically reduced the total number of detections and therefore these data were included in analyses. Tracks that could not reliably be assigned to a group, such as tracks that consisted only of scratch marks, were considered unidentified and not included in analyses.

Live-trapping data were analyzed to test for differences in the abundance of *Peromyscus leucopus* using a contingency table analysis of total number of individual live captures by treatment (PROC LOGISTIC; SAS Institute, 1999). The number of live captures was also compared to the number of track detections from track plate surveys by sampling point for detections of *P. leucopus* using the Pearson Correlation (PROC

CORR; SAS Institute, 1999). This was to test for a correlation of estimated abundances between the two survey methods.

Species richness was calculated by tallying the total number of nest predator species detected by treatment at each study site for both years. Data were analyzed using the generalized log-linear model (PROC GENMOD; SAS Institute, 1999) of location, treatment, and interaction (location x treatment).

A total of 17 vegetation variables were measured for this study. The distributions of most vegetation data were not normally distributed or transformable. Therefore the Wilcoxon-Mann-Whitney test was used to test for differences in vegetation variables between treatments within each study site. The sequential Bonferroni test was applied to table-wise p-values to reduce type I error inflation of multiple testing (Rice 1989).

An alpha level of 0.05 was considered statistically significant, and means are presented with standard errors (SE) unless indicated otherwise.

Results

Overall number of detections of avian and mammalian predators was significantly greater in FCTC (2002 total: 144; 2003 total: 187) in comparison with ASGA (2002 total: 100; 2003 total: 147) for both years (2002: χ^2=11.53, DF=1, p<0.001; 2003: χ^2=6.37, DF=1, p=0.011). Overall abundance was also significantly greater in 2002 than in 2003 (χ^2=24.81, DF=1, p<0.001). The interaction between year and location was not significant (χ^2=2.05, DF=1, p=0.152).

A total of 252 and 222 point counts were conducted in ASGA and FCTC respectively. The American crow was significantly more abundant at the forest edge than in the forest interior at FCTC in both years (Table 1, Figure 4). In ASGA, crows were more abundant at the edge in 2002, but not in 2003 (Table 2). Blue jay abundance did not differ significantly between the habitat edge and forest interior for either ASGA or FCTC (Figure 5). The brown-headed cowbird was significantly more abundant at edges in FCTC (Table 1, Figure 6). Other potential avian nest predator species present at study sites included red-tailed hawks (*Buteo jamaicensis*) and barred owls (*Strix varia*). These species were rarely observed, and therefore were not included in analyses.

In both years, there were a total of 364 and 317 track plate nights for ASGA and FCTC respectively. A total of 130 and 272 mammal tracks were observed with track plates at ASGA and FCTC, respectively. Of these tracks, approximately 70% were identifiable; the remaining tracks were considered unidentified and not included in analyses. Small mammal species identified included the white-footed mouse (*P. leucopus.*), house mouse (*Mus musculus*), voles (Genus: Microtus; *Microtus pennsylvanicus*, *M. ochrogaster,* and *M. pinetorum*), northern short-tailed shrew (*Blarina*

Table 1. Analysis[a] of the abundance of nest predators and the brood parasite at Fort Custer Training Center (FCTC) for two years[b] (2002 and 2003), two treatments[b] (edge and interior), and interaction[c]. Degrees of freedom are 1 for all factors.

Factor	Chi-square	P-value
American Crow		
Treatment	6.09	0.013
Year	0.03	0.870
Treatment*Year	0.36	0.545
Blue Jay		
Treatment	2.06	0.150
Year	13.95	< 0.001
Treatment*Year	0.09	0.764
Brown-headed Cowbird		
Treatment	4.53	0.033
Small Mammals		
Treatment	0.89	0.346
Year	17.71	< 0.001
Treatment*Year	0.12	0.727
Medium-sized Mammals		
Treatment	0.00	0.965
Year	2.20	0.138
Treatment*Year	6.34	0.011

[a] Repeated measures generalized log-linear model.

[b] Difference of Least Square Means Statistics.

[c] Type 3 GEE Analysis Statistics.

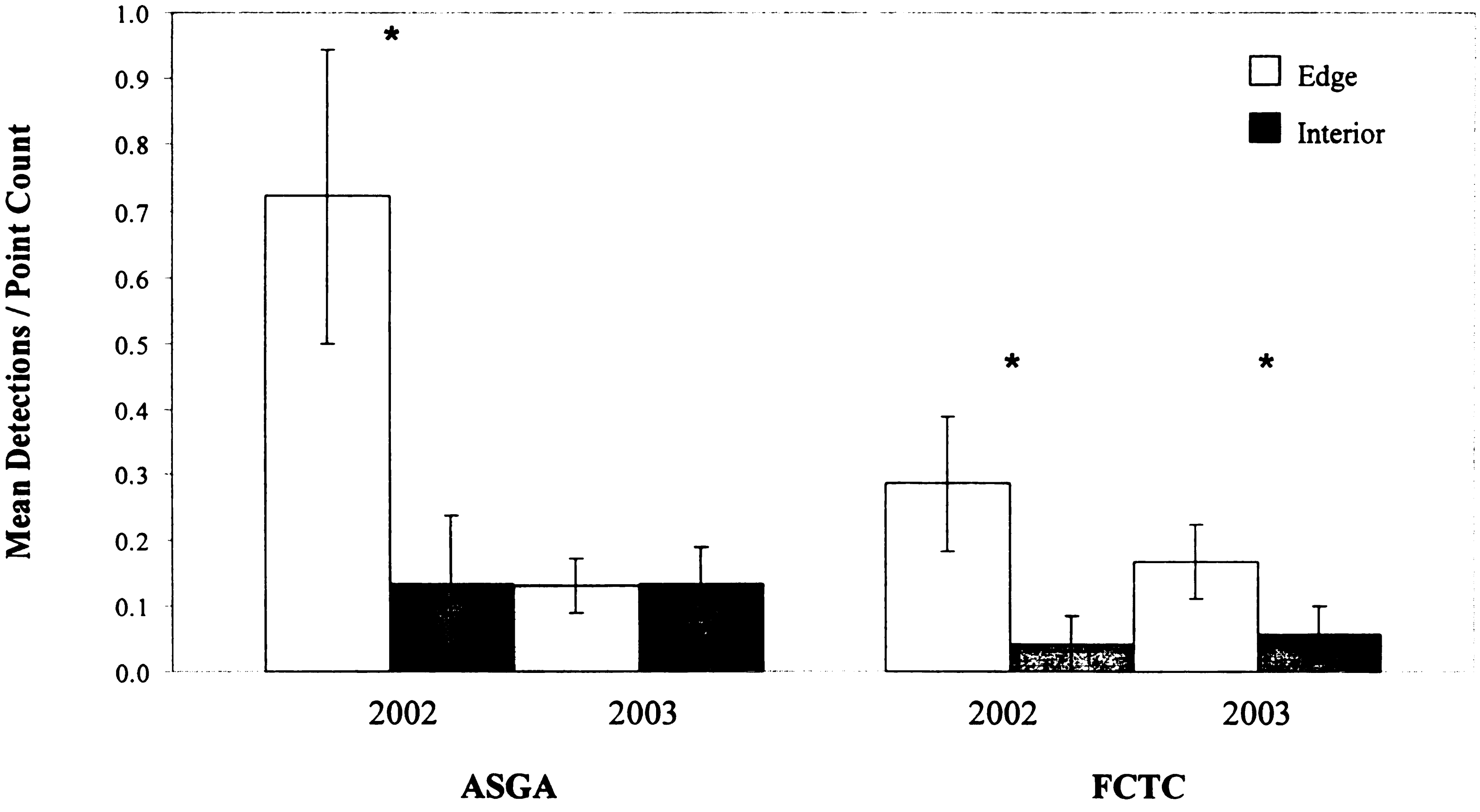

Figure 4. Mean number (± 1 SE) of crow detections per point count survey by landscape for both years (2002 and 2003). Sample size differed at Allegan State Game Area (ASGA) edge (N=18) and interior (N=10) and also at Fort Custer Training Center (FCTC) edge (N=14) and interior (N=8 in 2002; N=12 in 2003). The asterisk (*) denotes significant ($p<0.05$) differences between the edge and interior treatments at ASGA in 2002, and at FCTC in both 2002 and 2003.

