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The Role of Landscape in the Distribution of Deer-Vehicle Collisions in Two Counties in South-Central Mississippi

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THE ROLE OF LANDSCAPE IN THE DISTRIBUTION OF DEER-VEHICLE COLLISIONS IN TWO COUNTIES IN SOUTH-CENTRAL MISSISSIPPI

by

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ABSTRACT

THE ROLE OF LANDSCAPE IN THE DISTRIBUTION OF

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The number of deer killed by vehicle collisions each year in the United States exceeds the number of deer killed annually through hunting. Deer-vehicle collisions (DVCs) have a vast negative impact on the economy, traffic safety, and general well-being of otherwise healthy deer populations. To mitigate DVCs, it is imperative to gain a better understanding of the factors that play a role in their spatial distribution. Much of the existing research has been inconclusive, pointing to a variety of factors that cause DVCs that are specific to study site and region. Very little DVC research has been undertaken in the southern United States, which makes the region particularly important with regard to this issue. Through the use of GIS, remotely sensed imagery, and statistical analysis, this thesis evaluates landscape factors that contribute to the spatial distribution of DVCs within Forrest and Lamar Counties in Mississippi.

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TABLE OF CONTENTS

LIST OF TABLES

Table

LIST OF ILLUSTRATIONS

Figure

LIST OF ABBREVIATIONS

ASCII .... American Standard Code for Information Interchange

C-CAP ....Coastal Change Analysis Program

DVC .... Deer-vehicle collision

FAA .... Federal Aviation Administration

FSA .... Farm Services Agency

GIS .... Geographic information system

GPS .... Global positioning system

MARIS .... Mississippi Automated Resource Information System

MOHS .... Mississippi Office of Highway Safety

MVC .... Moose-vehicle collision

NAIP .... National Agriculture Inventory Program

NOAA .... National Oceanic and Atmospheric Administration

NWRC .... National Wildlife Research Center

SPSS .... Statistical Package for the Social Sciences

USDA .... U. S. Department of Agriculture

UVC .... Ungulate-vehicle collision

CHAPTER I

INTRODUCTION

1.1. Overview

Animal-vehicle collisions are one of the most common and dramatic forms of human-environment interaction (Gonser et al. 2009), posing a threat to traffic safety and animal welfare, and serving as a drain on economic development and conservation initiatives (Seiler 2005). In recent decades, animal-vehicle collisions, more specifically deer-vehicle collisions (DVCs), have become a matter of critical importance, given the significant impact of these collisions on wildlife, society, and the economy (Hussain et al. 2007). While the number of vehicles on U.S. roadways increased by 7 percent between 2004 and 2009, DVCs increased by 18.3 percent. Between July of 2007 and June of 2009, over 2.4 million DVCs occurred on U.S. roadways, the equivalent of 100,000 DVCs per month, or one DVC every 26 seconds (State Farm Insurance 2009)

Published research suggests that the primary factors underlying this dramatic increase in DVCs are twofold, one being the explosive growth in the North American White-Tailed Deer (*Odocoileus virginianus*) population over the past 50 years, and the other being the corresponding increase in human populations (Gonser and Horn 2007). As a result of human population growth, traffic volume has increased and transportation networks have expanded to satisfy increasing needs and demand (Hubbard et al. 2000).

The enormous white-tailed deer population has only recently become an issue, as the North American population was almost extinct in 1900 as a result of two historic periods of over-harvesting. The most recent of these occurred in the latter half of the 19th century when the North American White-Tailed Deer population declined from an estimated 18 million to a mere 500,000 individuals (Gonser and Horn 2007). Better management and reduction of natural predators (Rawinski 2008) helped the population to recover in the U.S. during the 20^{th} century. Contemporary estimates now place the population at more than 20 million individuals (Hubbard et al. 2000).

Another factor in the revival of the white-tailed deer was human migration from rural areas to cities, which left abandoned agricultural land where deer thrived (Gonser and Horn 2007). Unlike other species that do not tolerate environmental disturbance and land fragmentation, deer flourish in anthropogenic landscapes, and especially mosaics of forest and grassland. White-tailed deer are incredibly adaptable generalists endowed with exceptional survival skills. They need forested areas for protection, but thick canopies do not offer sufficient understory growth that is their preferred browse. They have therefore become adapted to exploiting human-altered environments, feeding in agricultural fields, lawns, and road sides (Rawinski 2008). Deer also take advantage of areas of secondary forest regeneration. As woody seedlings and grasses become the dominant plant species, deer turn to these areas as prime habitat for browsing. Likewise, deer are attracted to clear-cut areas and roads, in part because they create edge habitat. Most species tend to avoid ecological edges, but abrupt transitions from forest to grass communities keenly attract deer (Gonser and Horn 2007). Active and fallowed farmland, new road cuts, and clear-cut forest lands are key examples of how human activity creates heterogeneous landscapes (Turner 2005). Not only do these disturbances create a diverse mosaic of plant communities, but they also create patterns that can be useful in understanding where and why DVCs occur.

A growing deer population, however, is only one factor in the recent increase in DVCs. The expansion of road networks is also involved. Transportation networks constitute the most extensive human-built infrastructure in the United States (Malo et al. 2004). Road expansion leads to habitat fragmentation and increased edge habitat, which fundamentally alter how local ecosystems operate (Forman & Alexander 1998). Unlike many other species, deer are not adversely impacted by fragmentation and proliferation of edge habitat. In fact new road construction produces their ideal habitat by expanding or creating new grazing areas with easily accessible food sources (Gonser and Horn 2007). During the times of day when deer graze, primarily around dawn and dusk, deer routinely cross roads encompassed within their home range, intermingling with traffic as they move through open road corridors (Putnam 1997). Although roadside environments may appear to be ideal foraging habitat for deer, in reality they act as ecological traps. Millions of deer are killed annually along roads in the U.S. as they graze on or near roads (Coffin 2007). As seen in Figure 1, Conover (1997) hypothesized that when a small deer population is increasing, so will its net value with regard to monetary gains from hunting or simply from the pleasure of people seeing wild deer in rural settings. When a large deer population continues to grow, however, DVCs become more common, leading to negative aspects that outweigh these positive values. Rather than symbolizing majestic nature, deer are increasingly seen as pests in need of management or removal (Conover 1997).

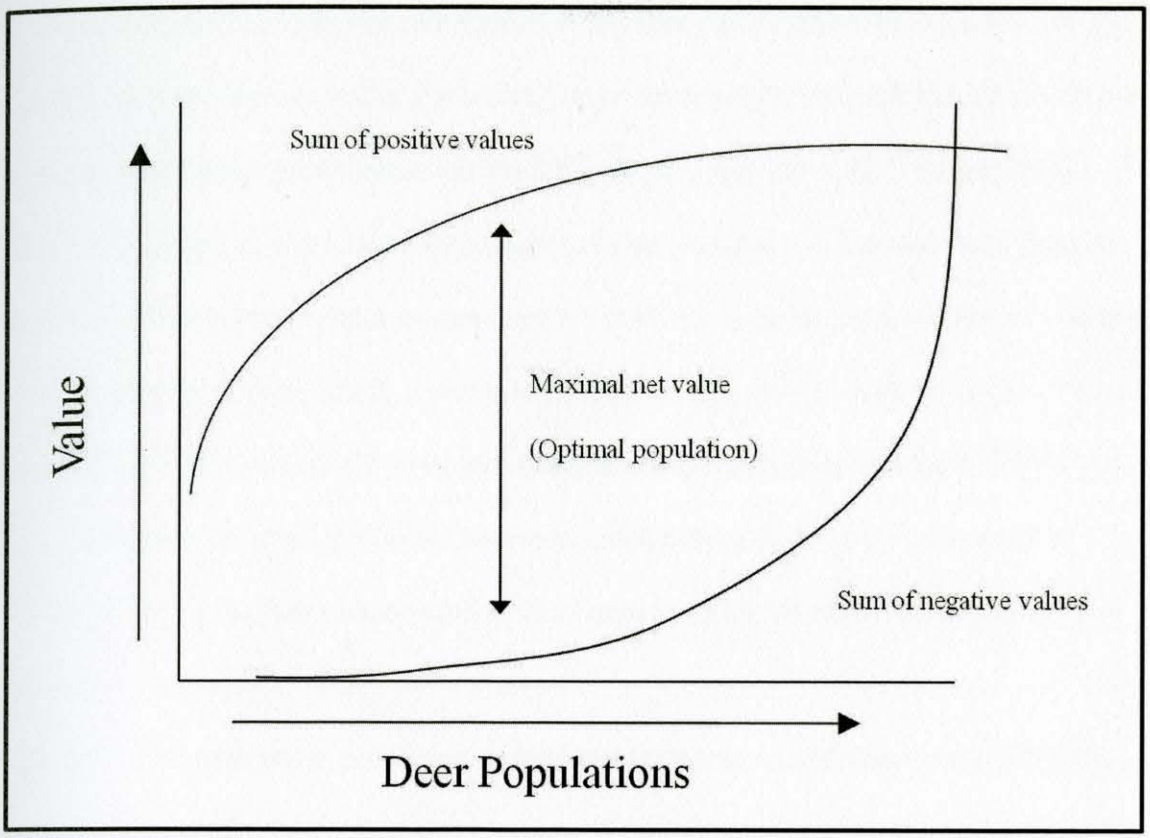

Figure 1. Net Value of Deer with Population Growth, from Conover (1997, p. 303).

1.2. Statement of the Research Problem

Over the past 30 years, DVCs have increased dramatically as a result of more vehicle traffic and higher populations of deer (Seiler 2005). Record deer populations can be attributed in part to the decline of hunting in the U.S. From 1991-2001, 5 percent of the total population of the U.S. participated in deer hunting. Between 2001 and 2006, the number of deer hunters declined by 4 percent (U.S. Fish and Wildlife Service 2006). While Mississippi has the third largest population of white-tailed deer (Woods 2006), estimated to be around 1.75 million in number, the state saw the largest decrease (-6.8%) in hunting license sales between 2007 and 2008 (Northway 2010).

The results and conclusions of previous studies are mixed regarding which factors play the most crucial role in DVC location and occurrence. Most agree that DVCs do not have a random spatial distribution (Seiler 2005, Gonser and Horn 2007, Gonser et al. 2009), but instead are associated with roads in close proximity to forested areas (Farrell and Tappe 2006). Less conclusive are findings that link vehicle speed and traffic volume to DVCs (Hubbard et al. 2000). Farrell and Tappe (2006) showed that the distribution of DVCs in Ohio was negatively related to cropland, whereas Gonser and Horn (2009) found the opposite in Indiana. This apparent contradiction in the role of farmland in DVC distribution is likely due to variations of deer grazing behavior between these two states.

The growing presence of deer in human environments and increases in DVCs in recent years has become a significant problem for wildlife and transportation managers in the U.S. (Butfiloski et al. 1997). DVC kill rates are not high enough to adversely affect the deer population, but there is a growing need to understand the geography of this phenomenon to minimize economic losses, improve traffic safety, prevent human injury, and decrease the number of deer injury and deaths from automobiles (Bissonette et al. 2008, Putnam 1997, Seiler 2005). As such, a great deal of research on DVCs remains to be done (Hubbard et al. 2000).

1.3. Significance of This Research

Over 1.5 million DVCs occurred in the United States in 2002 alone. These collisions were responsible for over $1 billion dollars in damages, and the deaths of 150 people and roughly 1.5 million white-tailed deer (Gonser and Horn 2007, Gonser et al. 2009, State Farm Insurance 2006). In 2009, State Farm Insurance used accident claims to estimate that 1,160,979 DVCs occurred in the United States that year, 14,327 of which were in Mississippi (State Farm Insurance 2010). DVCs increased by 38 percent in Mississippi between 2007 and 2009, ranking the state among the top five in the nation with the largest increase in DVCs (Northway 2009, State Farm 2009). To illustrate the need for DVC mitigation in Mississippi as well as the U.S., I collected national-level data on DVCs for 2009 (State Farm, 2010). After normalizing these data for population, registered vehicles, and land area, I created three choropleth maps classified into quintiles. With 191 DVCs per capita, Mississippi ranks 24th in the nation (Figure 2). When DVCs are normalized for land area, Mississippi ranks 26th (Figure 3). When normalized for registered vehicles, however, Mississippi ranks 13th with one incident per 141 automobiles (Figure 4).

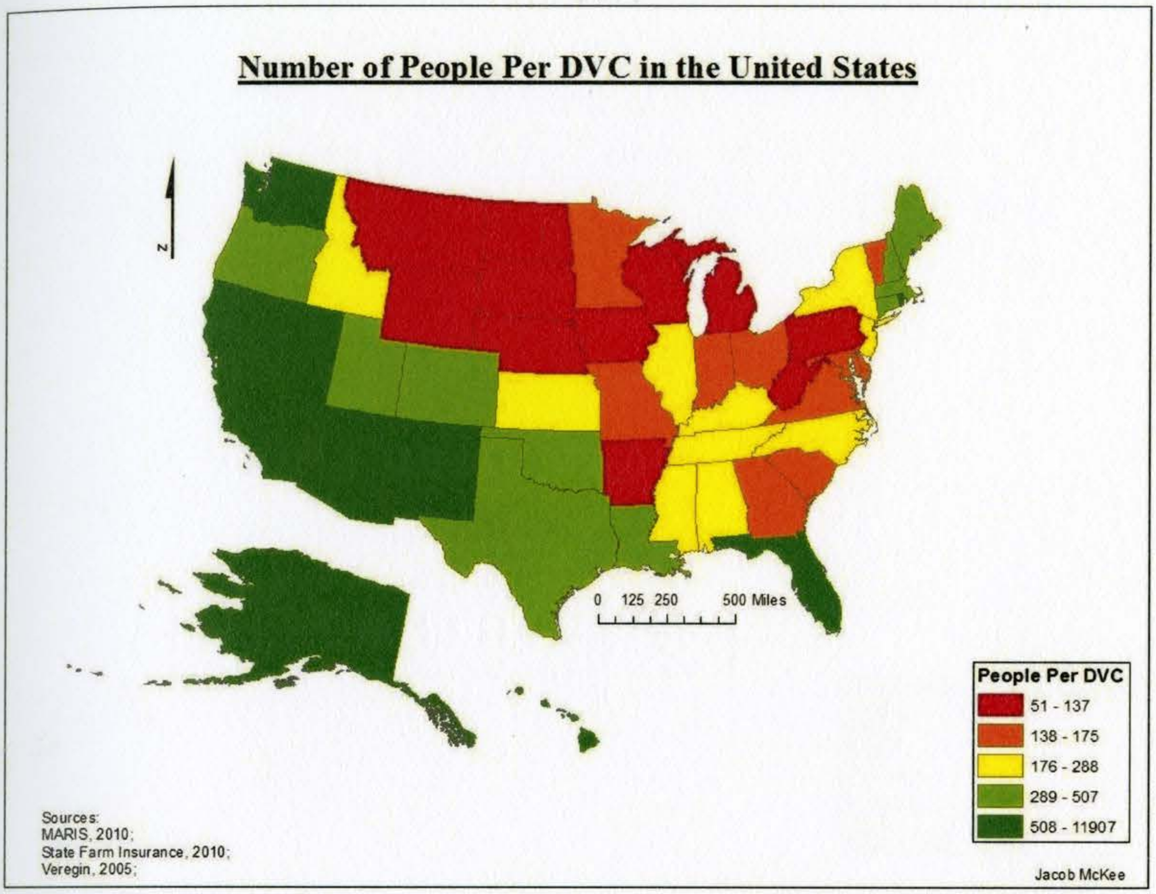

Figure 2. Deer-Vehicle Collisions, 2009 (Normalized for Population).

