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Putting the Wild Back into Wilderness: GIS Analysis of the Daniel Boone National Forest for Potential Red Wolf Restoration

A thesis submitted to the Graduate School of the University of Cincinnati in partial fulfillment of the requirements for the degree of Master of Arts in the Department of Geography of the College of Arts and Sciences

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Abstract

The red wolf (*Canis rufus*) is a keystone species because of its important ecological role as a top predator. Its restoration to historic ranges may help to promote ecosystem integrity, balance, diversity and health. However, as already outlined in the 2007 United States Fish and Wildlife Service Red Wolf Recovery Progress Report, at least two additional reintroduction sites within the species’ historic ranges are still required to support viable populations of red wolves. This thesis research aimed to contribute to the Red Wolf Species Survival Plan by identifying and evaluating potential sites within the Daniel Boone National Forest in eastern Kentucky for the reestablishment of red wolves.

In previous wolf habitat prediction models, road density served as the criteria for suitability. Researchers calculated simple road densities; however, the logistic regression models thus derived did not accurately predict wolf occupation. Roads with higher traffic volumes and areas with greater road densities should, in theory, pose greater risks to wolf mortality, and simple road density may not be an adequate measure to such purpose. This research, therefore, ranked roads by mortality risk and utilized kernel density estimation in Geographic Information Systems as a means to weight the road density and to predict suitable wolf habitat. This method may provide a better picture of the spatial reality of road influence. By using the red wolf habitat suitability model based on the rank class and kernel density estimation, nine potential restoration sites were predicted; whereas the suitability model based only on the simple density function failed to predict any sites.
However, the results of this research are not final. The human and coyote factors remain unknown, and validation of the model is impractical due to the lack of data and time constraints. Yet, efforts such as field verification have been made in an attempt to validate the model. If data are available, follow-up studies in North Carolina may be a feasible measure to further test the model.

Keywords: red wolf (*Canis rufus*); keystone species; habitat suitability model; kernel density estimation; geographic information systems (GIS); Daniel Boone National Forest
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Chapter One

Introduction

1.1 Challenges of Carnivore Conservation

Carnivores and wildlands are two sides of the same coin. The loss of one heralds the loss of the other. (Rasker and Hackman 1996: 992)

Aldo Leopold, the “father” of environmental ethics and wildlife management, considered carnivores “the ultimate test of a society’s commitment to conservation” (Noss et al. 1996). For an endangered flower, a small, sequestered woodlot will suffice. Large carnivores like the wolf, however, require large home ranges across a broad spectrum of landscapes, and, in a world of shrinking wildlands and expanding human landscapes, conservation proves challenging politically, economically, culturally and ecologically.

Habitat destruction, hunting, persecution, disease, and commercial trade of body parts have imperiled predators worldwide. As human population, resource demands and human-predator conflicts escalate, the outlook for mega-carnivores appears grim. Our historical relationship with predators exemplifies our predisposition to destroy rather than preserve. Carnivore conservation, therefore, involves cooperation and compromise on every scale—individual, local, regional, national and global—and on every level—biological, ecological, geographical, political, legal and social. Confounding the issue, every species has unique requirements for survival.

This thesis research was undertaken with the intent to contribute to the science of red wolf (Canis rufus) conservation and its reintroduction to the wild.
1.2 Research Objective

In Kentucky, the loss of keystone species in combination with the loss of the dominant forest canopy species, the American chestnut (*Castanea dentata*), has resulted in impoverished biotic communities (Maehr, Grimes and Larkin1999). Insect infestations and disease indicate environmental stress within the Daniel Boone National Forest (DBNF).

Ecosystems are holistic—if one aspect is degraded, the other components will be affected as well. In DBNF, drought, insect defoliation, soil compaction, changes in soil depth, and trunk and root injury weaken oak species. Trees weakened or suppressed by other factors such as fire and heavy browsing by wild ungulates become susceptible to the twolined chestnut borer (USDA 2008). In the northern portion of the DBNF, the black beetle has impacted the oak-hickory stands, an important ecosystem in the forest (USDA 2008). The Forest Service expects the arrival of the gypsy moth to the DBNF by 2010; undoubtedly, the overcrowded, maturing oak-hickory forests will face magnified losses due to gypsy moth defoliation (USDA 2008).