Table 2. Analysis[a] of the abundance of nest predator groups and the brood parasite at Allegan State Game Area (ASGA) for two years[b] (2002 and 2003), two treatments[b] (edge and interior), and interaction[c]. Degrees of freedom are 1 for all factors.

Factor	Chi-square	P-value
American Crow		
Treatment	2.50	0.113
Year	4.41	0.035
Treatment*Year	4.05	0.044
Blue Jay		
Treatment	0.01	0.917
Year	0.34	0.560
Treatment*Year	2.01	0.156
Brown-headed Cowbird		
Treatment	0.28	0.599
Small Mammals		
Treatment	2.02	0.154
Year	0.62	0.430
Treatment*Year	0.83	0.362
Medium-sized Mammals		
Treatment	2.45	0.117
Year	0.05	0.821
Treatment*Year	0.01	0.902

[a] Repeated measures generalized log-linear model.

[b] Difference of Least Square Means Statistics.

[c] Type 3 GEE Analysis Statistics.

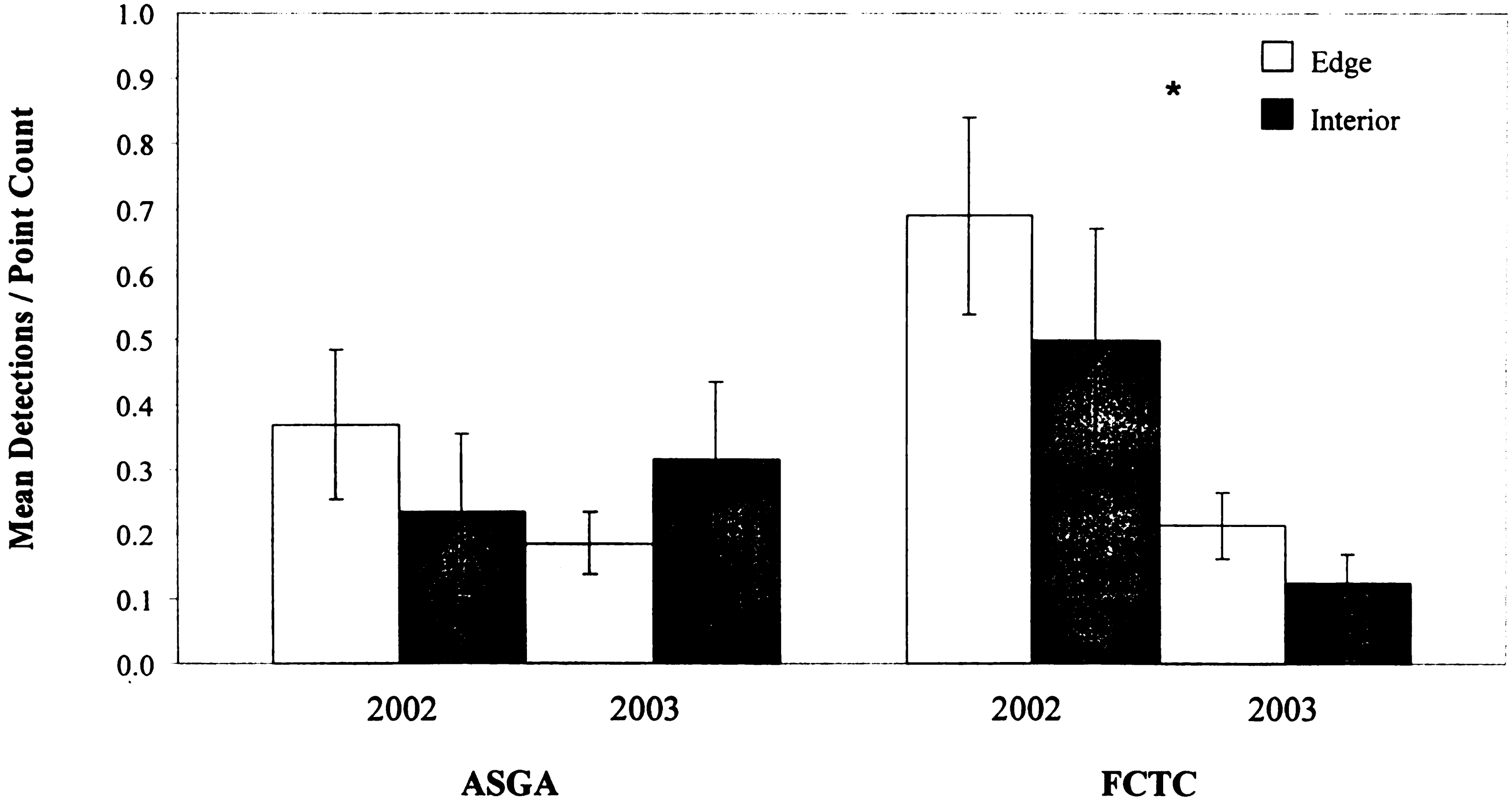

Figure 5. Mean number (± 1 SE) of blue jay detections per point count survey by landscape for both years (2002 and 2003). Sample size differed at Allegan State Game Area (ASGA) edge (N=18) and interior (N=10) and also at Fort Custer Training Center (FCTC) edge (N=14) and interior (N=8 in 2002; N=12 in 2003). The asterisk (*) denotes significant ($p<0.05$) differences between years (2002 and 2003) at FCTC.

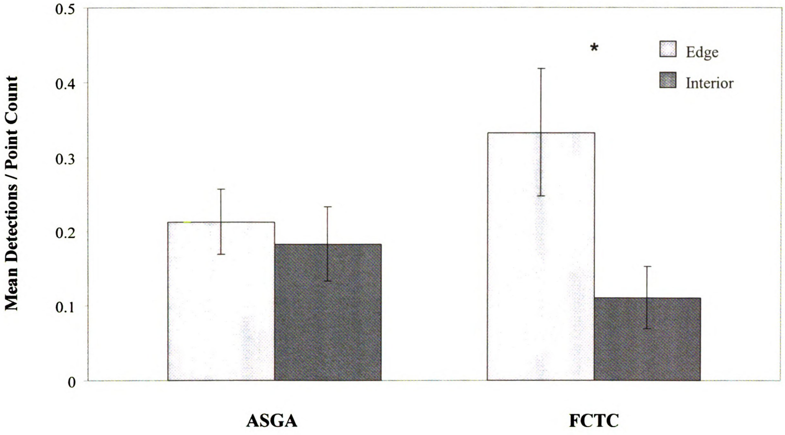

Figure 6. Mean number (± 1 SE) of brown-headed cowbird detections per point count survey by landscape for 2003 only. Sample size differed at Allegan State Game Area (ASGA) edge (N=18) and interior (N=10) and also at Fort Custer Training Center (FCTC) edge (N=14) and interior (N=12). The asterisk (*) denotes significant ($p<0.05$) differences between edge and interior treatments at FCTC.

brevicauda), and the meadow jumping mouse (*Zapus hudsonius*). Number of detections of small mammals did not differ significantly between the edge and interior at ASGA or FCTC (Table 2, Figure 7). There was, however, a significant decrease in small mammal abundance from 2002 to 2003 at FCTC (Table 1, Figure 7).

The number of detections of medium-sized mammals did not differ significantly between the habitat edge compared to the forest interior in ASGA (Table 2, Figure 8). At FCTC in 2002, medium-sized mammals were significantly more abundant in the forest interior; however, abundance was significantly greater at the habitat edge in 2003 (Table 1, Figure 8).

In two trap sessions, there were a total of 550 and 416 trap nights at ASGA and FCTC respectively. Small mammals captured with live-traps included white-footed mice (*P. leucopus*), eastern chipmunks (*Tamias striatus*), a northern short-tailed shrew, and a juvenile opossum (*Didelphis virginiana*). Although present in the study areas, the eastern chipmunk was rarely captured in either landscape (ASGA: n=4; FCTC: n=7) and therefore was not included in analyses. The single captures of a short-tailed shrew at ASGA and a juvenile opossum at FCTC were also not included in analyses. Mean detections of the white-footed mouse did not differ between the edge and interior in either landscape (ASGA: χ^2=0.27, p=0.602, DF=1; FCTC: χ^2=1.06, p=0.302, DF=1). The number of live captures and number of tracks detected from track plates were not correlated for detections of the white-footed mouse (r=0.117, p=0.567) indicating a lack of correlation in abundance estimates between the two survey methods.