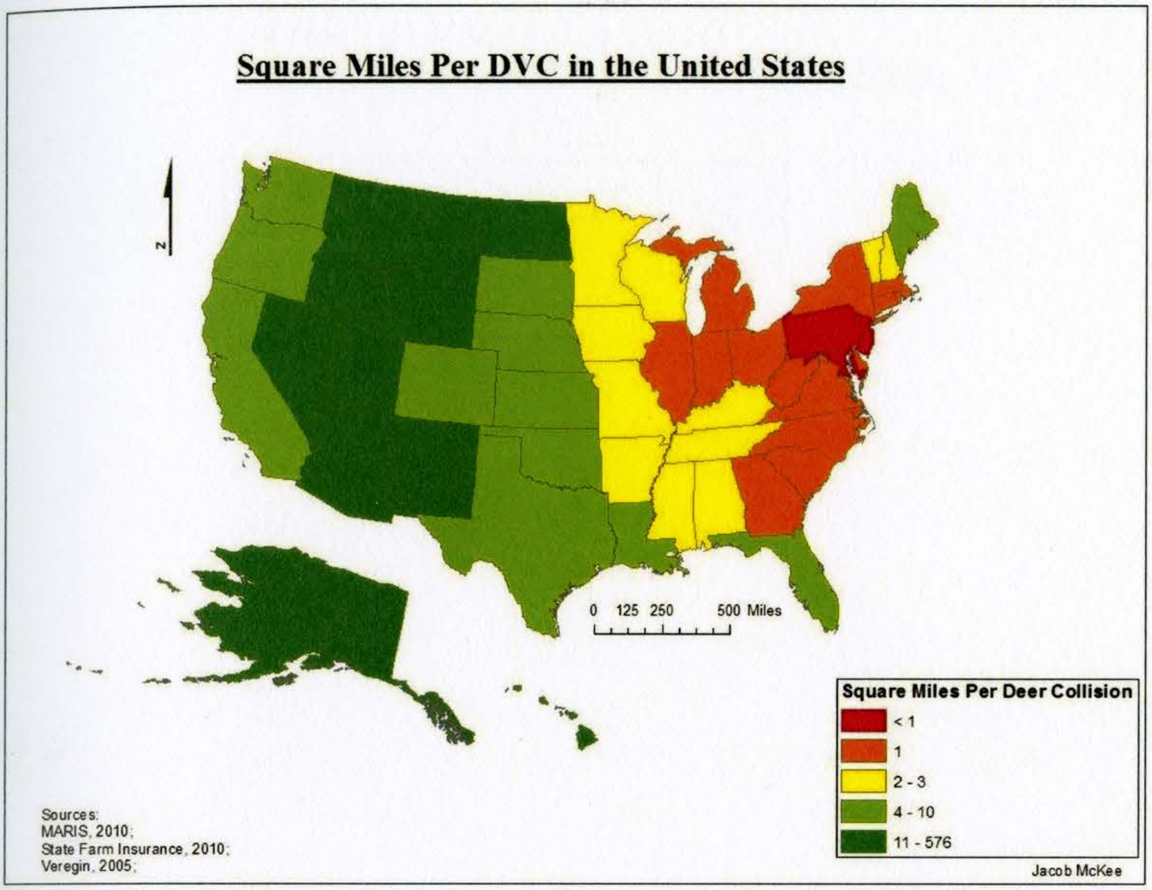

Figure 3. Deer-Vehicle Collisions, 2009 (Normalized for Land Area).

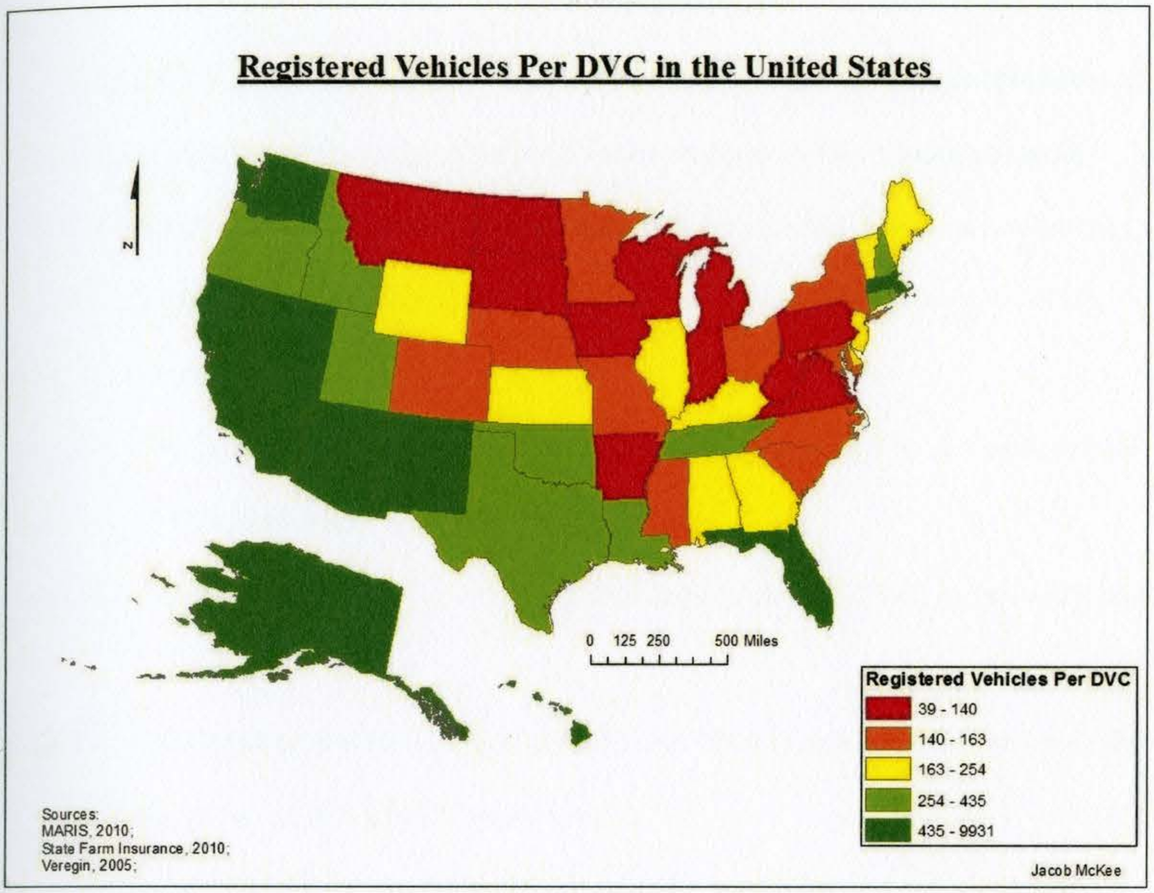

Figure 4. Deer-Vehicle Collisions, 2009 (Normalized for registered vehicles).

Estimated monetary losses associated with DVCs in 2004 ranged from $796 per vehicle in Savannah, Georgia (Butfiloski et al. 1997) to $1,500 nationwide (Malo et al. 2004). Unfortunately for motorists, not only have DVCs increased in recent decades but property damage associated with these accidents has as well, reaching a nationwide average of $3,050 per collision in 2009 (State Farm Insurance 2009). In Mississippi, DVC-associated property damage averages around $4,500 per collision and is estimated to account for 35 percent of all repair claims at auto body repair shops in the state (Woods 2006).

1.4. Research Objectives

To gain a better understanding of the factors that influence spatial distributions of DVCs, this thesis integrates geographic information systems (GIS), remotely sensed imagery, FRAGSTATS and inferential statistics to analyze DVCs in Forrest and Lamar Counties of South Mississippi between 2006 and 2009. The primary objectives of this research include:

1. Examining and explaining temporal trends (hourly, monthly, and seasonal) of DVCs in the study area.
2. Determining whether or not the spatial distributions of DVCs in the study area are random.
3. Determining whether a specific land-cover class is closely associated with the locations of DVCs in the study area.
4. Developing a statistical model to determine landscape patterns and features that influence the spatial distribution of DVCs.

1.5. Summary

The number of deer killed by vehicle collisions exceeds the number of deer killed through hunting each year in the United States (Coffin 2007). Published research has shown that DVCs have a significant impact upon society, the economy, and conservation initiatives (Conover 1997, Seiler 2005, Hussain et al. 2007). Research has also shown that these collisions are not random occurrences, but products of various environmental characteristics that can be quantitatively measured through remotely sensed imagery and geographic information systems (Seiler 2005). Furthermore, little research has been published on the factors that influence DVCs in the southeastern U.S. After analyzing the national data pertaining to DVCs in 2009, it is apparent that Mississippi is among the states where DVCs should be a top priority for mitigation. However, in order to do so, it is essential to gain a better understanding of the factors that influence the distribution of DVCs. Through the application of GIS, remotely sensed imagery, and statistical analysis, this thesis will identify what landscape features are most important in explaining the spatial distribution of DVCs in Forrest and Lamar Counties of South Mississippi.

CHAPTER II

A REVIEW OF RELATED LITERATURE

2.1. Human-Wildlife Conflicts

Interactions between humans and wildlife historically have occurred in rural areas, but recently have become increasingly common along urban fringes (Manfredo and Dayer 2004). Wildlife impacts human society in both positive and negative ways. Negative impacts, collectively referred to as human-wildlife conflicts, can arise from a variety of interactions in which wildlife negatively impact humans and/or their goals or when the actions of humans negatively impact wildlife and/or their resources (Conover et al. 1995, Messmer 2000, Madden 2004). Human-wildlife conflicts, however, are not easily categorized simply into two groups, because negative consequences can impact both parties and conflicts are not limited to single species (Messmer 2000, Manfredo and Dayer 2004). As wildlife populations have increased in the 20th century, conflicts also have increased as the needs of humans and wildlife further overlap, creating costs for both. As human activities intensify and encroach upon natural environments, wildlife will continue to impact economic development and human livelihood (Messmer 2000, Madden 2004).

Traditionally, wildlife management has been defined as initiatives that strive to maintain or increase populations of various wild species. More recently, in the 1980s and 1990s, as human-wildlife conflicts became a public concern, wildlife managers shifted their focus to human-wildlife conflict management. Since its origins, this field has integrated various techniques and strategies to mitigate negative interactions between humans and wildlife (Messmer 2000). In recent decades, organizations such as the

National Wildlife Research Center (NWRC) have expanded to help mitigate human-wildlife conflicts and correct misconceptions that humans have towards wildlife (USDA 2010). Numerous strategies have been developed to mitigate human-wildlife conflicts, but most are specific to particular regions or species (Madden 2004). The following paragraphs discuss published research on various conflicts that have clear and negative impacts for humans, wildlife, or both.

The most common human-wildlife conflicts with negative impacts for humans include pest damage of agricultural crops, livestock depredation, property damage, and human injury or fatality. A survey of U.S. agricultural producers showed that 89 percent of respondents experienced problems with wildlife, half of which reported losses of more than $500 due to crop damage (DeVault 2007, Messmer 2000). Wildlife-induced crop damage is often the result of invasive or exotic species such as feral hogs (*Sus scrofa*), which were introduced to North America by European colonists and have since become a major nuisance to farmers due to the damage they cause to agricultural fields. In 2003, feral hogs inhabiting the Savannas Preserve State Park of Florida inflicted severe damage to a 9,027 square-meter patch of land, creating furrows as deep as forty-five cm and removing virtually all vegetation cover. Feral hogs also created problems by degrading habitat and outcompeting native species. It was not until a removal program led to the capture of twenty-three hogs that the state park was able to recover (Engeman et al. 2007).

Native species such as beavers (*Castor canadensis*) also inflict excessive damage to agricultural production and to the timber industry. In the southeastern U.S., beaver impoundments routinely flood over 288,000 hectares of forest (Conover et al. 1995).

Impoundments not only destroy valuable stands of timber, but can also flood roadways and agricultural fields. Across the American Southeast, beavers annually cause an estimated $100 million in damaged timber stands. The NWRC recently has implemented several management strategies in Mississippi to reduce economic loss from beavers. For every dollar the NWRC spends on mitigation, the state saves approximately $39.4 to $88.5, making management and control economically viable (USDA 2010).

White-tailed deer (Odocoileus virginianus) and raccoons (*Procyon lotor*) cause damage to soybean and corn crops on a regular basis in the American Midwest. During late summer and fall, corn and soybeans can constitute up to 65% of these species' diet (DeVault et al. 2007), resulting in damage that can significantly reduce profit margins of farmers and increase costs for consumers (Messmer 2000). Regulated hunting of game species such as white-tailed deer can play an important role in the mitigation of this type of adverse impact (DeVault et al. 2007).

Human-wildlife conflicts of greatest public concern cause human injury, illness, or death, and are often the result of wildlife attacks or disease transmission. Two of the most common diseases with multiple wildlife vectors include rabies and Lyme disease (Conovor et al. 1995, USDA 2010). Lyme disease typically makes up 81 percent of all reported wildlife-related diseases (Conover et al. 1995). Annual public health costs associated with rabies testing, treatment, and vaccinations has risen to $300 million in the U.S. in recent years (USDA 2010). Turkey vultures (*Cathartes aura*) are perceived as a common vector for disease transmission to humans and are most commonly associated with unsanitary accumulations of fecal droppings. Although turkey vultures may also cause superficial damage to roosting areas such as rooftops, their feces and regurgitations tend to have a repugnant, ammonia odor, increasing concerns pertaining to sanitation and disease transmission (Ball 2009, USDA 2010). In several studies in North Carolina, the implementation of vulture effigies were effective in removing vulture roosts, resulting in resolution of this human-wildlife conflict with little or no negative impact on the animal population itself (Ball 2009).