In 1997, the first DBNF sighting of the hemlock woolly adelgid occurred (USDA 2008). The hemlock woolly adelgid interferes with tree growth and causes needle discoloration and needles to drop prematurely, which affects tree health. Hemlock stands are dying from the hemlock woolly adelgid in neighboring states of Virginia and Tennessee (USDA 2008).

More than 100,000 acres of pine forests in the DBNF have already been lost to the southern pine beetle, “the most devastating insect assault ever documented on the Daniel Boone National Forest in Kentucky” (USDA 2008). Stress from tornado, ice, fire,
drought and overcrowding primed the stage for the southern pine beetle (USDA 2008). With nearly 80% of pine habitats devastated, cascading effects, directly and indirectly, have affected 1/4th of the plant and animal species (USDA 2008). Biologists had to remove the remaining 15 endangered red-cockaded woodpeckers (*Picoides borealis*) from the DBNF, and the notable absence of pine warbler and its song raises concerns (USDA 2008). Climatic change may enhance drought and fire-prone conditions in the forest, further stressing the tree communities and increasing risks for infestation.

With the absence of the wolf, a top predator, herbivore and mesopredator populations have exploded and contributed to the degraded ecosystem (Ripple and Beschta 2004).

Intense herbivory reduces the vegetation community, structure, diversity, and health (Ripple and Beschta 2004), and the recent reintroduction of elk into the DBNF will further escalate herbivory intensity. White-tail deer, the main prey for the red wolf, has dominated the Kentucky landscape. Prior to European settlement and predator elimination, the estimated average density of deer in North America was 3.1-4.2 deer/km² (Rooney 2001). Thomas Barnes, a wildlife specialist at the University of Kentucky, stated in a news report that the deer population in Kentucky had risen to approximately 1 million (Pratt 2007)—an estimated deer density of 9.6 deer/km². Moreover, deer densities greater than 7.9 deer/km² negatively affect the regeneration process of the overstory. Browsing by overabundant deer eliminate species, create monocultures and reduce tree heights below the required 0-7.6m height interval for songbird nesting and foraging (DeCalesta 1997). Songbird populations, therefore, diminish due to the habitat degradation by ungulate herbivory (DeCalesta 1997).
Predation by the small to mid-sized predator such as the coyote (*Canis latrans*), skunk (*Mephitis mephitis*) and raccoon (*Procyon lotor*) reduces the populations of smaller vertebrates such as song birds, rodents and lagomorphs (Estes, Crooks and Holt 2001; Ripple and Beschta 2004; Côté et al. 2004). As mesopredators and herbivores increase in numbers, the biodiversity of smaller fauna and flora decrease.

Wolves in their “pivotal” ecological role may help to maintain biodiversity and habitat quality if the predator succeeds in controlling prey populations (Ripple and Beschta 2004). As one of North America’s most critically imperiled vertebrates (NatureServe 2007) and one of the world’s most critically endangered canids (IUCN 2006), the red wolf should be considered a conservation priority. Protection of the endangered red wolf, an umbrella species, would thereby encompass and protect its habitats and all the plant and animal species within its home-ranges, benefiting the forest community. Through reintroduction of the red wolf, a native species, the DBNF may recover the integrity of its ecosystems and preserve biodiversity.

In 1980, red wolves were removed from the wild and placed in a recovery program that entailed captive breeding and reintroduction into the wild. Currently, the red wolf population has risen to 208 in captivity and 130 in the wild (USFWS report 2007). Approximately 20 packs of an adult pair and their pups are distributed in North Carolina (Phillips, Henry and Kelly 2003). Despite the progress, analysis suggests a population of 550 (330 in captivity, 220 in the wild) to be a stable population for genetic diversity (Phillips, Henry and Kelly 2003; DeBelieu 1991). With this goal in mind, the United States Fish and Wildlife Service (USFWS) calls for additional establishment sites for the rare and endangered *Canis rufus* (USFWS report 2003). This thesis research aims
to contribute to the Red Wolf Species Survival Plan (RWSSP) by identifying and evaluating potential sites within the DBNF for the reestablishment of red wolves.