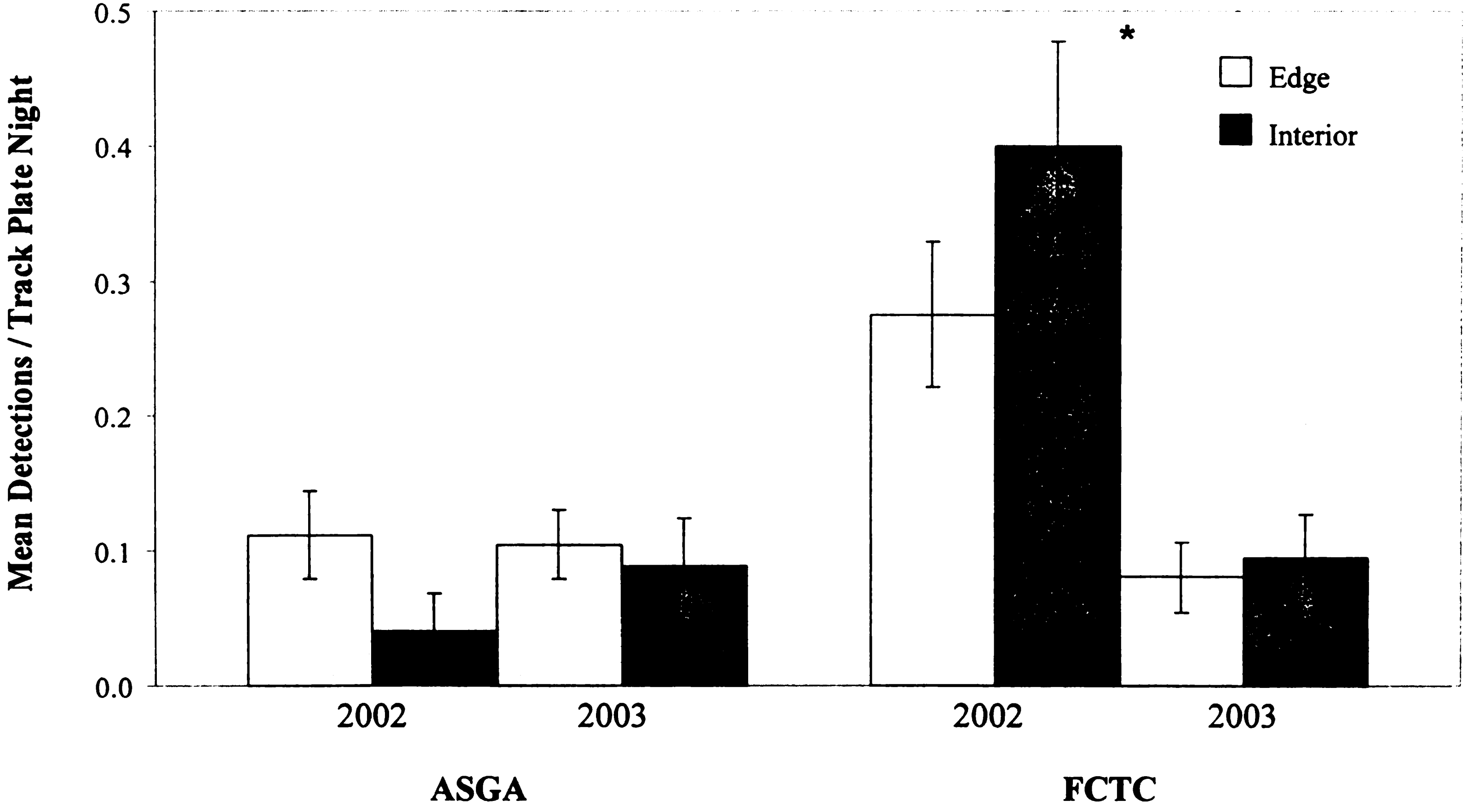

Figure 7. Mean number ($\pm$ 1 SE) of small mammal detections per track plate night by landscape for both years (2002 and 2003). Sample size differed at Allegan State Game Area (ASGA) edge (N=18) and interior (N=10) and also at Fort Custer Training Center (FCTC) edge (N=14) and interior (N=8 in 2002; N=12 in 2003). The asterisk (*) denotes significant ($p<0.05$) differences between years (2002 and 2003) at FCTC.

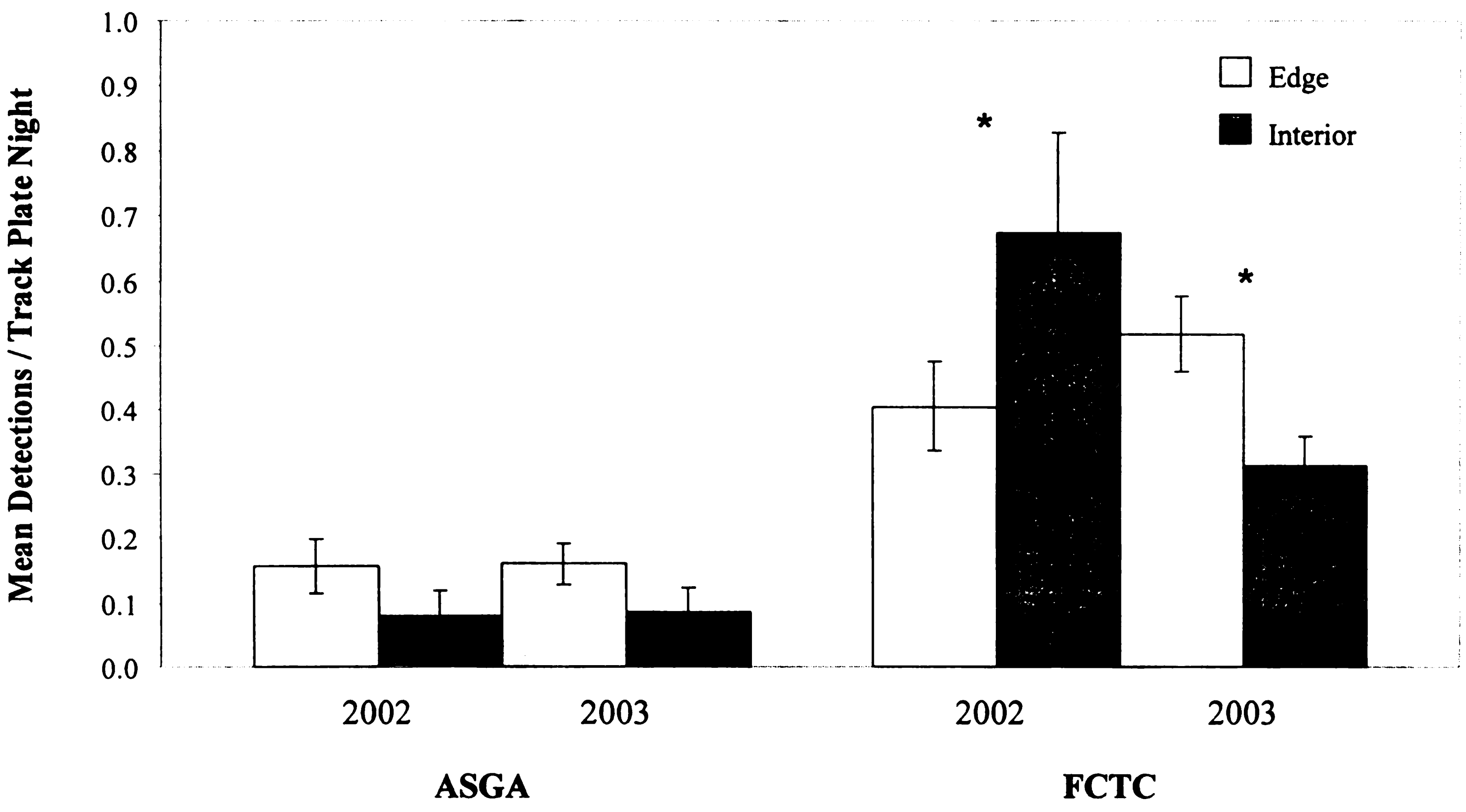

Figure 8. Mean number (± 1 SE) of medium-sized mammal detections per track plate night by landscape for both years (2002 and 2003). Sample size differed at Allegan State Game Area (ASGA) edge (N=18) and interior (N=10) and also at Fort Custer Training Center (FCTC) edge (N=14) and interior (N=8 in 2002; N=12 in 2003). The asterisk (*) denotes significant ($p<0.05$) differences between the edge and interior treatments at FCTC in both 2002 and 2003.