Conversely, human-wildlife conflicts with adverse impacts on wildlife are most commonly associated with wildlife injury, death, and habitat destruction. In northeastern India, cultivation and developmental activities have destroyed and further encroached on the natural habitat of many species. As forest cover has disappeared, conflicts between humans and elephants (*Elephas maximus*) have increased dramatically. Between 1980 and 2003, more than 370 elephants were killed in retaliation for crop trampling. Given northeast India is home to approximately 25 percent of the world's elephant population, this conflict represents a conservation dilemma of global importance. As elephants and humans compete for space, conflicts are inevitable and elephants will continue to suffer from habitat destruction or retribution. A wide range of strategies has been designed to mitigate this conflict. The most common involve allocating protected areas, preventing urban encroachment, and banning commercial logging (Choudhury 2004). Similarly, in certain areas of the U.S., the proliferation of wind farms has resulted in the disruption of migratory birds, such as the northern pintail (*Anas acuta*) and birds of prey, such as the golden eagle (*Aquila chrysaetos*). The California Energy Commission found that over 1,000 birds of prey were killed at one wind farm located near San Francisco. Since these conflicts directly affect conservation, numerous game and fish departments now impose guidelines that require potential wind farms to examine migratory patterns and to include development of strategies to monitor turbine related mortality on wildlife (Martin 2010).

Human perceptions of wild animals are an important factor in human-wildlife conflicts. During the 1960s, declines in black bear (*Ursus americanus*) populations in Louisiana and the listing of the species as threatened led to a reintroduction program that brought about a recovery of the species, but also resulted in conflicts when bears began to appear in residential areas scavenging for food. Although no bear attacks occurred, many residents supported extermination or removal, despite the fact that these methods were in conflict with conservation goals. Wildlife managers found the best way to resolve these conflicts was through outreach to address public fears and increase awareness of black bear behavior (Cotton 2008). Just as black bears are perceived as threats or nuisances in Louisiana, colobus monkeys (*Procolobus kirkii*) receive similar attention in Zanzibar, Tanzania. A highly endangered species, the colobus monkey primarily sustains itself by eating immature coconuts. In the mid-1990s, coconuts made up 90 percent of foreign exchange in Zanzibar. Because colobus monkeys routinely raided coconut plantations, residents felt they jeopardized their economic wellbeing and called for their removal. Researchers, however, were able to determine that the monkeys had only a limited impact on agricultural production and mitigation was redirected towards community education and conservation (Siex and Stuhsaker 1999).

Human-wildlife conflicts that result in negative outcomes for both humans and wildlife are most commonly associated with transportation collisions. Aircraft collisions with birds pose a serious threat to economics, safety, and conservation (Dolbeer et al. 2000, USDA 2010). According to the Federal Aviation Administration (FAA), there were 7,516 reported aviation bird strikes in 2008 (USDA 2010). Larger birds such as geese and vultures pose the greatest hazard, often causing aircraft to crash (Dolbeer et al. 2000). Between 1990 and 2008 wildlife collisions with aircraft cost the civil aviation industry approximately $614 million annually in the U.S. The NWRC is now studying the possibility of habitat modification and development of lighting systems to deter birds from living in the vicinity of airports (USDA 2010). Similarly, in Norway, collisions between moose (*Alces alces*) and trains are common events, occurring at a rate of 1,000 per year since the early 1990s. Researchers found that moose commonly use railroads as corridors and grazing land because they remain clear of heavy snow during the winter. Various mitigation strategies have been implemented, the most successful of which were scent marking and supplemental feeding. Scent marking involves spraying chemicals along rail lines that replicate the scents of bears or humans. Supplemental feeding areas located away from rail corridors offer safer feeding opportunities for moose. Research showed the scent-marking reduced the number of moose-train collisions by 85 percent and supplemental feeding proved a beneficial, although expensive, mitigation strategy as well (Andreassen et al. 2005). Similarly, kangaroo-vehicle collisions (*Macropus sp.*) are a common problem in Australia. During periods of drought, kangaroo mortality along roadways increases dramatically. Research found that in times of drought, irrigated pastures along roadways attracted foraging kangaroos. Establishment of supplemental feeding areas was successful in discouraging kangaroos and other wildlife species from areas along roadways (Lee et al. 2004).

From this review of published studies, it is apparent that a significant number of human-wildlife conflicts occur along roadways. Road systems not only encroach upon wildlife habitat, but provide environments in which automobiles and wildlife come in contact with each other, resulting in disastrous consequences (Litvaitis and Tash 2008).

2.2 Ungulate-Vehicle Collisions

Given the prevalence of roads in almost everywhere in the United States, it seems difficult to escape their effects (Litvaitis and Tash 2008). One consequence of growing road networks is an increase in the number of wildlife-vehicle collisions. Collisions between automobiles and ungulates have increased dramatically in Europe and North America in recent decades. For the purpose of this study, ungulate-vehicle-collisions (UVC) refer to ungulates in general, while DVC and MVC refer to deer and moose-vehicle collisions respectively. In Sweden, for example, UVC accounted for 60 percent of road accidents in the 1990s (Seiler 2004). These collisions are considered a major road-safety hazard and economic drain in Europe and North America (Groot-Bruinderink and Hazebroek 1996, Bissonette et al. 2008). In Europe and Canada the most prevalent UVC involves moose, while white-tailed deer are most common in the U.S. (Coffin 2007). Despite the wide variety of situations and species associated with UVCs, the common denominator of all wildlife-vehicle collisions is the involvement of humans. Because humans are responsible for the consequences of collisions with wildlife, finding ways to understand, mitigate, and prevent UVCs remains an important area of scientific study (Groot-Bruinderink and Hazebroek 1996, Manfredo and Dayer 2004). Research focuses on two major themes: social perceptions of UVCs and the identification of the factors that drive them for the purpose of modeling and prediction (Livaitis and Tash 2008). The following paragraphs discuss recent literature in these two areas of research in North America and Europe.

Some studies review the numbers of UVCs and corresponding economic losses, seeking to determine if there is sufficient evidence for mitigating these incidents, whereas other studies rely on identifying and influencing social perception. Humans must cope with the economic consequences of UVCs so it is logical to study social perceptions to determine if there is sufficient public interest in their mitigation. Annual costs associated with deer-vehicle collisions exceed $1 billion each year, suggesting that UVC mitigation could prevent economic loss. From a social perspective, however, the supposed benefits of reducing UVCs might not be worth the reduction of local or regional deer populations (Schwabe and Schuhmann 2002). At least one study has shown that societal attitudes of ungulates are inversely related to the amount of damage individuals have experienced. With this in mind, reducing DVCs should improve social perceptions toward ungulates, although this is not always the case (McShea et al. 2008).

Bissonette et al. (2008) assessed the costs of DVCs in Utah and found that human injuries, deaths, vehicle damage, and loss of wildlife are the greatest concern surrounding ungulate vehicle collisions that require attention and justify mitigation. Such mitigation can be costly, but this study found that 58% of DVCs were concentrated on only 11% of state roadways, making policy solutions both practical and cost effective. In southern Michigan, on the other hand, Marcoux and Riley (2010) studied the perceptions and attitudes of drivers toward DVCs. They found that 88% of survey respondents answered positively about seeing deer on their commute, while 94% simultaneously worried about potential DVCs. Marcoux and Riley found that while those involved in DVCs were less likely to view deer positively, almost 48% wanted the local deer population to remain the same and 8% wanted an increase. By contrast, only 38% of those involved in DVCs wanted a decrease in deer herd size. In a state with more than 60,000 DVCs per year, this study showed public preference of deer herd size in Michigan was not significantly hampered by concern about DVCs. Similarly, in Tompkins County, New York, Stout et al. (1993) found that 36% of respondents enjoyed the presence of deer without worrying about DVCs, while 54% enjoyed deer presence but were worried about the possibility of a DVC. Furthermore, 63% of respondents felt that the chance of personal involvement in a DVC within the next year was low. Most respondents regarded DVCs as dangerous and expensive accidents, but 49% preferred to maintain the deer population and 14% desired an increase. Butfiloski et al. (1997) conducted a study of a small community in Georgia experiencing numerous problems associated with the overabundance of deer. As the number of DVCs began to rise in the community, public opinion began to shift toward a reduction in herd size. The community unanimously approved a resolution to reduce the deer herd by removing or euthanizing 1,127 deer, providing an affordable and effective measure to reduce DVCs. Understanding public perceptions of risk associated with UVCs provides wildlife managers an understanding of local preferences for the size of deer populations, thus preventing further economic loss due to unwanted mitigation or management (Stout et al. 1993).

While social perceptions provide insight into public preference towards mitigation and wildlife management, other areas of study rely on environmental characteristics to determine where and when to implement mitigation strategies. Recent research has sought to better understand causal factors of UVCs by developing statistical models, thus making it possible to predict specific locations of potential threat. Some of these studies focus on highway characteristics such as traffic speed, traffic volume, road density, and road curvature, whereas others focus on local environmental factors, such as landscape structure and composition. The findings of these two bodies of research are inconsistent and often contradict each other, highlighting the site and regional specificities of UVCs (Madden 2004). The following paragraphs will discuss these two areas of research on UVCs in North America and Europe, further elucidating the apparent contradictions and gap in the published literature.

The results of numerous studies suggest that decreases in traffic speed and volume leads to a decline in the number of collisions, but other studies have found the opposite to be true. Bissonette and Kassar (2008), using multiple regression analysis to compare roads in Utah based on traffic volume and posted speed limit, failed to find a significant relationship between DVCs and road characteristics. Similarly, in a study of wildlife-vehicle collisions in New Hampshire, Litvaitis and Tash (2008) created a multivariate model using traffic volume, but could explain 55% of variation. Bashore et al. (1985), using highway characteristics to predict DVCs in Pennsylvania, found a significant negative relationship between traffic speed and DVCs, demonstrating that as speed limits increased, the probability of DVCs decreased.

Whether or not highway characteristics influence locations and occurrences, it is apparent that a wide variety of factors influence the frequency and distribution of UVCs, suggesting that the reason animals are hit by vehicles are related to the spatial arrangement of resources, thus animals die when they are searching for these resources (Coffin, 2007). It is for this reason that recent research has taken an explicitly geographical approach to better understand spatial distributions and patterns of UVCs.

Many of these studies integrate landscape ecology and geography in an effort to more accurately predict these UVCs.

Speed limit seems like an important factor in UVCs given that a vehicle moving at 120 km/hr (75 mph) has a slower reaction time than a vehicle traveling at 50 km/hr (30 mph). Although some studies that focus on the influence of highway characteristics have found that traffic speed and volume are important causal factors of UVCs, other studies have found no such relationship. Studying moose-vehicle collisions (MVCs) in Sweden, for example, Seiler (2004) concluded that increased traffic volume explained 85% of the variance in a regression analysis model and that MVCs increased on roads with speed limits over 90 km/hr (55 mph). Although the regression model used by Seiler had only a small number of variables, it correctly classified 81.2% of all collisions that occurred during the study period. Similarly, Ng et al. (2008) used a regression model to determine that DVCs in Minnesota were twice as likely to occur on roads with higher speed limits and lower road densities. Similar results were found in Virginia where highway characteristics of primary roads failed to explain DVCs, but traffic volume on secondary roads was correlated with the probability of a DVC (McShea et al. 2008). Furthermore, Gunson et al. (2004) found that increased traffic volume was the primary reason for greater numbers of UVCs in the Canadian Rocky Mountains.

Landscape ecology integrates both geography and ecology by focusing on how landscape structure and composition affects the abundance and distribution of organisms across geographical space (Kent 2007). It studies the interactions between spatial pattern and ecological process, based on the premise that spatial patterns significantly influence ecological processes (Gustafson 1998, Turner et al. 2001, Turner, 2005). In the context of understanding wildlife-vehicle collisions, landscape is essentially the surrounding area that is spatially heterogeneous in at least one factor of interest, most commonly land-use or land-cover (Turner 2005). Recently, many federal and state wildlife management agencies have adopted landscape ecology approaches to assess wildlife-vehicle collisions to develop location and species-specific management strategies. Due to the influence of land-use activities and disturbance patterns, a great deal of spatial variation exists in the distribution and quality of wildlife. It is therefore imperative to understand and identify the distribution and availability of habitat within a landscape to understand and recognize patterns of UVCs and effectively mitigate them (Felix et al. 2007). As noted earlier, UVCs in North America and Europe are site and species specific, which suggests that regional-scale research is necessary to understand the factors that govern them. The following paragraphs provide a survey of various spatial, predictive models that have been developed in recent years to analyze UVCs.

One of the earliest published studies that integrated landscape ecology, highway characteristics, and predictive modeling was conducted in Pennsylvania by Bashore et al. (1985), who found that as the number of residences and commercial buildings increased, the likelihood of DVCs decreased. This finding is supported by other studies that focus on space usage of white-tailed deer and demonstrate that this species tends to avoid dwellings and built environments (Storm et al. 2007). Bashore et al. (1985) also found that as landscapes become less wooded, the chances of DVCs increases, indicating that deer more commonly used grass and underbrush habitats for browsing. The regression model used to determine this finding was particularly robust, correctly classifying 85% of DVC sites and 89% of control locations. Similarly, Gonser et al. (2009) studied DVCs in

Indiana and found that planted or cultivated land was the land-cover type most closely associated with DVC sites. Nielsen et al. (2003) corroborated these findings in their study of DVCs in Minnesota. Their model only had an accuracy of 77%, but they found that forest cover did not significantly contribute to the location of DVCs and that landscape diversity and the amount of public land was higher surrounding the collision site. Besides finding that traffic volume played a role in DVCs in Virginia, McShea et al. (2008) also found that the highest rates of DVCs occurred in agricultural regions. In Alberta, Canada, DVCs were most common around highly productive non-forest vegetation and were 29% more likely to occur near water (Ng et al. 2008).

In contrast, Malo et al. (2004) found that MVCs in Spain most commonly occurred in areas of high landscape diversity and non-riparian forest. Furthermore, they were able to correctly classify 87% of low risk areas by identifying areas dominated by agriculture and urban land-cover. Similarly, in Alabama, Hussain et al. (2007) found that an increase in the proportion of cropland reduced the probability of a DVC. Hubbard et al. (2000) found that large agricultural fields decreased the chance of a DVC occurring in an area. With a classification accuracy of 63.3%, their model showed that as the amount of woody patches, bridges, and lanes of traffic increased, so did the probability of a DVC. These findings were similar to those in Arkansas, which showed that DVCs were influenced most by urbanization and human population densities (Farrell and Tappe 2007). Other studies have demonstrated that there are numerous causal factors of UVCs. In Sweden, Seiler (2005) developed a model that utilized highway characteristics and a second one based on landscape variables. With a classification accuracy of 62%, Seiler (2005) found that MVCs occurred more frequently in areas where there was no highway fencing and on roads that traversed clear-cuts or young forests. Around Glacier National Park in Canada, MVCs were found to most likely occur in areas that were closer to water and wetlands, which are used by moose to forage for herbaceous and woody plants (Hurley et al. 2009). In the same way, MVCs in Ontario, Canada routinely occur around salt pools that accumulate after snowmelt. Salt licks have proven to be influential in MVCs due to their attractive resources for wildlife that have struggled to find natural mineral licks during harsh winters (Groot-Bruinderink and Hazebroek 1996).