### 1.3 Problem Statement

Restoration of the red wolf (*Canis rufus*) to historic ranges in eastern Kentucky will enable the reestablishment of a keystone species. Its top predator role may help to maintain ecosystem integrity, balance, diversity, and health. Thus far, however, researchers have not determined if the Daniel Boone National Forest (DBNF) in eastern Kentucky has the capacity to support viable populations of red wolves. Habitat analysis of the DBNF, therefore, rectifies this gap as it can provide relevant knowledge about the potentiality for red wolf restoration to the DBNF. It will be helpful to the United States Fish and Wildlife Service and red wolf recovery teams.

As indexes for wolf suitability, researchers on wildlife reintroduction focus mainly on density of prey, roads and humans. Mech (1995) denoted road density as the “yardstick” by which agencies and recovery teams measured wolf habitat suitability; however, previous researchers only used simple road densities without assigning a weighing factor. Roads with higher traffic volumes and areas with greater road densities should, in theory, pose greater risks to wolf mortality. To address this inadequacy, this research, therefore, ranked roads by mortality risk and utilized kernel density estimation in Geographic Information Systems (GIS) as a means to weight the road density and to predict suitable wolf habitat. This will provide a better picture of the spatial reality of road influence.
Chapter Two

Literature Review

2.1 Introduction

2.1.1 Historical and Current Status of the Red Wolf

Early European settlers brought not only their religious and political beliefs to the New World, but also their negative views of the wolf (Fritts et al. 1997), a “nearly fanatical hatred of wolves” (DeBlieu 1991). Myth and folklore have portrayed them as “savage and demonic” (DeBlieu 1991). People, thereby, associated the wolf with evil, darkness, sorcery, malicious and cunning, and they feared superstitions of werewolves. By the eighteenth century, the wolf symbolized “the untamed land that had to be subdued in the name of civilization” (Fritts et al. 1997), and, at the time, taming the wildlands represented humankind’s manifest destiny (Phillips, Henry and Kelly 2003).

In Eurasia, persecution had eliminated wolves except in the central Appenine Mountains of Italy, the Cantabrian Mountains of northern Spain, the Carpathians of Eastern Europe, the northern parts of the former Soviet Union, and the central plains and mountainous regions of Asia (Mech 1995). In America, man defeated the wolf in the two-hundred year long war against the predator (Phillips, Henry and Kelly 2003), with wolf populations surviving only in Canada, Alaska, and in the wildlands of northern Minnesota and nearby Isle Royale National Park in Lake Superior (Mech 1995).

disease control, for recreation, and for fear (Musiani and Paquet 2004). However, the majority of wolf-human conflicts stemmed from agricultural and pastoral practices, whereby wolf depredation of domesticated livestock instigated eradication measures (Musiani and Paquet 2004, Mech 1995, DeBlieu 1991).

Red wolves (*Canis rufus*), in particular, exemplify the fervor of persecution in the United States. As human populations expanded into the wildlands, the wildlands and the red wolf disappeared. DeBlieu (1991) outlined the systematic extermination of the red wolf from the southeastern seaboard and from Alabama, Mississippi, Tennessee, Kentucky, Missouri, Oklahoma, and Arkansas. Two subspecies, *Canis rufus floridanus* and *Canis rufus gregoryi*, became extinct. The remaining subspecies, *Canis rufus rufus*, found refuge in the swamps of Texas and Louisiana, where parasitic infestation and hybridization with coyotes further compromised the population of the species (Philips, Henry and Kelly 2003).

By 1967, the United States listed *Canis rufus* as an endangered species, and the USFWS determined recovery for the red wolf would only occur in a captive breeding program (Phillips, Henry and Kelly 2003, DeBlieu 1991, USFWS 2007 report). The USFWS removed 400 red wolves and declared the species extinct in the wild. Of the 400, only 43 were believed to be red wolves, and only 14 were considered pure enough to enter into the breeding stock for the recovery program (Phillips, Henry and Kelly 2003, Brownlow 1996).