A total of 20 avian and mammalian species were identified in both landscapes, with 18 and 19 species detected at both the edge and interior, respectively. Most species were observed in both landscapes and both edge and interior. Three species, including a fox (Family: Canidae, one detection), porcupine (*Erethizon dorsatum*, one detection), and jumping mouse (one detection), were only observed in the interior habitat. A badger (*Taxidea taxus*) and opossum, both with one detection each, were found only at edges. There were no significant differences in species richness between locations (χ^2=1.33, p=0.248, DF=1) or treatments (χ^2=0.17, p=0.679, DF=1).

Cameras monitored nests for a total of 1,712 camera hours. Of the eight nests monitored, six were depredated, and the other nests were abandoned. Three predation events were captured on film, two of red squirrels (*Tamiasciurus hudsonicus*) and one of a blue jay.

Vegetation characteristics did not differ between the forest edge and interior for either ASGA or FCTC (Table 3, Table 4). Based on the landscape-level analysis, total edge density was greater in ASGA than in FCTC (Table 5). The number of patches was higher in ASGA than in FCTC, however, mean patch size was greater in FCTC than ASGA. Other categories of land use were comparable in both ASGA and FCTC, including the relatively dominant cover types of agriculture, shrub land, and urban land uses.

Table 3. Vegetation characteristics for Allegan State Game Area (ASGA) by treatment (edge and interior). All sampling points, 10 interior and 18 edge, were surveyed once for vegetation. Degrees of freedom are 1 for all factors.

	Means			
Variable	Edge	Interior	U[a]	P-value[b]
No. Small Snags	9.3	10.6	163.5	0.386
No. Large Snags	0.8	1.0	138.0	0.738
Canopy Height (m)	14.3	16.0	184.0	0.053
Litter Depth (cm)	3.7	4.0	145.5	1.000
No. of Logs	1.2	1.4	142.5	0.919
No. of Stumps	0.27	0.80	155.5	0.527
No. Small Stems	64.2	50.6	131.0	0.517
No. Large Stems	2.3	3.9	162.5	0.385
No. Branches	22.8	22.9	139.5	0.810
% Grass	0.023	0.049	168.5	0.270
% Bare Ground	0.019	0.002	117.0	0.185
% Leaves	0.857	0.868	151.5	0.773
% Ground Vegetation	0.002	0.012	145.5	1.000
% Saplings	0.007	0.012	158.5	0.533
% Shrub	0.030	0.042	167.0	0.302
% Log / Snag	0.017	0.016	156.0	0.614
% Canopy	0.748	0.833	173.5	0.179

[a] Test statistic for the Wilcoxon-Mann-Whitney test.

[b] Sequential Bonferroni test applied to presented p-values. Statistical significance is designated at the level of $\alpha < 0.002$.

Table 4. Vegetation characteristics for Fort Custer Training Center (FCTC) by treatment (edge and interior). All but one sampling point, 12 interior and 13 edge points, were surveyed once for vegetation. Degrees of freedom are 1 for all factors.

	Means			
Variable	Edge	Interior	U[a]	P-value[b]
Small Snags	9.6	10.8	138.0	0.339
Large Snags	0.6	2.2	201.5	0.010
Canopy Height	16.0	17.0	186.0	0.103
Litter Depth	1.9	2.2	169.5	0.479
No. of Logs	1.0	3.0	187.0	0.084
No. of Stumps	0.15	0.08	150.5	0.629
No. Small Stems	49.2	49.6	167.5	0.549
No. Large Stems	3.9	4.1	146.0	0.602
No. Branches	36.1	35.1	175.0	0.313
% Grass	0.022	0.002	116.0	0.029
% Bare Ground	0.079	0.022	161.5	0.785
% Leaves	0.705	0.743	162.5	0.744
% Ground Vegetation	0.057	0.007	129.5	0.157
% Saplings	0.007	0.003	156.0	1.000
% Shrub	0.162	0.130	132.5	0.210
% Log / Snag	0.016	0.038	184.5	0.127
% Canopy	0.728	0.847	201.5	0.014

[a] Test statistic for the Wilcoxon-Mann-Whitney test.

[b] Sequential Bonferroni test applied to presented p-values. Statistical significance is designated at the level of $\alpha < 0.002$.

Table 5. Comparison of spatial statistics [a] for both study areas [b], Allegan State Game Area (ASGA) and Fort Custer Training Center (FCTC).

		Study Area	
Level	Metrics	ASGA	FCTC
Landscape	Number of Patches	80187	74815
	Mean Patch Size (ha)	0.877	0.940
	Median Patch Size (ha)	0.09	0.09
	Total Edge (m)	27,239,593	25,039,009
	Edge Density (m/ha)	387	356
	Mean Patch Edge (m/patch)	339	334
Class [c]	Forest Area (ha) (%)	26326 (37%)	24711 (35%)
	Agriculture Area (ha) (%)	24033 (34%)	25878 (36%)
	Shrub Area (ha) (%)	12984 (18%)	8881 (12%)
	Urban Area (ha) (%)	3568 (5%)	7321 (10%)
	Water Area (ha) (%)	2241 (3%)	2047 (3%)
	Wetlands (ha) (%)	987 (1%)	1323 (2%)
	Sand/Soil/Other (ha) (%)	194 (<1%)	163 (<1%)

[a] Analysis of landscape and class metrics (Patch Analyst 3.0) is based on a 15 km radius landscape centered in each area, clipped from the 2000 southern lower peninsula coverage (Data Source: DNR). Minimum mapping unit for data set was 30 m^2.

[b] Total area assessed for both sites is standardized for comparisons (70,327 ha).

[c] Class metrics are rounded to the nearest whole number.

Discussion

Overall abundance of avian and mammalian predators was significantly higher in FCTC than ASGA for both years. The study areas did not differ substantially in amount of forest cover or edge density, although these factors are suggested as being important for the distribution and abundance of species. It is possible that slight differences in amount or type of forest may influence nest predator communities. This study examined fragmentation with a very general land use classification which may have been too coarse to detect biologically meaningful differences in extent of fragmentation. Other landscape variables not examined in this study, such as configuration or patch isolation, may also impact overall nest predator abundance.

Previous studies indicated that birds and mammals respond differently to edges (e.g. Nour et al. 1993). Therefore it was expected that different nest predator taxa would demonstrate various patterns in relative abundance at edge and interior sites. In general, this study demonstrated few significant differences in nest predator abundance between forest interior and edge habitats. Only one nest predator species, the American crow, was significantly more abundant at edges at one location, FCTC, for both years. At ASGA, crows were more abundant at the edge habitat in comparison to the interior in 2002 only. This finding supports results of previous studies, indicating that avian predators are more abundant (Chalfoun et al. 2002*b*) and more active at edges (Nour et al. 1993, Söderström et al. 1998, Zanette and Jenkins 2000). However, this study did not find consistent differences in blue jay abundance, which contrasts with results of previous studies conducted in forested landscapes. For example, Robinson and Robinson (1999) determined blue jays were more abundant in clearcuts than in intact forest. Blue jays were also more abundant in forest-pasture edges than forest interior habitats (Chalfoun et al. 2002*b*). However, other studies in the Midwest region (Thompson et al. 1992, Donovan et al. 1997) indicated no significant differences in the relative abundances of avian predators at edge and interior sites. Landscape matrix, or the surrounding landscape type, is considered an important factor influencing abundance of avian predators (Chalfoun et al. 2002*a*). For example, Andrén (1992) indicated that corvid abundance was positively associated with increasing levels of agriculture in the landscape matrix. Therefore, it is important to examine the landscape matrix as this will also affect the nest predator communities.