Much of the research on spatial distributions of DVCs has been conducted in the northeastern or midwestern regions of the United States. To date, there have been only three published DVC studies conducted in the American South, one being a statewide study in Alabama (Hussain et al. 2007), another in Clarke County, Virginia (McShea et al. 2008), and a third that focused on DVCs in Arkansas (Farrell and Tappe 2007). Even fewer studies have been conducted at the county-level, with most of these focused on the upper Midwest. The only published county-level study of DVCs in the American South is that of Farrell and Tappe (2007), which was conducted in Arkansas. Owing to the lack of published literature regarding local factors influencing deer collisions in the southern U.S., there is a clear gap in the DVC literature that needs to be filled. Furthermore, a review of available literature confirms that factors influencing the location of DVCs are inconsistent, varying with correlative factors highly site specific (Ng et al. 2008). This finding further supports the need for county-level studies of DVCs and associated factors in the southern U.S.

2.3. Mitigation

The financial cost of UVCs suggests that effective mitigation strategies have the potential to prevent significant economic loss associated with these conflicts. While such an argument is relatively straightforward, what seems unclear is which strategies are most effective (Schwabe and Schuhmann 2002). Predictive models seem to be effective at identifying areas of high UVC incidence thereby potentially increasing the success rate of mitigation (Malo et al. 2004, Litvaitis and Tash 2008, Ng et al., 2008). Currently, there are over forty different mitigation policies in place designed to reduce automobile collisions with ungulates (Huijser et al. 2009), most of which are documented in publications by various state and federal agencies (Romin and Bissonette 1996, Danielson and Hubbard 1998). Within this literature, there are three types of mitigation strategies: modifying motorist behavior, modifying ungulate behavior, and reducing ungulate populations (Mastro et al. 2008).

Mitigation policies designed to modify motorist behavior include reducing speed limits, erecting warning signs, modifying roadside vegetation, and public education. Some studies have shown that speed limit is a factor in UVCs, but recent research suggest that reducing travel speeds is not necessarily an effective form of mitigation (Curtis and Hedlund 2005). In fact, some studies have found that reducing speed limits has actually led to an increase in collisions (Mastro et al. 2008). Furthermore, unless speed limits are adhered to through enforcement, they do not necessarily lead to reduction in travel speeds (Curtis and Hedlund 2005).

Deer-crossing warning signs are the most common mitigation strategy in the U.S. (Romin and Bissonette 1996, Putnam 1997, Danielson and Hubbard 1998). Various types of road signs help warn motorists of specific locations where DVCs are common (Curtis and Hedlund 2005). Caution signs are the familiar yellow diamond with a black silhouette of a deer (Mastro et al. 2008). Although these signs are the most frequent, it is doubtful whether deer caution signs are effective in the long term. Often motorists become complacent with signs unless they are reinforced through an actual encounter (Putnam 1997, McShea et al. 2008). Enhanced signs have been used to increase the effectiveness of warnings by making them more visible with lights and flags. These can be replaced by electronic message signs permanently positioned above the roadway or on a portable trailer that flash warning messages alerting drivers to potential danger. Animal activated warning signs detect ungulates along the road through the use of infrared light, radar, laser, or thermal sensors and then flash warning lights to caution passing motorists (Curtis and Hedlund, 2005, Mastro et al. 2008).

Increasing driver visibility by clearing roadside vegetation has also been shown to decrease UVCs. At least one study has shown that clearing a twenty-meter zone on each side of a road decreased UVCs by 20%, however, maintenance costs and the potential ecological impacts make this technique problematic (Mastro et al. 2008). Approximately half of all U. S. states use public and private funds to implement education programs to inform motorists about the danger of DVCs and how to avoid them (Donaldson 2006, Mastro et al. 2008). Through press releases, television commercials, brochures, and web pages, government agencies and private organizations attempt to warn motorists about the seasonality of DVCs, reiterating that DVCs are not random and suggesting measures to avoid collisions (Donaldson 2006).

Mitigation practices designed to modify ungulate behavior include overpasses and underpasses, repellents, vegetation modification, and fencing. These techniques focus on preventing ungulates from crossing roadways through physical control or by creating other areas more attractive than road corridors (Curtis and Hedlund 2005). Wildlife underpasses and overpasses can either be designed specifically for wildlife or consist of infrastructure modified to encourage use by animals. These crossings enhance habitat connectivity and offer safe corridors for wildlife to safely cross roadways, thus reducing effects of habitat fragmentation (Donaldson 2006). Recently, Michael Van Valkenburgh and Associates won a nationwide competition to design and construct a wildlife overpass near Vail, Colorado. Their proposed ecological bridge is expected to be fifty meters wide and cultivated with native trees, shrubs, and grasses to encourage wildlife (Wald 2011).

Common repellents used to deter ungulates include a variety of hazing techniques such as reflectors or chemical repellents. Reflectors are devices attached to posts along roadways that capture light from oncoming traffic and transmit it as a continuous barrier of white, red, or blue-green light parallel to the road (Putnam 1997, Curtis and Hedlund 2005, Mastro et al. 2008). Such devices are limited, however, because they are only effective at night and various studies have shown that deer cannot see red light and are not frightened by blue-green light (Putnam 1997, Mastro et al. 2008). Furthermore, deer eventually become adjusted to the light, rendering reflectors ineffective (Putnam 1997). Research in British Columbia and Germany has used chemical repellants that mimic the odors of bears and humans, but their effectiveness in preventing DVCs has not yet been adequately evaluated (Mastro et al. 2008).

Vegetation on road corridors grows rapidly with ample light and moisture from road drainage (Forman and Alexander 1998). As such, it is not surprising that a great deal of research has been conducted on the effects of roadside vegetation modification on the frequency of DVCs. Cultivating unpalatable plants along roadways might possibly decrease the number of ungulates foraging in these areas (Rea 2003, Donaldson 2006, Mastro et al. 2008).

Fencing, when combined with warning signs, has shown to be the most effective way to reduce UVCs (Curtis and Hedlund 2005, Donaldson 2006). Several studies have found that to prevent deer from entering roadways, fencing must be at least 2.4 meters high (Curtis and Hedlund 2005). Furthermore, one-way gates allow for the safe exit of ungulates trapped on roadways (Putnam 1997). Research in Banff National Park, near Alberta, Canada found that after construction of fencing, UVCs decreased by 80% (Clevenger and Waltho 2000, Clevenger et al. 2001).

Perceptions of risk associated with UVCs can influence public preference toward the size of ungulate populations, thus supporting various management objectives that attempt to reduce population numbers (Stout et al. 1993). Several studies have shown that the implementation of ungulate relocation programs have reduced the number of UVCs, but due to high costs, risk of disease transmission, unavailable release sites, and concerns over animal cruelty, live-trapping and relocation of ungulates is not an option in most states (DeNicola and Williams 2008, Mastro et al. 2008). State transportation departments rated the most effective UCV control strategy to be culling large herds of wild grazers (Curtis and Hedlund 2005). In small towns in Iowa, New Jersey, and Ohio, sharpshooters eliminated 950 deer in Iowa, 1,455 in New Jersey, and 1,002 in Ohio.

Following this management strategy, DVCs declined by rates ranging from 49-78%, with the greatest decline in Iowa (DeNicola and Williams 2008). Similarly, in a residential community near Savannah, Georgia, killing 1,127 deer proved to be a cost effective strategy to reduce DVCs (Butfiloski et al. 1997). In Lynchburg, Virginia, wildlife specialists culled 2,600 deer over a ten-year period, decreasing DVCs by 50% (Donaldson 2006). More recently, a bill was introduced into the Mississippi Legislature that proposes to cull the state's large deer population in attempt to curb DVCs (Northway 2010). Despite its apparent effectiveness, culling deer populations remains politically controversial (Conover 1997, Storm et al. 2007, Mastro et al. 2008).

Analysis of the economic costs of mitigation and economic losses associated with UVCs is complex (Putnam 1997). With the number of UVCs increasing each year, there is a growing need for more research on mitigation strategies. Although fencing increases the amount of habitat fragmentation, it appears to be most effective at reducing UVCs when used with deer warning signs. Ultimately, however, due to the site and regional specificity of UVCs, there is likely no universally appropriate strategy to mitigate this environmental hazard for both humans and wildlife (Putnam 1997, Curtis and Hedlund 2005, Donaldson 2006, Mastro et al. 2008).

CHAPTER III

METHODOLOGY

3.1. Study Area

The study area for this thesis research encompasses two counties in south-central Mississippi: Forrest and Lamar (Figure 5). Both are situated in the Level IV Southern Pine Plains and Hills ecoregion of Mississippi, as defined by Environmental Protection Agency (Chapman et al. 2004). Ecoregions are areas with homogenous soils, vegetation, physiography, or other environmental characteristics (Griffith et al. 2003). The most important unifying environmental characteristic of the two counties of the study area is the dominance of Longleaf and other pine species. Forrest County, which forms the eastern half of the study area, has a total land area of 467 square miles, 1,038 linear miles of road, and a density of 2.2 miles of road per square mile of land area. Lamar County, which borders Forrest to the west, has a total land area of 497 square miles with 887 linear miles of road and 1.8 miles of road per square mile of land area (U.S. Census 2000). These two counties were chosen because they are located in the same ecoregion, are adjacent to each other, and are accessible from The University of Southern Mississippi. Mississippi is an ideal study area not only because of the dearth of DVC studies in the state, but also because there are no current mitigation strategies directed towards deer collisions other than common deer crossing signs, unlike numerous other states, which have implemented highway underpasses, fencing, and habitat alteration as mitigation strategies (Romin and Bissonette 1996).

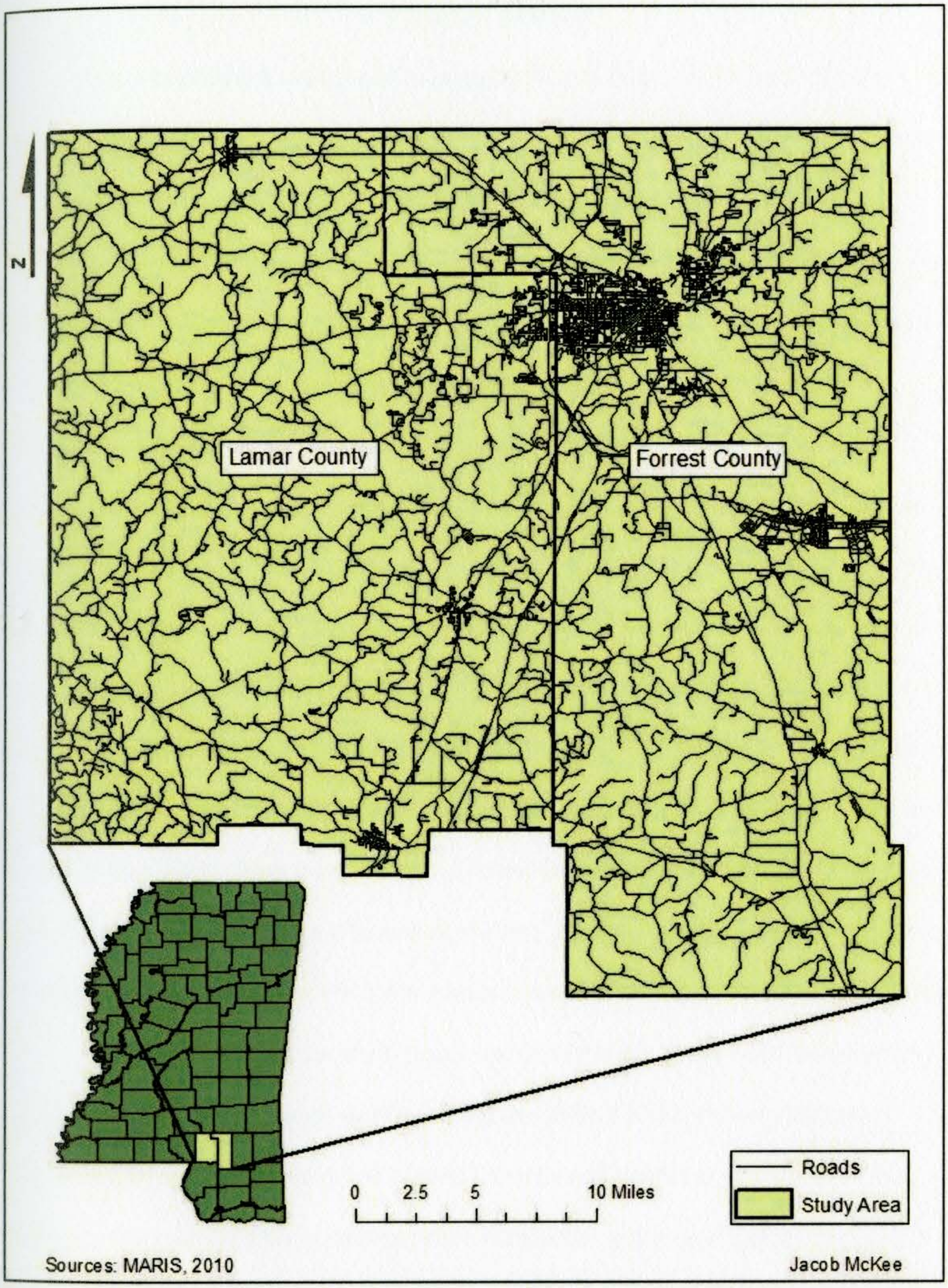

Figure 5. Study Area in Forrest County and Lamar County, MS.

3.2. Data Acquisition

Data was obtained from the Mississippi Office of Highway Safety (MOHS) on the locations of DVCs in Forrest and Lamar Counties between 2006 and 2009. These data were pulled from a larger database of vehicle collision reports maintained by MOHS. Deer-vehicle collisions are coded as a type "31" accident, which is derived from official collision reports completed by responding public safety officers. Code "31" refers only to collisions involving deer and does not include collisions resulting from a driver swerving to miss a deer. In addition to collision information, these data contain the geographic coordinates of DVC sites in decimal degrees with an estimated accuracy of seven meters. The responding public safety officer recorded these measurements at the scene of each collision with the aid of a handheld GPS device. As well as locational information, the data set also includes date, time of day, and weather conditions at the time of the collision (Sennett 2010). Between 2006 and 2009, 347 type "31" collisions occurred in Forrest and Lamar Counties. In order to assess the accuracy of these points, a United States Census Tiger Line File was obtained from Mississippi Automated Resource Information System (MARIS), which is a data layer in vector format containing all roads within Forrest and Lamar Counties, along with street names and numbers (MARIS 2010).

A land cover map of the study area from the National Oceanic and Atmospheric Administration Coastal Change Analysis Program (NOAA C-CAP) was obtained to quantify landscape composition and structure around each collision site (NOAA 2010). These remotely sensed images were created in 2006 and have a spatial resolution of 30 meters. C-CAP images are raster files that have been classified into twenty land-cover classes and have an overall target accuracy of 85 percent (NOAA 2010). In an effort to account for landscape features not represented in C-CAP imagery, high-resolution aerial photography was obtained from the National Agriculture Inventory Program (NAIP) (Nielsen et al. 2003). The U.S. Department of Agriculture Farm Services Agency (USDA-FSA) acquires 1-meter natural-color digital imagery during the growing season on a yearly basis for all counties in the continental U.S. (MARIS 2010). The NAIP imagery collected for this study was acquired during the 2010 growing season.

3.3. Data Preprocessing

To check the accuracy of the DVC points, the MOHS data was integrated as a point layer in ArcMap 10, using latitude and longitude coordinates of each collision. The location of each point was then verified with the U.S. Census Tiger Line File to determine that each collision site documented on the official crash report coincided with its correct road. If a point was located on a road that did not match what was listed in the collision report, it was classified as invalid and excluded from further analysis. Nine DVC sites in were eliminated this way as a result of conflicting information between latitude, longitude, and road location, reducing the data set to 338 functional DVC sites.

In order to analyze landscape structure and composition of each DVC site, the researcher reclassified the twenty land-cover classes of the C-CAP imagery in ArcMap 10. Following methods used in Hubbard et al. (2000) and Nielsen et al. (2003), the twenty classes were collapsed into five: developed land (urban), agriculture/grassland, forest, wetland, and water (Figure 6). Lastly, in order to account for variables that might not be represented in C-CAP imagery, NAIP imagery was used to visually classify the number of road lanes at each site as 2, 4, or 6 (Nielsen et al. 2003). This variable was used as a proxy for traffic volume, with more road lanes representing higher volumes of traffic.

3.4. Nearest Neighbor Analysis

To determine whether DVC sites in the study area are randomly distributed, the average nearest neighbor was calculated for each DVC site in ArcMap 10, using a 95% confidence level (Gonser et al. 2009). This created an index based on distance from one DVC site to the closest adjacent site. The resulting output is a ratio (standardized nearest neighbor ratio) of the observed distance divided by a hypothetically expected random distance, which allows for comparison across spatial areas. Any data set whose nearest neighbor ratio is greater than or equal to 1 has a random spatial distribution. Similarly, data sets with values less than 1 have non-random distributions (McGrew and Monroe, 2000). As Gonser et al. (2009) points out, a central assumption of the nearest neighbor analysis is that all points are able to be located anywhere within a study area. DVCs must occur on roads, however, and therefore do not have the freedom to be located at any geographical location. To control for the inherent non-randomness associated with road networks, ArcMap 10 was used to generate 338 random points onto the U.S. Census Tiger Line road file. The average nearest neighbor was then calculated for this randomly generated set of points to compare with the DVC point data.

3.5. GIS Techniques

With 338 functional DVC sites georeferenced to the Tiger Line File in ArcMap (Figure 7), slightly less than half of the collision sites (n = 160) and an equal number of control sites (n = 160) were randomly selected from the study area for further analysis (Nielsen et al. 2003). A grouping variable (binary dependent variable) was added to both sets of points, classifying them either as "1" for a collision site and "2" for a control site. Using Spatial Analyst in ArcMap 10, a buffer with a radius of 1,200 meters was created around each collision site. This buffer size was chosen as an estimate of the average diameter of a white-tailed deer's home range, which is normally about 2,400 meters (Gonser et al. 2009).

In order to attain the most definitive representation of a control sample, roads that contained a DVC were selected. This selection was then reversed within the attribute table, which only allowed roads that did not contain a DVC to be displayed. These roads were then exported as a new layer. From there, the researcher added the layer that only contained roads with no DVCs along with the layers of the collision site buffers. Road segments that overlapped with a collision site buffer were then erased, leaving only roads that contained no DVCs and did not lie in a collision site buffer. These control roads were then assigned a number through a random number generator, and after sorting them in ascending order, the first 160 road segments were selected. These control road segments were then bisected so that the researcher could insert control site buffers. This process allowed the researcher to choose only control sites that were not located on a road where a DVC occurred and were not inside a collision site buffer. The researcher did, however, allow collision buffers to overlap control buffers and vice versa. The total land area within each buffer was approximately 7.1 square kilometers.

All collision and control site buffers were then added as layers in ArcMap 10, along with the reclassified C-CAP imagery, to create a buffer/land-cover map. Using the randomly selected collision and control sites, as seen in Figure 8, land-cover data was extracted from each buffer to create 320 individual landscape units (1 = 160, 2 = 160), which were then converted to ASCII (American Standard Code for Information Interchange) format to ensure compatibility with FRAGSTATS 3.3.

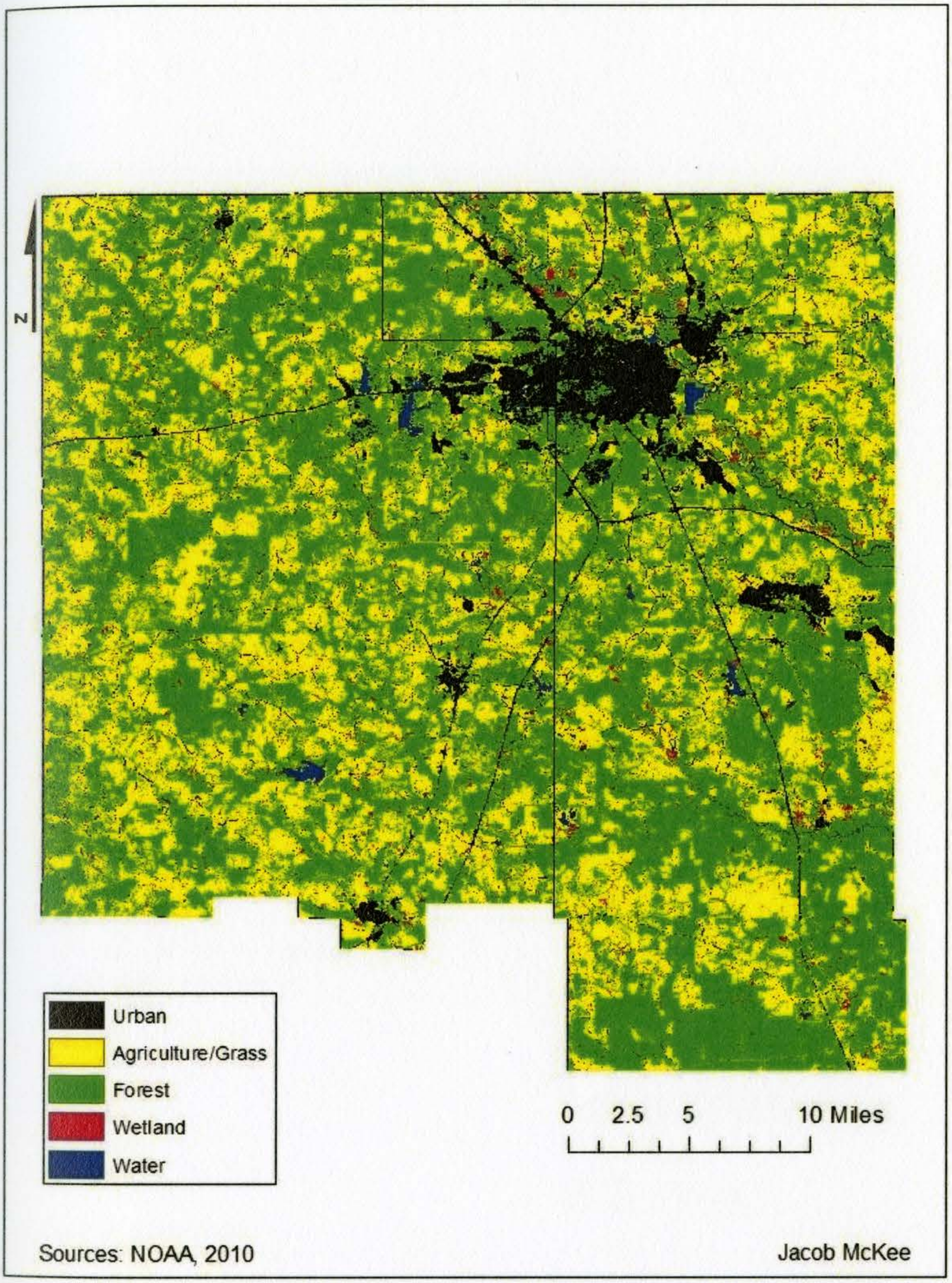

Figure 6. Study Area with Reclassified C-CAP Imagery.

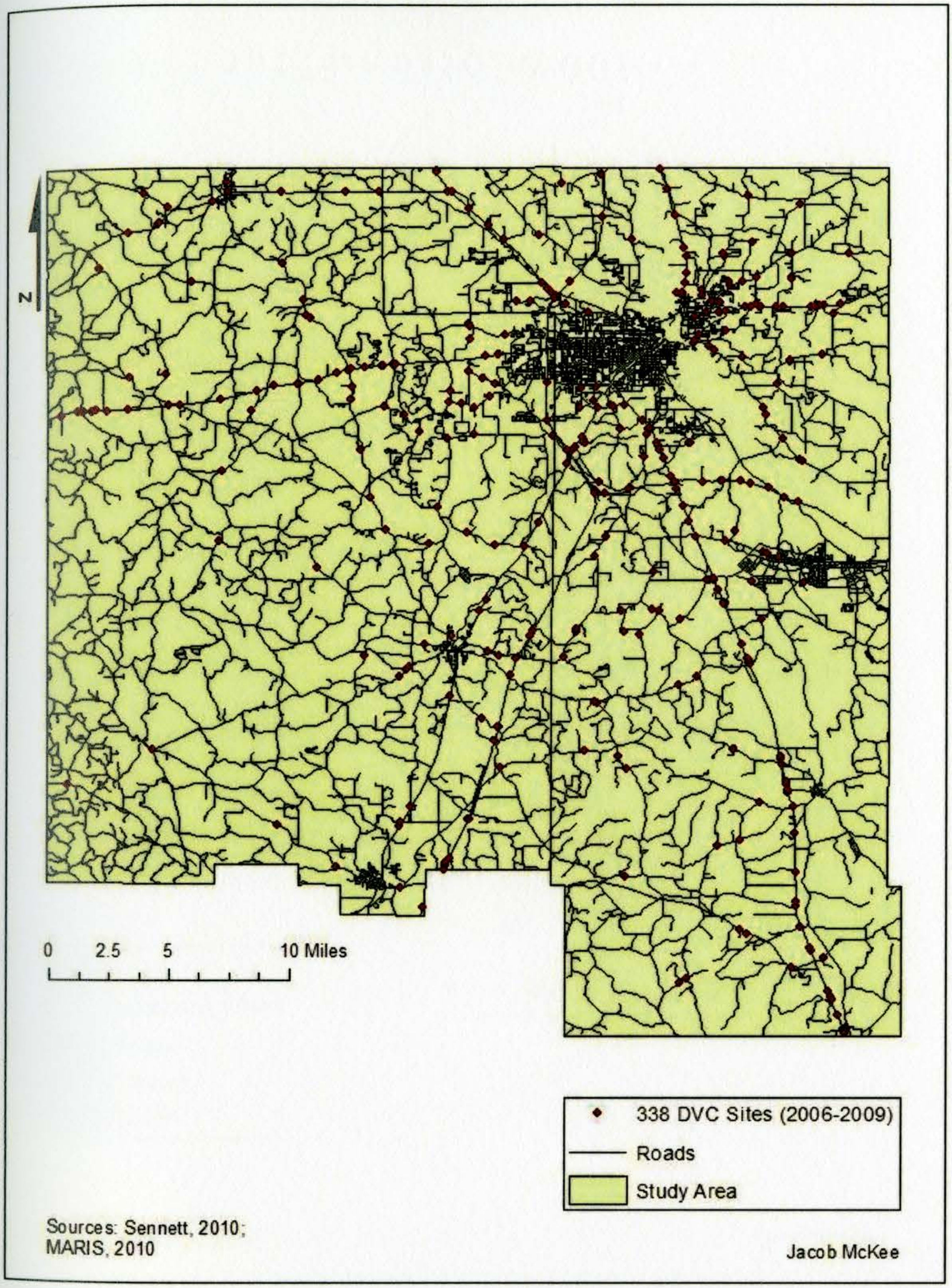

Figure 7. Study Area with 338 Georeferenced DVC Sites.