A dramatic decline in the abundance of avian predators was evident over the course of this study, which may be indicative of population declines from 2002 to 2003. Confirmed cases of West Nile Virus (WNV) in wildlife were reported in Michigan in 2001 (CDC, Internet) and positive cases of WNV in wild birds were reported in both study locations in 2002 (USGS, Internet). While it is not possible to quantify the effect of the epidemic on study populations, it is possible that WNV may have impacted study populations. Reduced population sizes may account for the differences in avian nest predator abundance between years.

Brown-headed cowbirds were significantly more abundant at edges in one study area, FCTC. This finding is consistent with other studies that demonstrated increased abundance of cowbirds with proximity to forest canopy openings (Coker and Capen 1995, Evans and Gates 1997). However, other studies have found results contradictory to those of this study (Thompson et al. 1992, Hahn and Hatfield 1995, Bielefeldt and

Rosenfield 1997, Robinson and Robinson 1999). Differences in results may be attributed to regional differences in cowbird population dynamics (Hahn and Hattfield 1995), differences in habitat types, or relative densities of host species (Gates and Gysel 1978, Evans and Gates 1997). This study did not evaluate host densities, habitat types, or population dynamics for the brown-headed cowbird and therefore cannot identify the source of variation.

Small mammals were surveyed by live-trapping and track plates. Abundance estimates from each method by sampling point were not significantly correlated. Track plates generally underestimated abundance of small mammals, specifically *P. leucopus*. One potential explanation for this lack of correlation between the abundance estimates from the two methods is the reliability of track plate surveys for small mammals. As small mammals are smaller in body mass, it may be more difficult to obtain adequate track registries of small mammals compared to larger mammals. As a result, this group may be more likely to leave undistinguishable tracks. This study did have a large number of unidentifiable tracks (approximately 30%), and the exclusion of these data from analyses may have influenced results. However, if it is reasonable to expect that the chance of detection of small mammals does not vary between edge and interior sites, it is unlikely that track plate estimates of small mammals varied in any consistent manner between edges and interiors.

This study found no patterns in relative abundance of small mammals with either survey method, consistent with previous research in forested landscapes (Heske 1995, Donovan et al. 1997, DeGraaf et al. 1999, Chalfoun et al. 2002*b*). However, other studies found contrasting results to those reported here. For example, the deermouse (*P.*

maniculatus) was detected significantly less than predicted at shrubland edges (Kristan et al. 2003). Similarly, the deer mouse was also significantly more abundant at farm edges than the forest interior in Saskatchewan (Bayne and Hobson 1998). In contrast, a study of the white-footed mouse determined that abundance was greatest in the forest interior rather than the edge in an agricultural landscape (Wolf and Batzli 2002). Most other studies were conducted in different landscape types, which may account for differences in results. However, variables at different scales may also influence abundance of small mammals.

Abundance of small mammals may be affected by numerous variables, such as resource availability, abundance of predators and parasites, and presence of understory woody debris (Bellows et al. 2001, Wolf and Batzli 2002). Woody debris variables, such as branches and logs, did not differ between treatments within either location. Vegetation structure likely did not affect small mammal distributions at the spatial scale examined in this study. Future studies of small mammals should examine local variables, such as abundance of resources and risk of mortality, to determine the causes of differential patterns of small mammal abundance.

High variability in abundance of small mammals between years was also evident in previous research (Yahner 1992, Heske 1995, Bayne and Hobson 1998, Chalfoun et al. 2002*b*, Carignan and Villard 2002). Fluctuations of small mammal populations are common, and therefore it is reasonable to expect temporal variability (Oehler and Litvaitis 1996). Short-term studies may not be capable of differentiating between changes in population dynamics and edge response. Therefore, long-term studies may be necessary to account for population oscillations of small mammals.

In concordance with other studies in forested landscapes (Heske 1995, Donovan et al. 1997, Chalfoun et al. 2002*b*), this study found no consistent differences in abundance of medium-sized mammals. However, in a species-specific study of medium-sized mammals, Dijak and Thompson (2000) found that raccoons associated most with forest edges adjacent to agriculture. A study by Chalfoun et al. (2002*b*) was conducted in a similar region and landscape; however, their results indicated no significant differences in raccoon abundance between edge and interior sites. It may be important to examine raccoons separately as they may respond to edges differently than other medium-sized mammals or patterns of raccoon abundance may vary in different landscapes. While raccoons did comprise a majority of the medium-sized mammal detections in this study, there were too few detections for a separate analysis.

This study differed from others in that track plates were unbaited and scent-eliminator products were used for the duration of the study. Detections of mammals were incidental, as attractants were deliberately minimized in this study. There is evidence to suggest that wild canids avoid track plates, and abundance of these species may have been underestimated by track plate surveys (Oehler and Litvaitis 1996). It is also possible that small mammals avoided track plates as well. This inference is supported by the lack of a correlation between live-trapping and track plates for *P. leucopus*, which indicated that small mammal abundance was underestimated with the track plate method. However, relative abundance of mammals between the edge and interior were comparable. Oehler and Litvaitis (1996) suggest that, while track plates are not suitable to detect temporal variation, it is an appropriate method to assess spatial variation.

Several hypotheses have been proposed to explain spatial variation in the relative abundance of predators, specifically medium-sized mammals. The mesopredator hypothesis suggests that populations of medium-sized mammals, such as raccoons and squirrels, are released from predation pressure with the elimination of top predators, such as coyotes or bobcats (Rogers and Caro 1998). Based on this explanation, mesopredator abundance and nest predation should be directly proportional, and top predator abundance and nest success should exhibit a positive relationship. Other potential explanations regarding the increased abundance of nest predators include the "spill-over" hypothesis, increased resources at habitat edges, edges as travel lanes (see Lariviere 2003), and generalist adaptability (Manolis et al. 2002). It is necessary to examine factors influencing nest predator relative abundance to determine causes of fluctuations and potential impacts on songbird populations.

Few studies have assessed species richness of nest predators with respect to edge habitats (see Marini et al. 1995, Chalfoun et al. 2002*b*). This study did not find differences in species richness of nest predators at habitat edges in comparison to forest interior. This finding is consistent with that of Marini et al. (1995) that found no difference in species richness of avian and mammalian predators in an agricultural landscape in Illinios. However, it may be important to examine all potential nest predators of an area. For example, Chalfoun et al. (2002*b*) also included reptilian predators and did observe significant differences in species richness. The current study did not assess the relative abundance of reptilian nest predators, although they are considered important nest predators (Blouin-Demers and Weatherhead 2001, Thompson and Burhans 2003). Based on the findings of Chalfoun et al. (2002*b*), future research should examine species richness of all taxonomic groups.

This study sought to identify nest predators by recording nest predation events at active nests with remote cameras. Few predators were identified due to limited time and resources. Of the three predators observed, two were red squirrels and one was a blue jay. While this identifies these species as local nest predators, this provides insufficient information as to how important these species are with regard to overall predation rates. The technique of remotely triggered cameras was relatively successful, and would be valuable for future research. This would provide information on the assemblage of local nest predators, as well as the relative importance of different species as nest predators. Knowledge on predator composition and importance would be important to the development of effective management strategies (Martin and Joron 2003, Thompson and Burhans 2003).

The extent of biological changes induced by habitat edges varies greatly with taxon or ecological process. Increased depredation of songbird nests is considered to occur within 50 m of habitat edges, although some studies indicate that this effect extends beyond 50 m (see Paton 1994). This lack of consensus for general patterns has been attributed to poor experimental design and lack of appropriate edge definitions and descriptions (Murcia 1995). Habitat change and predation may also interact; however, few studies, including the present study, have examined potential interactions (Murcia 1995, Evans 2004). Species may also perceive the environment differently, and it may be necessary to define habitat edges based on species-specific parameters. One such approach is through the use of ecologically scaled landscape indices (ESLI) to assess species-specific responses to fragmentation and edges based on natural history characteristics modeled across multiple spatial scales (Vos et al. 2001, Gehring and Swihart 2003). Instead, many studies, present study included, have focused on comparisons of distance-classes using broadly define terms of 'edge' and 'interior'. Future research could address this issue by examining processes at several different spatial scales (Stephens et al. 2003).