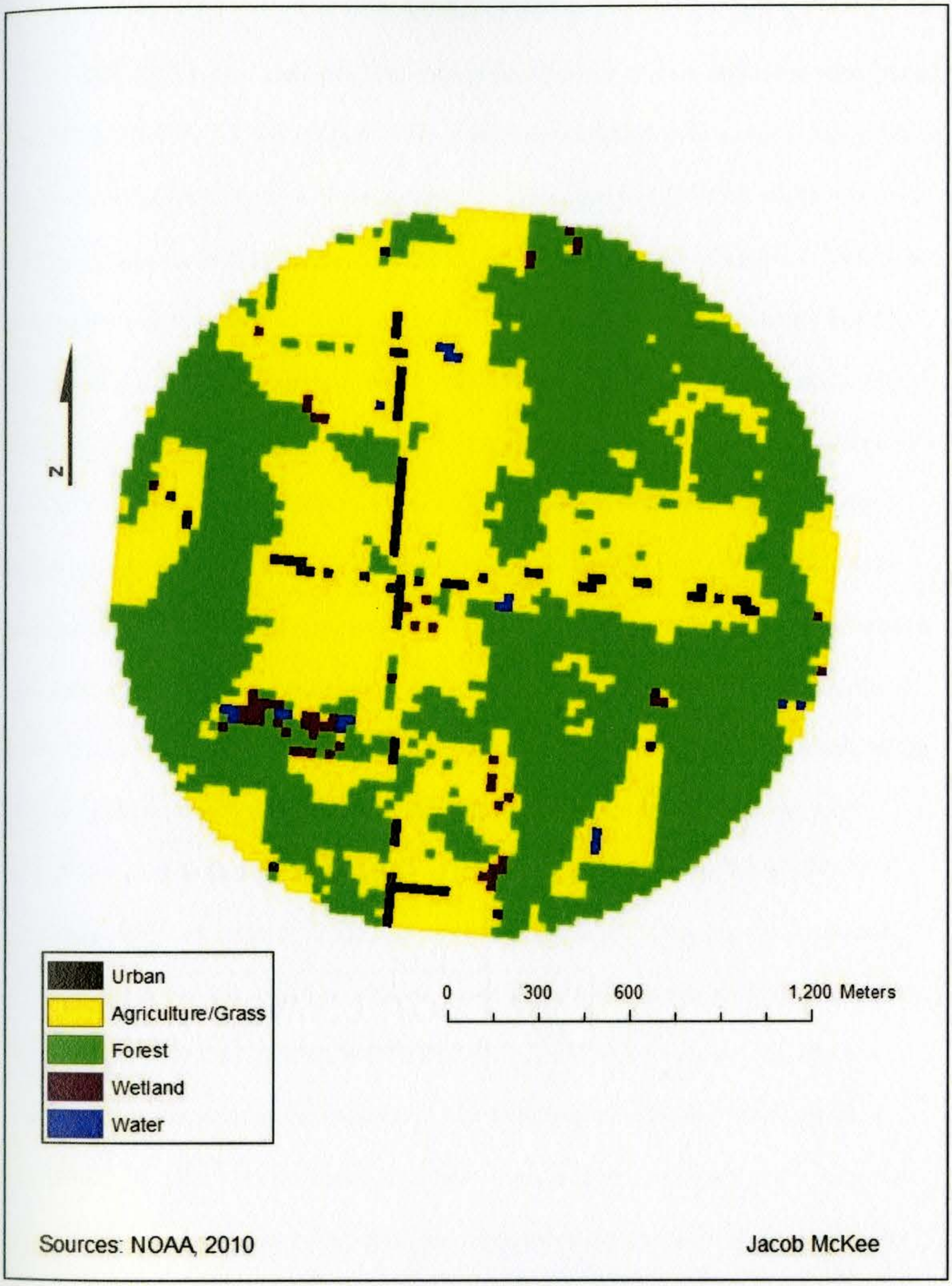

Figure 8. Land-Cover Data Extraction from DVC Buffer 134.

3.6. FRAGSTATS

The ASCII files containing the individual landscapes (*n* = 320) were then loaded into FRAGSTATS 3.3. This application quantifies landscape structure and composition, computing landscape metrics to understand the landscape patterns that might influence ecological processes (McGarigal and Marks 1995). Following Nielsen et al. (2003), the 8-neighbor rule was used to calculate the following landscape-level metrics (Table 1): number of patches, patch size coefficient of variation, edge density, mean patch edge, mean shape index, mean perimeter-area ratio, mean patch fractal dimension, Shannon's diversity index, and Shannon's evenness index. Shannon's diversity and evenness indices measure the diversity of land-cover classes in a landscape and determine the proportional equality of all land-cover classes (Turner 2001). Landscape-level metrics calculate composition and configuration for an entire buffer area. For example, for a given buffer in the study area, FRAGSTATS calculated total number of patches, using the entire landscape in the buffer for analysis. Similarly, at the class-level, the 8-neighbor rule was used to calculate the following metrics (Table 1) for the five land-cover classes: percentage of landscape, number of patches, patch size-coefficient of variation, edge density, mean patch area, mean shape index, mean perimeter-area ratio, and mean patch fractal dimension (Nielsen et al. 2003). These class-level metrics calculate the values for each land-cover class within in a landscape. For example, FRAGSTATS calculates number of patches for each of the five land-cover classes in a given buffer, meaning that each metric has separate values for each of the land-cover classes. Using FRAGSTATS 3.3, I calculated forty-nine variables for each buffer to verify which landscape features best distinguish between DVC sites and control sites. In total, fifty variables were calculated for analysis.

Table 1

Metrics Quantified with FRAGSTATS

Landscape Level Metrics	Class Level Metrics (for five cover types)
Number of patches	Percentage of landscape
Patch size coefficient of variation	Patch size coefficient of variation
Edge density	Number of patches
Mean patch area	Edge density
Mean shape index	Mean patch fractal dimension
Mean perimeter-area ratio	Mean shape index
Mean patch fractal dimension	Mean perimeter-area ratio
Shannon's diversity index	Mean patch area
Shannon's evenness index	

Note. Adapted from Nielsen et al. 2003.

3.7. Statistical Analysis

Once all variables had been calculated, the statistical software SPSS 18 (Statistical Package for the Social Sciences) was used to achieve my research objectives. To determine if one land-cover classes was more associated with DVC sites than others, a two-sample difference of means test was conducted for the percentage of each land-cover class surrounding every collision site. Because the agriculture/grassland class was the only variable that was normally distributed, the researcher used a two-sample difference of means test and the non-parametric Wilcoxon-Rank Sum for the other four land-cover classes. For the difference of means, a 95% confidence level was used to determine if the percentage of each land-cover class significantly differed between collision and control sites. The researcher also used discriminant analysis to create a statistical model to understand which landscape variables best distinguish between collision and control sites. This multivariate statistical technique is used on classed data to assess differences between classes and to predict the classes of unknown data (Huberty 1994). This procedure is specifically developed to estimate parameters in models utilizing grouping variables, such as collision site (1) and control site (2) (Allen 1997). This technique sequentially creates discriminant functions until all of the variance in a data set is explained. Essentially, discriminant functions are best-fit lines that are drawn through a multivariate dataset and used for classification and prediction (James 1985).

The discriminant analysis was conducted using a step-wise method with the collision (160) and control sites (160). The significance of the variable's *F*-statistic and a 95% confidence interval were used for variable selection, meaning the significance of each variable had to be less than 0.05 (statistically significant) for selection. This strategy of statistical modeling adds individual variables in a series of steps. If the significance of a variable's *F*-statistic was less than 0.05, meaning that it explained a significant amount of the variability between two classes, it became implemented in the model. Once a variable had been selected, however, it could be removed from the model in the next step if the significance of the variable's *F*-statistic became greater than 0.10 and was no longer statistically significant (James 1985, Younger 1985). The discriminant analysis created one discriminant function utilizing seven of the fifty variables that explained 100% of the variance. Although this procedure cross-validates the model's accuracy by removing each collision/control site individually and allowing the model to classify each site, the accuracy was manually assessed by randomly selecting twenty

DVC sites that had not been used in the analysis. For this process, the researcher randomly selected collision sites from the unused portion of the data set, created buffers for these sites, and derived their landscape metrics. These sites and their corresponding variables were then entered into the model to determine if the model would classify them as collision sites (1).

CHAPTER IV

ANALYSIS OF DATA

4.1. Temporal Trends

In order to observe the temporal trends in the 338 DVCs that occurred in the study area between 2006 and 2009, I divided the DVCs into the following temporal categories: seasonal (Figure 9), monthly (Figure 10), and hourly (Figure 11). Figure 9 shows DVCs increase from summer to winter, with a slight increase in spring, yet the largest increase occurs in fall and winter. Figure 10 shows an increase in DVCs in October, November, and December, with a peak in January. Similarly, Figure 10 also shows a slight increase in DVCs during May. DVCs are most likely to occur in late fall and early winter when male deer are most bold and mobile due to the breeding season (Sudharsan et al. 2006, McShea et al. 2008, Ng et al. 2008). The breeding season for white-tail deer generally occurs during the first two months of each year in South Mississippi, which explains the high numbers of DVCs in January ($n = 72$), accounting for 21% of the total, and in February ($n = 145$), with 43% of the total (MDWFP 2009). This winter peak in DVCs also coincides with deer hunting season, which might induce deer to be more mobile than at other times of the year. Likewise, the slight increase in DVCs during the late spring can be explained in part by the dispersal of fawns with their mothers and foraging during the early green-up of roadside vegetation (Ng et al. 2008). These findings are similar to other studies, such as Sudharsan et al. (2006), Madsen et al. (2002), McShea et al. (2008), and Ng et al. (2008), which all found that DVCs were more frequent during the white-tail breeding season.

Figure 11 shows that DVCs occur most often around dawn and dusk. From 2006 to 2009, seventy-two DVCs occurred between the hours of 4:00 and 8:00 a.m., which accounts for 21 percent of reported collisions. Some 165 DVCs, or 49 percent of the total, occurred between the hours of 5:00 and 10:00 p.m. Because white-tailed deer are crepuscular species, meaning they are most active around dawn and dusk, they are most vulnerable during these hours as they move from feeding areas to bedding areas and vice versa. This explains to a great extent why DVCs are more common during these hours. These hourly patterns also suggest another possibility regarding DVCs. During winter months when the sun sets earlier, the period of high deer mobility coincides with higher automobile traffic volumes associated with afternoon rush hour. During the summer months, on the other hand, the sun sets later in the evening and the period of high deer mobility does not coincide with the afternoon rush hour. This likely explains why substantially fewer DVCs occurred during the summer.

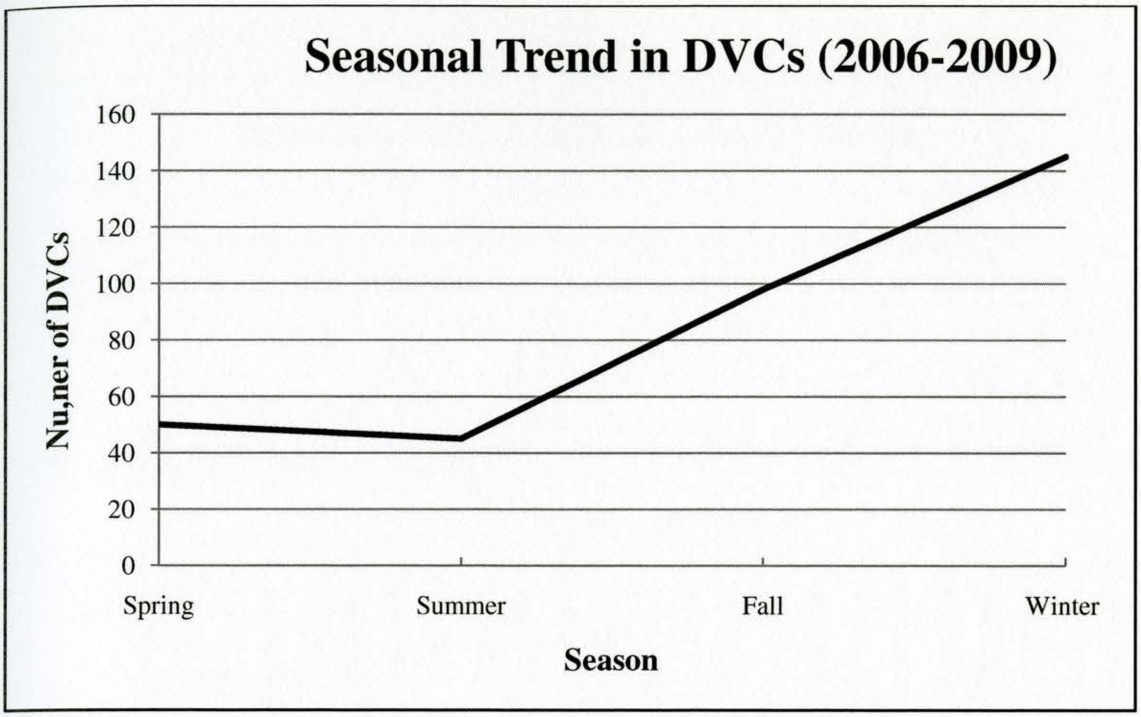

Figure 9. Seasonal Trend of DVCs in the Study Area from 2006-2009.

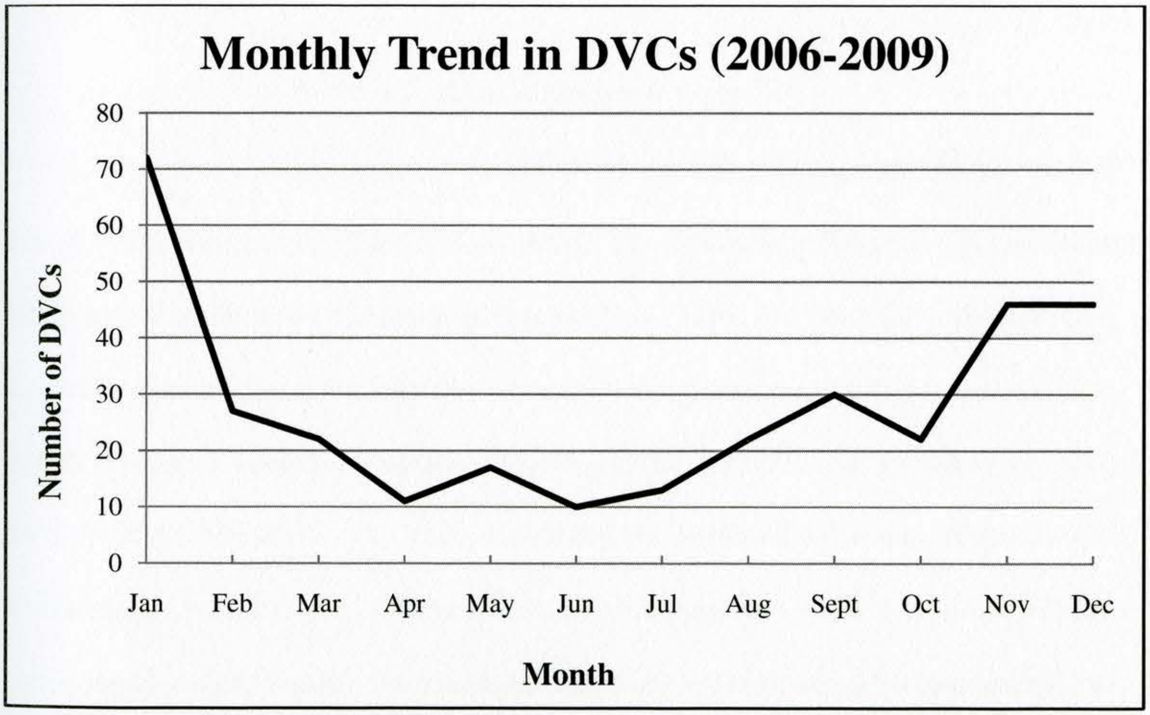

Figure 10. Monthly Trend of DVCs in the Study Area from 2006-2009.