Edge effects may be augmented by regional fragmentation, which has been cited as a potential cause for regional declines of songbirds (Robinson et al. 1995). Persistence of wildlife with large area or specialized habitat requirements is often dependent on amount of core forest area in a region (Wilcove et al. 1986). Therefore, preservation of core area is considered to be an important management strategy to minimize disturbance and edge effect, which may help to mitigate effects on nesting success of songbirds (Whitcomb et al. 1981, Wilcove et al. 1986, Robinson et al. 1995, Fahrig 1999, Kremsater and Bunnell 1999, Marzluff and Restani 1999).

Conclusions

Overall abundance of avian and mammalian nest predators was significantly higher in FCTC, although this could not be attributed to the examined landscape metrics. Higher abundance may be associated with other factors that were not examined in this study, such as landscape configuration. The crow was relatively more abundant at the habitat edge in FCTC for both years, and ASGA for one year. Furthermore, the brown-headed cowbird was also more abundant at edges in FCTC for 2003. This indicates that increased amounts of edge may promote crow and brown-headed cowbird populations, and subsequently decrease songbird population viability through reduced nest success. This study demonstrated no difference in abundance or species richness of other nest predator species between habitat edges and the forest interior. However, variability in the relative abundance of many nest predator species was high, which supports the findings of other research. The lack of edge effects may also be attributed to high levels of regional fragmentation. It is possible that negative effects extend well beyond the forest interior, as defined in this study, and that core forest area is sparse in this region. Therefore, preservation of core forests areas may be important to mitigate effects of regional fragmentation. Nest predator assemblages and landscape context must be taken into consideration when developing conservation management strategies. It is also necessary to determine the scale at which nest predator species perceive habitat edges, in which information on species-specific movement and foraging habits would be necessary.

APPENDICES

APPENDIX A

Table A1. Total detections of species at Allegan State Game Area (ASGA) and Fort Custer Training Center (FCTC) by treatment (edge or interior) for both years (2002 and 2003) arranged by common name.

		ASGA				FCTC			
		2002		2003		2002		2003	
Common Name	Scientific Name	Edge	Interior	Edge	Interior	Edge	Interior	Edge	Interior
American Crow	*Corvus brachyrhynchos*	39	4	14	8	12	1	14	4
American Mink	*Mustela vison*	1	0	0	0	0	1	3	0
Badger	*Taxidea taxus*	0	0	0	0	1	0	0	0
Blue Jay	*Cyanocitta cristata*	20	7	20	19	29	12	18	9
Brown-headed Cowbird	*Molothrus ater*	0	1	23	11	7	4	28	8
Chipmunk	*Tamias striatus*	0	1	5	0	6	4	11	10
Eastern Gray Squirrel	*Sciurus carolinensis*	0	2	0	0	0	0	1	0
Fox Squirrel	*Sciurus niger*	1	0	0	0	1	1	1	1
Gray Fox	*Urocyon cinereoargenteus*	0	0	0	0	0	1	0	0
House Mouse	*Mus musculus*	5	0	2	1	4	0	1	2
Long-tailed Weasel	*Mustela frenata*	0	0	0	0	2	3	1	0
Meadow Jumping Mouse	*Zapus hudsonius*	0	0	0	0	0	1	0	0
Northern Short-tailed Shrew	*Blarina brevicauda*	1	0	0	0	0	2	0	0
Opossum	*Didelphis virginiana*	1	0	0	0	0	0	0	0
Porcupine	*Erethizon dorsatum*	0	0	0	0	0	1	0	0
Raccoon	*Procyon lotor*	4	1	6	2	9	9	20	11
Red Squirrel	*Tamiasciurus hudsonicus*	1	0	3	1	1	0	1	0
Red-tailed Hawk	*Buteo jamaicensis*	1	0	0	0	0	0	0	1
Striped Skunk	*Mephitis mephitis*	1	0	0	0	4	2	4	2
Vole	*Microtus* spp.	0	1	0	0	5	1	0	0
Unidentified Mammals	-	8	5	23	13	12	15	29	20

Table A2. Total number of detections by group at Allegan State Game Area (ASGA) and Fort Custer Training Center (FCTC) by treatment (edge or interior) for both years (2002 and 2003).

	ASGA				FCTC			
	2002		2003		2002		2003	
Group	Edge	Interior	Edge	Interior	Edge	Interior	Edge	Interior
American Crow	39	3	14	8	12	1	14	4
Blue Jay	20	7	20	19	29	12	18	9
Brown-headed Cowbird	0	1	23	11	7	4	28	8
Small Mammals [a]	9	2	15	7	19	16	9	9
Medium-sized Mammals [b]	14	4	23	7	28	27	58	30
Unidentified	8	5	23	13	12	15	29	20

[a] Small mammal group includes families Muridae, Soricidae, and Dipodidae.

[b] Medium-sized mammal group included the families Mustelidae, Sciuridae, Didelphidae, Procyonidae, Erethizontidae, Canidae and Felidae.

Table A3. Analysis[a] of total species richness for both locations (Allegan State Game Area and Fort Custer Training Center) and treatment (edge and interior). Data from 2002 and 2003 are combined for both locations. Degrees of freedom are 1 for all factors.

Factor	Chi-Square	P-value
Treatment	0.17	0.679
Location	1.33	0.248
Treatment*Location	0.79	0.374

[a] Generalized log-linear model.

APPENDIX B

Table B1. Total number of live-captures, recaptures, and sampling effort[a] for white-footed mice (*Peromyscus leucopus*) at each point at Allegan State Game Area.

Point	Distance Class[b]	Captures	Recaptures	Effort[a]
1	E	4	2	15
2	E	3	6	21
3	E	2	4	18
4	E	5	4	24
5	E	3	1	17
6	E	1	0	23
7	E	8	6	17
8	E	2	5	22
9	E	0	0	22
10	E	4	8	21
11	E	4	1	22
12	E	2	1	22
13	E	3	3	17
14	E	4	5	18
15	E	1	1	22
16	E	1	0	23
17	E	0	0	24
18	E	2	4	22
19	I	1	0	19
20	I	5	6	22
21	I	0	0	14
22	I	5	1	23
23	I	4	5	20
24	I	0	0	13
25	I	0	0	21
26	I	1	0	13
27	I	3	3	18
28	I	2	1	17

[a] Sampling effort did not include traps that were found closed.

[b] Categorization of points as either edge (E) or interior (I).

Table B2. Total number of live-captures, recaptures, and sampling effort[a] for white-footed mice (*Peromyscus leucopus*) at each point at Fort Custer Training Center.

Point	Distance Class[b]	Captures	Recaptures	Effort[a]
1	E	5	1	16
2	E	2	3	18
3	E	2	0	11
4	E	1	0	11
5	E	3	5	18
6	E	2	0	5
7	E	0	0	4
8	E	3	1	14
9	E	3	0	19
10	E	2	0	21
11	E	1	4	23
12	E	0	0	20
13	E	0	0	17
14	E	0	0	8
15	I	1	1	14
16	I	5	0	18
17	I	2	1	18
18	I	5	0	18
19	I	1	0	13
20	I	6	4	19
21	I	2	2	17
22	I	4	9	20
23	I	3	1	17
24	I	2	0	21
25	I	0	3	22
26	I	1	1	14

[a] Sampling effort did not include traps that were found closed.

[b] Categorization of points as either edge (E) or interior (I).

Table B3. Comparison of captures[a] and detections[b] of white-footed mice (*Peromyscus leucopus*) by each point at Allegan State Game Area.

Point	Distance Class[c]	Captures	Detections
1	E	4	0
2	E	3	0
3	E	2	0
4	E	5	1
5	E	3	0
6	E	1	0
7	E	8	0
8	E	2	0
9	E	0	1
10	E	4	2
11	E	4	1
12	E	2	1
13	E	3	1
14	E	4	0
15	E	1	1
16	E	1	0
17	E	0	0
18	E	2	1
19	I	1	0
20	I	5	0
21	I	0	1
22	I	5	0
23	I	4	0
24	I	0	0
25	I	0	0
26	I	1	0
27	I	3	2
28	I	2	1

[a] Total number of individuals captured.

[b] Total number of tracks detected.