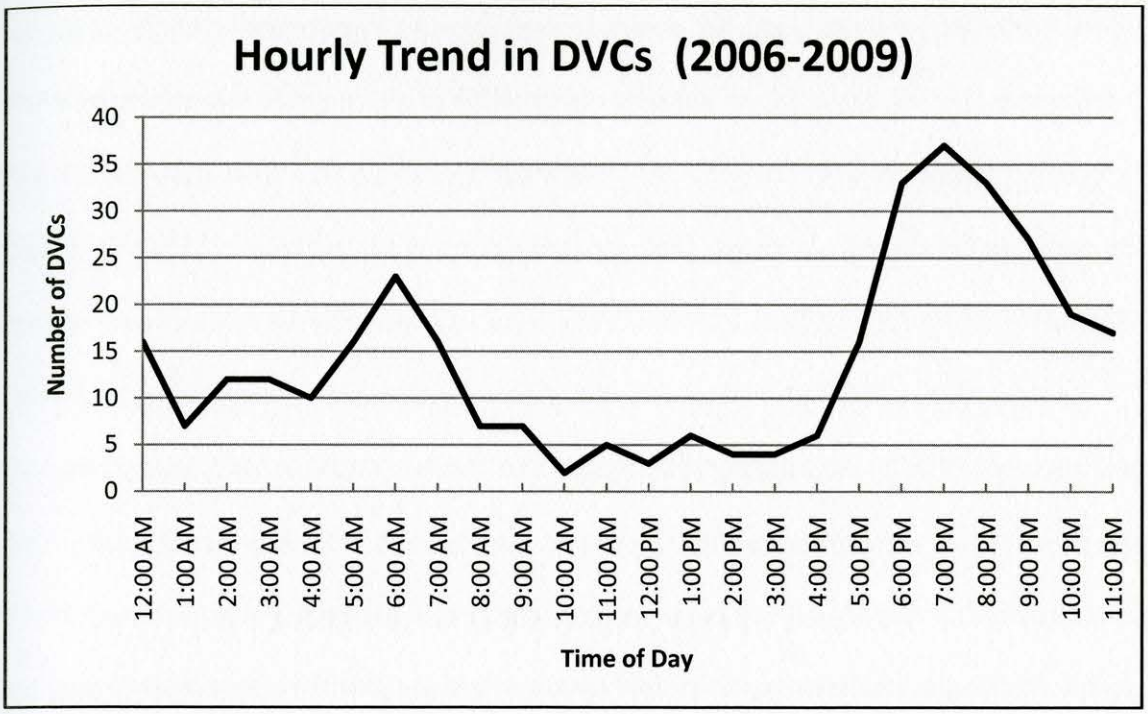

Figure 11. Hourly Trend of DVCs in the Study Area from 2006-2009.

4.2. Nearest Neighbor Analysis

Recent studies have shown that DVCs are not only aggregated in time, but also across space (Seiler 2005, Gonser et al. 2009). To determine whether this is true for my study area, I conducted a nearest neighbor analysis (Table 2). The null hypothesis for this test states that the spatial distribution of features is random (McGrew and Monroe 2000). I used a 95 percent confidence interval to calculate the average nearest neighbor for each of the 338 DVC sites. I also calculated the standardized nearest neighbor ratio, which ranges from 0.0 to 2.149 and allows direct comparison of results across different phenomena or spatial units. Nearest neighbor ratios vary according to the distribution of a data set, ranging from perfectly clustered (value of 0) to perfectly dispersed (value of

2.149), and to random distribution (value of 1) (McGrew and Monroe 2000). Since I used a confidence interval of 95%, a p-value less than .05 required that I reject the null. The analysis produced a p-value of 0.00, which allowed me to reject the null hypothesis that the collision sites were randomly distributed. Analysis of the control sites yielded a p-value of 0.557, which led me to accept the data were randomly distributed as expected. For the standardized nearest neighbor ratio, the DVC sites (0.611) have a more clustered pattern, suggesting that various factors influence the location of DVCs as opposed to random chance. The nearest neighbor ratio of the 338 random sites (0.978) indicates that their pattern is random. These results are supported by other studies that have shown that DVCs are not spatially random, and further expand upon existing research that pertains to the spatial distribution of DVCs in the southern United States (Neilsen et al. 2003, Seiler, 2005, Gonser et al. 2009).

Table 2

Results of Nearest Neighbor Analysis

Nearest Neighbor Variable	338 DVC Sites	338 Random Sites
Observed Distance	0.008	0.017
Expected Distance	0.014	0.018
Nearest Neighbor Ratio	0.611	0.978
Z-Score	-13.654	-0.603
p-value	0.00	0.557

4.3. Statistical Analysis

The null hypothesis for the two-sample difference of means test states there is no significant difference in the percentage of agriculture/grass land-cover between collision sites and control sites. Using a 95% confidence interval, the null is rejected if the p-value is less than 0.05. The mean percentage of agriculture/grass land-cover at collision sites was 36.15% ± 1.23 and 25.77% ± 1.73 for control sites. This test produced a Z-score of 4.88 and a p-value of 0.000, leading me to reject the null hypothesis because the amount of agriculture/grass land-cover was significantly greater at collision sites. For the other four land-cover classes, I used the non-parametric Wilcoxon-Rank Sum to test the significance of the percentage of land-cover between collision and control sites. As these data were not normally distributed, the Wilcoxon test ranks sample observations to measure the magnitude of differences in ranked positions instead of using the mean (McGrew and Monroe 2000). Table 3 shows the mean rank for the percentage of urban, forest, wetland, and water land-cover at both collision and control sites. Because all variables produced a p-value less than 0.05 (95% confidence interval), I rejected the null hypothesis because the percentages of land-cover classes were significantly different between collision and control sites. The mean rank shows that urban land-cover was significantly less at collision sites, while forest, water, and wetlands covered significantly larger areas.

Table 3

Results of Wilcoxon-Rank Sum Test

Land-Cover	Collision	Control	p-value	Test-Statistic
Percentage Urban (Mean Rank)	147.8	173.2	0.014	2.456
Percentage Forest (Mean Rank)	188.13	132.87	0.000	-5.343
Percentage Water (Mean Rank)	182.53	138.47	0.000	-4.26
Percentage Wetland (Mean Rank)	195.88	125.12	0.000	-6.842

The discriminant analysis created one function that utilized seven of the original fifty variables (number of road lanes, patch size coefficient of variation, mean patch area of water, mean patch area, percentage of forest, mean patch area of agriculture/grassland, and mean perimeter-area ratio of urban land-cover) and explained 100% of the variation.

Table 4 shows the stepwise process of variable selection and removal depending on the variable's significance. Using a 95% confidence interval, variables had to have a p-value less than 0.05 to be implemented in the model. If the p-value of any variable became statistically insignificant, it was removed from the model, as demonstrated by the variable edge density (ED) in step six of Table 4. Furthermore, a variable in the model with a p-value below 0.05 can be used to discriminate sites as collision or control, but as its p-value increased above 0.05, it could no longer be used in the model. Although the two-sample difference of means and Wilcoxon-Rank Sum tests showed that the percentage of each land-cover class was significantly different between collision and control sites, the discriminant function only used percentage of forested land-cover. This was due to the fact that when all variables are analyzed together, one variable might no longer be statistically significant, and other variables might explain more of the variation, such as the case with mean perimeter-area ratio of urban land-cover (MPAR Urban). Percentage of urban land-cover was not used in the analysis, because the variable, MPAR Urban, was more effective at statistically differentiating between classes and explaining more of the variance.

Table 4

Stepwise Process of Variables in the Model

Step	Entered	Removed	Wilks' Lambda		
			Statistic	Exact F	
				Statistic	p-value
1	# of Road Lanes		0.735	114.548	0.000
2	ED		0.633	91.715	0.000
3	PSCV		0.612	66.805	0.000
4	MPA Water		0.601	52.262	0.000
5	MPA		0.589	43.826	0.000
6		ED	0.59	54.813	0.561
7	% Forest		0.58	45.452	0.000
8	MPA Ag		0.571	39.139	0.000
9	MPAR Urban		0.558	35.257	0.000

The canonical discriminant function coefficients show the relationship between variables in the model and the classification of sites, and also demonstrate the strength of each variable in the classification. As shown in Table 5, the variables most important to the model and classification include number of road lanes and mean patch area (MPA), followed by the mean patch area of water (MPA Water) and percentage of forest land-cover (Percent Forest).

The structure matrix (Table 6) shows the strength of the correlation for each variable with regard to the best-fit line of the model. This table shows that the number of road lanes is most strongly correlated with the model, which means that it is the primary variable for prediction and classification, followed by patch size coefficient of variation (PSCV), mean patch area (MPA), and percentage of forest land-cover (Percent Forest). The structure matrix also demonstrates that as the number of road lanes, percentage of forest land-cover, MPAR Urban and mean patch area of agriculture/grassland (MPA Ag) increase, so does the probability of a site being classified as a collision site (Figure 12). In other words, as a landscape becomes more dominated by forest cover, patches of agriculture/grassland become larger, or the number of road lanes increases, so does the probability of a DVC.

Landscapes with high proportions of forest cover or large uninterrupted areas of agriculture/grass land-cover are beneficial for deer, offering numerous forest patches to seek cover, yet have large contiguous patches of agriculture/grassland in which to forage. DVCs might be more common in areas dominated by forest because deer dispersing to foraging areas may not be clearly visible to motorists until it is too late to avoid a collision (Nielsen et al. 2003). DVCs might also be more common around areas with unbroken agriculture/grassland patches because deer move to these areas as part of their foraging behavior. These results are similar to those in Iowa by Hubbard et al. (2000), who found that the probability of a DVC increased as average size of grassland patches and the interior of wooded areas grew. Gonser et al. (2009) also found that planted and cultivated land was the primary land-cover around DVC sites in Indiana, followed by forest. Bashore et al. (1985) also found that DVCs in Pennsylvania were more common when approaching a woodland-field interface.

Mean perimeter-area ratio is a measure of shape complexity with large simple shapes such as circles or squares having a small ratio because the perimeter or edge is approximately equal to the core area (McGarigal and Marks 1995). As land-cover shapes become smaller or more complex, this ratio increases. The probability of a DVC increases as the mean perimeter-area ratio of urban land-cover increases, changing from large, relatively simple shapes of continuous urban cover to smaller and more complex shapes interspersed across the landscape. In large contiguous areas of urban land-cover surrounding a city, DVCs are not likely to occur due to lack of preferred natural habitat. Moving away from a city, small, complex features increasingly characterize exurban and rural landscapes and it is here that the likelihood of DVCs increases. These landscapes offer more edge habitat for deer, and these patches are typically interspersed with prime deer habitat of forest and undeveloped land, thus increasing the likelihood of DVCs as humans encroach on deer habitat. Research has also shown that deer home ranges shift toward residential areas in winter to browse on landscape plantings due to the scarcity of food in rural, undeveloped areas (Kilpatrick and Spohr 2000). Exurban landscapes also typically have low deer harvest rates from hunting, allowing deer over-population to become a significant problem (Storm et al. 2007).

The number of road lanes was an important variable in the model, demonstrating that as the number of road lanes increased, so did probability of DVCs. Because this variable was used as a proxy for traffic volume, traffic intensity could possibly be an indicator of collision sites, similar to the findings of Seiler (2005), Farrell and Tappe (2007), and McShea et al. (2008) which found that traffic was a significant factor in DVC prediction and Hubbard et al. (2000) who also found that more lanes of traffic increased DVC probability.

In contrast to variables that increase the probability of a DVC, mean patch area, patch size coefficient of variation, and mean patch area of water (MPA Water) are more associated with control sites (Figure 13). The mean patch area is the average patch size of all land-cover types in a given landscape. On average, as land-cover patches grow in size, the likelihood of a DVC occurring nearby decreases. This illustrates that DVCs are more common around fragmented landscapes with smaller patches. This suggests that in areas of more contiguous cover types and resources, deer might not have to travel as far for sustenance, thus lowering the probability of their crossing roadways. As landscape patches become smaller, however, an area might not contain enough resources to sustain the deer, forcing them to disperse further afield for resources (Hussain et al. 2007). The patch size coefficient of variation measures variability of patch size across all land-cover types as a percent, with higher percentages representing higher variability. As variability in patch sizes increases, the probability of a DVC decreases. Hubbard et al. (2000) also found that the probability of DVCs decreased with patch size variability. High variability in patch size typically means large proportions of one land-cover type with only a few small patches of other types. This raises the possibility that large areas of urban land-cover interspersed with recreational or green areas are not conducive for deer habitat. Similarly, as mean patch area of water becomes larger, the chance of a DVC decreases. This result contrasts with Ng et al. (2008) who found a weak, yet significant relationship between water and probability of a DVC in Alberta, Canada.

Table 5

Standardized Discriminant Function Coefficients

	Function 1
# of Road Lanes	-0.872
% Forest	-0.019
MPA Ag	-0.037
MPAR Urban	0.002
MPA Water	0.146
MPA	0.363
PSCV	0.003
(Constant)	-1.658

Table 6

Structure Matrix

Function 1	
# of Road Lanes	-0.675
PSCV	0.539
MPA	0.464
% Forest	-0.39
MPAR Urban	-0.186
MPA Water	0.115
MPA Ag	-0.092

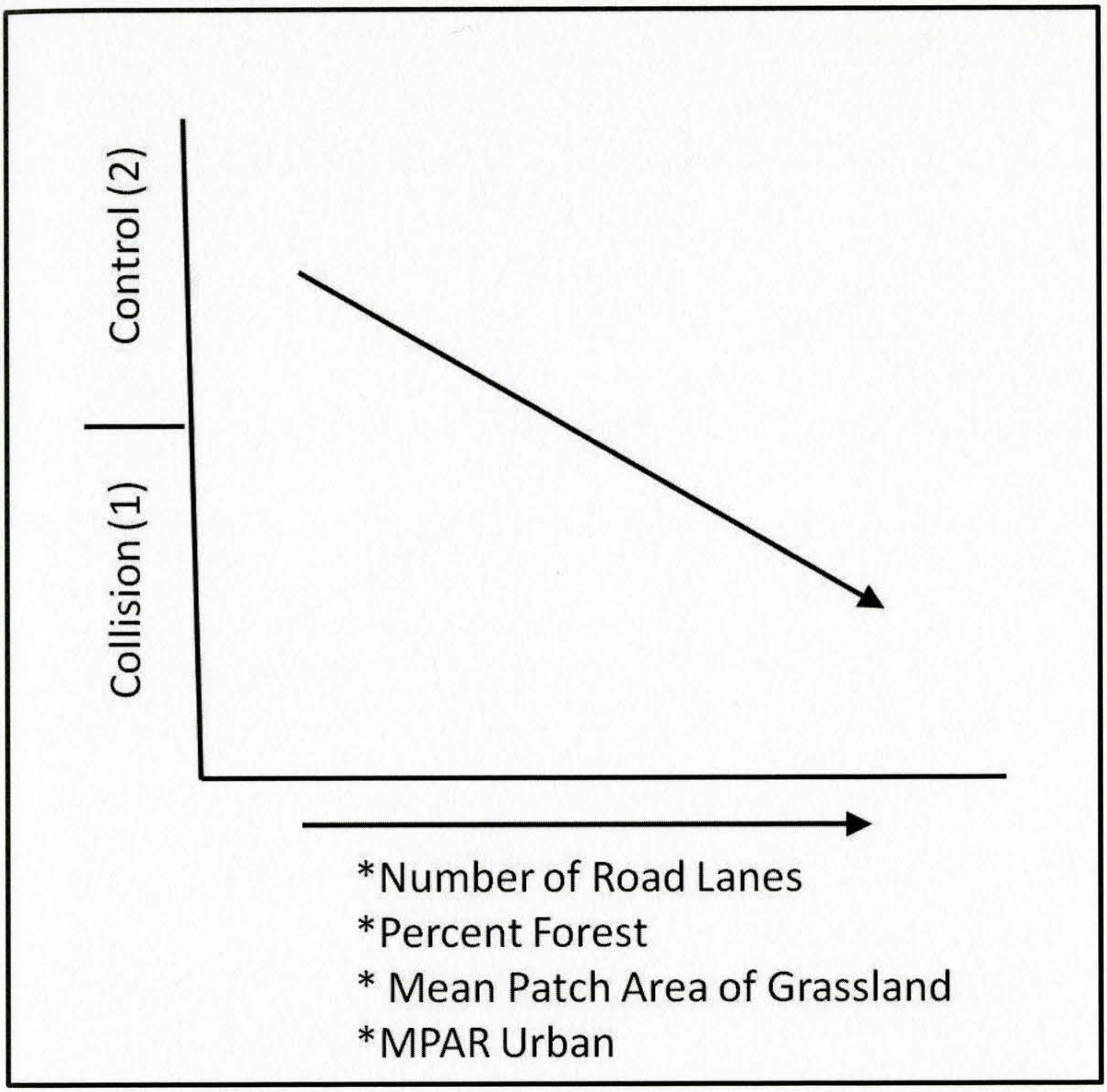

Figure 12. Variables Which Increase the Probability of a DVC.