[c] Categorization of points as either edge (E) or interior (I).

Table B4. Comparison of captures[a] and detections[b] of white-footed mice (*Peromyscus leucopus*) by each point at Fort Custer Training Center.

Point	Distance Class[c]	Captures	Detections
1	E	1	1
2	E	5	0
3	E	2	0
4	E	5	2
5	E	1	0
6	E	6	1
7	E	2	4
8	E	4	2
9	E	3	0
10	E	2	0
11	E	0	1
12	E	1	1
13	E	5	1
14	E	2	0
15	I	2	3
16	I	1	1
17	I	3	1
18	I	2	2
19	I	0	0
20	I	3	0
21	I	3	0
22	I	2	0
23	I	1	2
24	I	0	0
25	I	0	1
26	I	0	0

[a] Total number of individuals captured.

[b] Total number of tracks detected.

[c] Categorization of points as either edge (E) or interior (I).

LITERATURE CITED

LITERATURE CITED

Andrén, H. 1992. Corvid density and nest predation in relation to forest fragmentation: a landscape perspective. Ecology, 73: 794-804.

Andrén, H., and Angelstam, P. 1988. Elevated predation rates as an edge effect in habitat islands: experimental evidence. Ecology, 69: 544-547.

Bayne, E.M., and Hobson, K.A. 1997. Comparing the effects of landscape fragmentation by forestry and agriculture on predation of artificial nests. Conserv. Biol. 11: 1418-1429.

Bayne, E.M., and Hobson, K.A. 1998. The effects of habitat fragmentation by forestry and agriculture on the abundance of small mammals in the southern boreal mixedwood forest. Can. J. Zool. 76: 62-69.

Bellows, S.A., Pagels, J.F., and Mitchell, J.C. 2001. Macrohabitat and microhabitat affinities of small mammals in a fragmented landscape on the Upper Coastal Plain of Virginia. Am. Midl. Nat. 146: 345-360.

Bielefeldt, J., and Rosenfield, R.N. 1997. Reexamination of cowbird parasitism and edge effects in Wisconsin forests. J. Wildl. Manage. 61: 1222-1226.

Blouin-Demers, G., and Weatherhead, P.J. 2001. Habitat use by black rat snakes (*Elaphe obsolete obsolete*) in fragmented forests. Ecology, 82: 2882-2896.

Böhning-Gaese, K., Taper, M.L., and Brown, J.H. 1993. Are declines in North America insectivorous songbirds due to causes on the breeding range? Conserv. Biol. 7: 76-86.

Bollinger, E.K., and Peak, R.G. 1995. Depredation of artificial avian nests: a comparison of forest-field and forest-lake edges. Am. Midl. Nat. 134: 200-203.

Bowers, M.A., Matter, S.F., Dooley, J.L., Dauten, J.L., and Simkins, J.A. 1996. Controlled experiments of habitat fragmentation: a simple computer simulation and a test using small mammals. Oecologia 108: 182-191.

Canfield, R.H. 1941. Application of the line interception method in sampling range vegetation. J. For. 39: 388-394.

Carignan, V. and Villard, M. –A. 2002. Effects of variations in mico-mammal abundance on artificial nest predation in conifer plantations and adjoining deciduous forests. For. Ecol. Manage. 157: 255-265.

CDC. West Nile Virus. [Online] Available http://www.cdc.gov/ncidod/dvbid/westnile/surv&control03Maps99_01.htm, 11/23/03.

Chalfoun, A.D., Thompson III, F.R., and Ratnaswamy, M.J. 2002*a*. Nest predators and fragmentation: a review and meta-analysis. Conserv. Biol. 16: 306-318.

Chalfoun, A.D., Ratnaswamy, M.J., and Thompson III, F.R. 2002*b*. Songbird nest predators in forest-pasture edge and forest interior in a fragmented landscape. Ecol. Appl. 12: 858-867.

Coker, D.R., and Capen, D.E. 1995. Landscape-level habitat use by brown-headed cowbirds in Vermont. J. Wildl. Manage. 59: 631-637.

Dale, V.H., Brown, S., Haeuber, R.A., Hobbs, N.T., Huntly, N., Naiman, R.J., Riebsame, W.E., Turner, M.G., Valone, T.J. 2000. Ecological principles and guidelines for managing the use of land. Ecol. Appl. 10:639-670.

DeGraaf, R.M., Maier, T.J., and Fuller, T.K. 1999. Predation of small eggs in artificial nests: effects of nest position, edge, and potential predator abundance in extensive forest. Wilson Bull. 111: 236-242.

Dijak, W.D., and Thompson, F.R. 2000. Landscape and edge effects on the distribution of mammalian predators in Missouri. J. Wildl. Manage. 64: 209-216.

DNR. IFMAP Southern Michigan Landcover.[Internet] Available http://www.dnr.state.mi.us/spatialdatalibrary/metadata/2000_slp_landcover.htm, 11/23/03.

Donovan, T.M., Jones, P.W., Annand, E.M., and Thompson III, F.R. 1997. Variation in local-scale edge effects: mechisms and landscape context. Ecology, 78: 2064-2075.

Drennan, J.E., Beier, P., and Dodd, N.L. 1998. Use of track stations to index abundance of Sciurids. J. Mammal. 79: 352-359.

Evans, D.R., and Gates, J.E. 1997. Cowbird selection of breeding areas: the role of habitat and bird species abundance. Wilson Bull. 109: 470-480.

Evans, K.L. 2004. The potential for interactions between predation and habitat change to cause population declines of farmland songbirds. Ibis, 146: 1-13.

Fagan, W.F., Cantrell, R.S., and Cosner, C. 1999. How habitat edges change species interactions. Am. Nat. 153: 165-182.

Fahrig, L. 1999. Forest loss and fragmentation: which has the greater effect on persistence of forest-dwelling animals? *In* Forest fragmentation: wildlife and management implications. *Edited by* J.A. Rochelle, L.A. Lehmann, and J. Wisniewski. Brill, Leiden. pp. 87-95.

Fahrig, L. 2003. Effects of habitat fragmentation on biodiversity. Annu. Rev. Evol. Syst. 34: 487-515.

Gates, J.E., and Gysel, L.W. 1978. Avian nest dispersion and fledging success in field-forest ecotones. Ecology, 59: 871-883.

Gehring, T.M., and Swihart, R.K. 2003. Body size, niche breathe, and ecologically scaled responses to habitat fragmentation: mammalian predators in an agricultural landscape. Biol. Conserv. 109: 283-295.

Hahn, D.C., and Hatfield, J.S. 1995. Parasitism at the landscape scale: cowbirds prefer forests. Conserv. Biol. 9: 1415-1424.

Hartley, M.J. and Hunter Jr., M.L. 1998. A meta-analysis of forest cover, edge effects, and artificial nest predation rates. Conserv. Biol. 12: 465-469.

Heske, E.J. 1995. Mammalian abundances on forest-farm edges versus forest interiors in southern Illinois: is there an edge effect? J. Mammal. 76: 562-568.

Heske, E.J., Robinson, S.K., and Brawn, J.D. 2001. Nest predation and neotropical migrant songbirds: piecing together the fragments. Wildl. Soc. B., 29: 52-61.

Hoover, J.P., Brittingham, M.C., and Goodrich, L.J. 1995. Effects of forest patch size on nesting success of word thrushes. Auk, 112: 146-155.

James, F.E., and Shugart Jr., H.H. 1970. A quantitative method of habitat description. Audubon Field Notes, 24: 727-736.

Justice, K.E. 1961. A new method for measuring home ranges of small mammals. J. Mammal. 42: 462-470.

King, D.I., DeGraaf, R.M., and Griffin, C.R. 1998*a*. Edge related nest predation in clearcut and groupcut stands. Conserv. Biol. 12: 1412-1415.

King, D.I., Griffin, C.R., and DeGraaf, R.M. 1998*b*. Nest predator distribution among clearcut forest, forest edge and forest interior in an extensively forested landscape. For. Ecol. Manage. 104: 151-156.

Kremsater, L. and Bunnell, F.L. 1999. Edge effects: theory, evidence and implications to management of western North American forests. *In* Forest fragmentation: wildlife and management implications. *Edited by* J.A. Rochelle, L.A. Lehmann, and J. Wisniewski. Brill, Leiden. pp. 117-153.

Kristan III, W.B., Lynam, A.J., Price, M.V., and Rotenberry, J.T. 2003. Alternative causes of edge-abundance relationships in birds and small mammals of California coastal sage scrub. Ecography, 26: 29-44.

Lahti, D.C. 2001. The "edge effect on nest predation" hypothesis after twenty years. Biol. Conserv. 99: 365-374.

Lariviere, S. 2003. Edge effects, predator movements, and the travel-lane paradox. Wildl. Soc. B. 31: 315-320.

Lindenmayer, D.B., Cunningham, R.B., and Pope, M.L. 1999. A large scale "experiment" to examine the effects of landscape context and habitat fragmentation on mammals. Biol. Conserv. 88: 387-403.

Linzey, A.V. 1989. Response of white-footed mouse (*Peromyscus leucopus*) to the transition between disturbed and undisturbed habitats. Can. J. Zool. 67: 505-512.

Mahan, C.G. and Yahner, R.H. 1999. Effects of forest fragmentation on behaviour patterns in the eastern chipmunk (*Tamias striatus*). Can. J. Zool. 77: 1991-1997.

Manolis, J.C., Andersen, D.E., and Cuthbert, F.J. 2002. Edge effect on nesting success of ground nesting birds near regenerating clearcuts in a forest dominated landscape. Auk, 119: 955-970.

Manson, R.H., Ostfeld R.S., and Canham, C.D. 1999. Responses of a small mammal community to heterogeneity along forest-old-field edges. Landscape ecology, 14:355-367.

Marini, M.A., Robinson, S.K., and Heske, E.J. 1995. Edge effects on nest predation in the Shawnee National Forest, southern Illinois. Biol. Conserv. 74: 203-213.

Martin, T.E. 1987. Artificial nest experiments: effects of nest appearance and type of predator. The Condor, 89:925-928.

Martin, T.E. 1993. Nest predation, nest sites, and birds: new perspectives on old patterns. BioSci. 43: 523-532.

Martin, J. –L. and Joron, M. 2003. Nest predation in forest birds: influence of predator type and predator's habitat quality. Oikos, 102: 641-653.

Marzluff, J.M., and Restani, M. 1999. The effects of forest fragmentation on avian nest predation. *In* Forest fragmentation: wildlife and management implications. *Edited by* J.A. Rochelle, L.A. Lehmann, and J. Wisniewski. Brill, Leiden. pp. 155-169.

Morin, P.J. 1999. Community ecology. Blackwell Science, Inc., New Brunswick.

Murcia, C. 1995. Edge effects in fragmented forest: implications for conservation. TREE, 10: 58-62.

Nour, N., Matthysen, E., and Dhondt, A.A. 1993. Artificial nest predation and habitat fragmentation: different trends in bird and mammal predators. Ecography, 16: 111-116.

Oehler, J.D. and Litvaitis, J.A. 1996. The role of spatial scale in understanding responses of medium-sized carnivores to forest fragmentation. Can. J. Zool. 74: 2070-2079.

Paton, P.W. 1994. The effect of edge on avian nest success: how strong is the evidence? Conserv. Biol. 8: 17-26.

Ralph, J.C., Geoffrey, G.R., Pyle, P. Martin, T.E., DeSante, D.F. 1993. Handbook of field methods for monitoring landbirds. Gen. Tech. Rep. PSW-GTR-144. Albany, CA: Pacific Southwest Research Station, Forest Service, U.S. Department of Agriculture; 41p.

Rempel, R.S. and Carr, A.P. 2003. Patch Analyst extension for ArcView: version 3. [Internet] Available http://flash.lakeheadu.ca/~rrempel/patch/index.html, March 12, 2004.

Rice, W.R. 1989. Analyzing tables of statistical tests. Evolution, 43: 223-225.

Riffell, S.K. and Riffell, B.D. 2002. Can observer clothing color affect estimates of richness and abundance? An experiment with point counts. J. Field Ornithol. 73: 351-359.

Robinson, W.D., and Robinson, S.K. 1999. Effects of selective logging on forest bird populations in a fragmented landscape. Conserv. Biol. 13: 58-66.

Robinson, S.K., Thompson III, F.R., Donovan, T.M., Whitehead, D.R., Faaborg, J. 1995. Regional forest fragmentation and the nesting success of migratory birds. Science, 267: 1987-1990.

Rogers, C.M., and Caro, M.J. 1998. Song sparrows, top carnivores, and nest predation: a test of the mesopredator release hypothesis. Oecologia, 116: 227-233.

SAS Institute Inc. 1999. SAS/STAT® User's Guide, Version 8.0e, fourth ed. Cary, North Carolina.

Sisk, T.D., and Haddad, N.M. 2003. Incorporating the effects of habitat edges into landscape models: Effective area models for cross-boundary management. Pp. 208-240 *in* Liu, J. and Taylor, W.W. (Eds.). Integrating landscape ecology into natural resource management. Cambridge University Press, Cambridge, U.K.

Söderström, B., Pärt, T., and Rydén, J. 1998. Different nest predator faunas and nest predation risk on ground and shrub nests at forest ecotones: an experiment and review. Oecologia, 117: 108-118.

Stephens, S.E., Koons, D.N., Rotella, J.J., and Willey, D.W. 2003. Effects of habitat fragmentation on avian nesting success: a review of the evidence at multiple spatial scales. Biol. Conserv. 115: 101-110.

Swihart, R.K., Gehring, T.M., Kolozsvary, M.B., Nupp, T.E. 2003. Responses of 'resistent' vertebrates to habitat loss and fragmentation: the importance of niche breadth and range boundaries. Diversity and Distributions, 9: 1-18.

Tewksbury, J.J., Hejl, S.J., and Martin, T.E. 1998. Breeding productivity does not decline with increasing fragmentation in a western landscape. Ecology, 79: 2890-2903.

Thompson, F.R., Dijak, W.D., Kulowiec, T.G., and Hamilton, D.A. 1992. Breeding bird populations in Missouri Ozark forests with and without clearcutting. J. Wildl. Manage. 56: 23-30.

Thompson, W.L., White, G.C., and Gowan, C. 1998. Monitoring vertebrate populations. Academic Press, Inc., San Diego.

Thompson, F.R. and Burhans, D.E. 2003. Predation of songbird nests differs by predator and between field and forest habitats. J. Wildl. Manage. 67: 408-416.

USGS. West Nile Virus Map. Web: http://westnilemaps.usgs.gov/. Accessed 11/23/03.

van Apeldoorn, R., el Daem, M., Hawley, K., Kozakiewicz, M., Merriam, G., Nieuwenhuizen, W., and Wegner, J. 1993. Footprints of small mammals. A field method of sampling data for different species. Mammalia, 57: 407-422.

Vos, C.C., Verboom, J., Opdam, P.F.M., and Ter Braak, C.J.F. 2001. Towards ecologically scaled landscape indices. Am. Nat. 183: 24-41.

Whitcomb, R.F., Robbins, C.S., Lynch, J.F., Whitcomb, B.L., Klimkiewicz, M.K., Bystrak, D. 1981. Effects of forest fragmentation on avifauna of the eastern deciduous forest. *In* Forest island dynamics in man-dominated landscapes. *Edited by* R.L. Burgess and D.M. Sharpe. Springer-Verlag, New York. pp 125-204.

Wilcove, D.S. 1985. Nest predation in forest tracts and the decline of migratory songbirds. Ecology, 66: 1211-1214.

Wilcove, D.S., McLellan, C.H., and Dobson, A.P. 1986. Habitat fragmentation in the temperate zone. *In* Conservation biology: the science of scarcity and diversity. *Edited by* M.E. Soulé. Sinauer, Sunderland, MA. pp 237-256.

Wolf, M. and Batzli, G.O. 2002.Effects of forest edge on populations of white-footed mice *Peromyscus leucopus*. Ecography, 25: 193-199.

Yahner, R.H. 1992. Dynamics of a small mammal community in a fragmented forest. Am. Midl. Nat. 127: 381-391.

Zanette, J.C., and Jenkins, B. 2000. Nesting success and nest predators in forest fragments: a study using real and artificial nests. Auk, 117: 445-454.