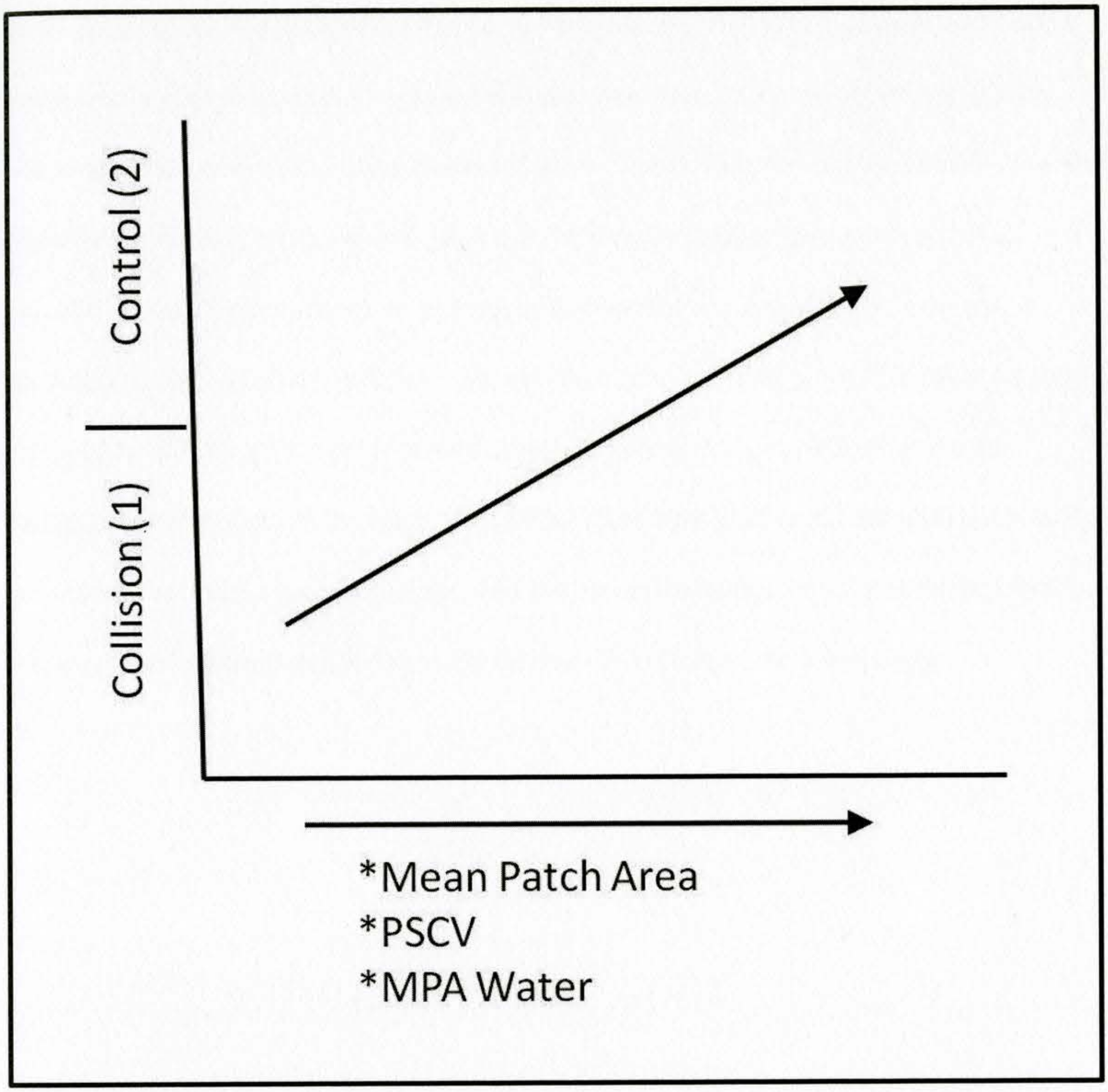

Figure 13. Variables Which Decrease the Probability of a DVC.

To assess the accuracy of the model's predictive capabilities, I ran each site through the discriminant function to determine how the model classified them. I then cross-validated the predictive model by removing each site individually and letting the model classify it. The discriminant function classified 81.3 percent of the sites correctly and 80.3 percent after cross-validation. To manually assess the accuracy of the function, I randomly selected twenty collision sites from the data points not used in the analysis. I generated buffers and derived landscape variables using FRAGSTATS and NAIP high-resolution aerial photography. I then ran these sites through the model as unclassified points to observe how the model classified them. Table 7 shows that the model correctly classified seventeen of the twenty sites as collision, equivalent to an accuracy of 85 percent. The P (G/D) column is the probability that the site actually falls into the predicted group. Most of the seventeen sites that were correctly classified have a strong probability of falling into the predicted group, demonstrating the effectiveness and strength of the model. In the same way, of the three sites that were classified incorrectly, two had a near-50% chance of falling into the predicted group, which means they likely contained landscape characteristics similar to both collision and control sites.

Table 7

Accuracy Assessment of Model

Actual Group	Predicted Group	P (G/D)
Unclassified (1)	2	0.518
Unclassified (1)	1	0.659
Unclassified (1)	1	0.972
Unclassified (1)	2	0.569
Unclassified (1)	1	0.675
Unclassified (1)	1	0.517
Unclassified (1)	1	0.504
Unclassified (1)	1	0.517
Unclassified (1)	2	0.786
Unclassified (1)	1	0.511
Unclassified (1)	1	0.979
Unclassified (1)	1	0.988
Unclassified (1)	1	0.981
Unclassified (1)	1	0.971
Unclassified (1)	1	0.978
Unclassified (1)	1	0.966
Unclassified (1)	1	0.986
Unclassified (1)	1	0.976
Unclassified (1)	1	0.970
Unclassified (1)	1	0.686

CHAPTER V

CONCLUSION

5.1. Overview

Deer-vehicle collisions are one of the most common and dramatic forms of human-environment interaction (Gonser et al. 2009), posing a threat to both traffic safety and animal welfare, and serving as a drain on economic development and conservation initiatives (Seiler 2005). Unlike other species that do not tolerate environmental disturbance and land fragmentation, deer flourish in anthropogenic landscapes and have adapted by exploiting human-altered environments, feeding in agricultural fields, lawns, and roadsides (Rawinski 2008), thus intensifying the problems associated with human population growth. Furthermore, road expansion leads to habitat fragmentation and increased edge habitat, fundamentally altering how local ecosystems operate, yet creating a landscape well suited for DVCs (Forman and Alexander 1998). Unlike many other species, deer are not adversely impacted by fragmentation and proliferation of edge habitat. In fact, new road construction produces their ideal habitat by expanding or creating new grazing areas with easily accessible food sources (Gonser and Horn 2007). During times of the day when deer graze, primarily around dawn and dusk, they routinely cross roads encompassed in their home range, intermingling with traffic as they travel along road corridors (Putnam 1997). Not only do these disturbances and anthropogenic activities create a diverse mosaic of plant communities, but they also create patterns that can be useful in understanding where and why DVCs occur (Turner 2005).

Between July of 2007 and June of 2009, over 2.4 million DVCs occurred on U. S. roadways (State Farm Insurance 2009), far exceeding the number of deer killed annually through the affects of hunting in the United States (Coffin 2007). Given the significant impact of these collisions on wildlife, society, and the economy, it is imperative to gain a better understanding of the factors associated with these collisions in order to develop strategies to mitigate these incidents (Hussain et al. 2007). Though the results of published research are mixed regarding which factors play the most crucial role in DVC location and occurrence, most agree that DVCs do not have a random spatial distribution (Seiler 2005, Gonser and Horn 2007, Gonser et al. 2009), but instead are associated with roads in close proximity to forested areas (Farrell and Tappe 2006). Less conclusive are findings that link vehicle speed and traffic volume to DVCs (Hubbard et al. 2000). Furthermore, little research has been published on the factors that influence DVCs in the southeastern U.S. After analyzing the national data pertaining to DVCs in 2009, it is apparent that Mississippi is among the states where DVCs should be a top priority for mitigation. To gain a better understanding of the factors that influenced the distribution of DVCs, my thesis integrated GIS, remotely sensed imagery, FRAGSTATS and inferential statistics to analyze DVCs in Forrest and Lamar Counties of South Mississippi between 2006 and 2009. The primary findings of this research demonstrated that, for my study area, DVCs were most common during winter months around the hours of dawn and dusk, were not random spatial occurrences, and that landscape patterns can be quantified and developed into a predictive statistical model which utilized seven variables to predict areas of potential risk to DVCs with an accuracy of 85 percent.

5.2. Management Implications

Due to the fact that Mississippi has relied entirely on warning signs for mitigation, there are numerous possibilities for the application of these results. Roadways can be buffered and landscapes surrounding roads can be quantified and input in the model to determine potentially hazardous areas for the installation of warning signs, fencing, or vegetation modification. Studies have shown that deer-proof fencing along with warning signs is the most effective DVC mitigation strategy, but given the length of roadways in the study area, fencing might not be financially realistic given the cost of installation and maintenance (Putnam 1997, Clevenger 2001). Modifying roadside vegetation might be a beneficial countermeasure given the large number of deer seen foraging along roadsides in the study area. Although unappealing from an aesthetic standpoint, clearing roadside vegetation not only eliminates food resources for deer but also increases the ability of motorists to see deer approaching roadways (Mastro et al. 2008). Other research has shown that cutting roadside vegetation in early spring shortly after green-up causes plants to have lower nutritional value and palatability than when cut during the middle of the growing season, which can help push deer to browse in other areas (Rea 2003).

Currently, twenty-two states have implemented public education and awareness programs (Romin and Bissonette 1996). Studies have shown that many people believe DVCs are random, unavoidable events, so a public campaign to inform residents of the study area that they are in fact aggregated in both time and space might increase public awareness about the potential hazards of DVCs. This effort could be achieved through newspaper, radio, television, or website announcements by local and state government agencies (Curtis and Hedlund 2005, Seiler 2005, Mastro et al. 2008).

Lastly, another mitigation strategy utilized with success in Georgia, New Jersey, Iowa, and Ohio is the culling of excessive deer populations. Case studies have shown that reducing deer herd size successfully decreases DVCs by 49 to 78 percent (Butfiloski et al. 1997, DeNicola and Williams 2008). A recent bill proposing to establish a system of culling Mississippi's large deer population passed overwhelmingly in the State House of Representatives and is currently waiting passage in the State Senate (Northway 2010). If the bill does pass, DVCs might possibly decrease over the next decade and remain low as long as the population is kept in check.

5.3. Limitations

Deer-related accidents occur not only when a deer and vehicle collide, but also when accidents result from motorists' attempts to avoid deer (Stout et al. 1993). My data are conservative in that I only used collisions that were coded as type "31", which is limited to collisions involving a vehicle striking a deer. Collisions in which the driver swerved to avoid hitting a deer and collided with another object were not included. Furthermore, numerous studies have shown that annual rates of DVCs are underestimated given that only 17 to 50 percent are actually reported (Bissonette et al. 2008, Marcoux and Riley 2010). If the same underestimation found in these studies also occurs in my study area, there could have been as many as 395 to 507 DVCs during the 2006-2009 study period.

It is also possible that the 30 x 30 meter sensor resolution of my data was not adequately sensitive to identify subtle patterns or edges used by deer (Hubbard et al. 2000). A general rule of thumb is that the object being observed must be at least 1.5 times the size of a pixel to be detected. Therefore, in some areas, the 30 square meter resolution might be too coarse to identify patches of roadside vegetation. Furthermore, landscape patterns change based on the geographical scale at which they are represented (Benson and MacKenzie 1995, Greenberg et al. 2001). Landscapes with a considerable local variability are most affected by spatial resolution. Subtle patterns are best illustrated with higher resolutions whereas course resolutions are likely adequate for homogeneous landscapes (Benson and MacKenzie 1995, Greenberg et al. 2001).

Lastly, given that speed limit and traffic volume data were not consistently and accurately available for the entire study area, forcing me to use the number of road lanes as a proxy for traffic volume. Given the importance of this proxy variable to my analysis, subsequent research would benefit from more detailed and accurate data on traffic speeds and traffic volume.

5.4. Conclusion

Though the growing presence of deer in human environments has caused DVCs to increase in recent years, creating a significant problem for wildlife and transportation managers in the U.S. (Butfiloski et al., 1997), research has shown that DVC kill rates are not high enough to adversely affect local deer populations and the problem continues to grow. There is clearly a growing need to better understand the geography of this phenomenon and develop predictive models in order to minimize economic losses, improve traffic safety, prevent human injury, and decrease the number of injuries and deaths from DVCs (Putnam 1997, Seiler 2005, Bissonette et al. 2008). Citing the relative dearth of published research pertaining to DVCs in the southern U.S., this thesis expands the available literature and demonstrates that DVCs can be better understood and accurately predicted through use of statistical analysis and quantification of landscape structure and composition. Further, through the use of this or similar predictive modeling, effective action steps may be taken at the local level which will improve traffic safety with regards to DVCs and minimize associated economic loss.

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