The red wolf recovery program has proved successful. At present 208 red wolves exist in captivity and nearly 130 red wolves, distributed in 20 packs (Phillips, Henry and Kelly 2003), exist in North Carolina (USFWS report 2007).
2.1.2 Red Wolf Biology and Behavior

Red wolves, intermediate in size between the gray wolf (*Canis lupus*) and the coyote (*Canis latrans*), weigh on average between 45-80 pounds (USFWS), or 20-36 kg in metric scale (Roth, Murray and Steury 2008), with a body height of 66 cm and length of 1.2 m (USFWS). Disproportionately large ears and long legs distinguish the red wolf from the gray wolf and coyote (Phillips, Henry and Kelly 2003, Phillips and Henry 1992). Despite the characteristic red color (USFWS 2008), the red wolf (Fig.1) displays a range of fur colors: red, brown, even a black phase, but most often a mixture of gray, black and cinnamon-buff (Phillips, Henry and Kelly 2003).

![Figure 1: Red Wolf. Photo taken from the National Audubon Society Field Guide to North American Mammals](image)

Like the gray wolf, the red wolf lives as a family pack consisting of an alpha breeding pair and their offspring (Phillips, Henry and Kelly 2003, Phillips and Henry 1992), typically with five to eight animals (USFWS 2008). A hierarchal social structure
exists within the pack: the alpha pair leaders followed by the second-ranked beta wolf, young subordinates, juveniles and pups, and the omega wolf, the scapegoat (Greeley 1996). A wolf’s place in the echelon dictates its behavior. Upon greeting, for instance, a subordinate wolf shows submission, respect and affection to the dominate wolf by lowering its body and licking or nuzzling the other’s muzzle. Wolves communicate through these non-verbal displays and others as well as a range of vocalizations, including howls, barks, snarls, growls, and whimpers. Success of the pack depends on their communication and cooperation.

For wolves, cooperative strategies during hunts enable the smaller predator to take down the larger prey such as the white-tailed deer. Scat analysis from the Alligator River National Wildlife Refuge (ARNWR) revealed white-tailed deer, raccoon, and marsh rabbits constituted 88.7% of the biomass consumed by the red wolf (Phillips 1995). Rodent consumption, preferred by juveniles, decreased with age (Phillips, Henry and Kelly 2003). In ARNWR, food habits differed among the packs. For example, the Milltail pack (pack names coincided with their habitat) relied on small prey (rodents and rabbits), and the Gator pack relied on larger prey (deer and raccoons) (Phillips, Henry and Kelly 2003). Variances stemmed from prey abundance and distribution. In the agricultural fields used by the Milltail pack, rodents and rabbits occurred in higher densities than within the wooded and gum swamp habitats used by the Gator pack (Phillips, Henry and Kelly 2003). Phillips, Henry and Kelly (2003) suggest the “differential use of prey may have played a role in determining their home range sizes.” For the red wolf, home range sizes spanned 25 km² to 130 km², averaging 88.5 ± 18.3 SD km² for individuals and 123.4 ± 53.5 SD km² for packs (Phillips, Henry and Kelly 2003).
Wolves in the pack may disperse at the age of sexual maturity between the months of September and March, with 72% occurring between November and February (Phillips, Henry and Kelly 2003). Only the alpha pair in the pack breeds, annually producing one litter of 1-5 pups after a 63-day gestation period. From mid-April to mid-July, red wolves utilize dens, whether underground burrows, hollow logs, ditches or windrow nests in agricultural fields (Phillips, Henry and Kelly 2003). Phillips (1995) documented the 1994 survivorship of wild-born red wolf pups. Of the 66 pups, 54% were free-ranging, 8% were returned to captivity, 23% have fates unknown and 15% died. Researchers discovered wild-born pups survived for longer periods than the released adults and captive-born pups.

2.1.3 Ecological Benefits of Red Wolf

The extirpation wolves allowed the expansion of coyotes east of the Rocky Mountains. As an apex predator, the organism at the highest trophic level, the wolf theoretically exerts top-down control on the ecosystem, creating trophic cascades, chain reactions across multiple trophic levels (Estes, Crooks and Holt 2001). Removal of the apex predator results in a proliferation of the mesopredator, the small to mid-sized predator such as the coyote, skunk (*Mephitis mephitis*) and raccoon (*Procyon lotor*), which, as a cascading result, reduces the populations of smaller vertebrate species such as rodents, birds and lagomorphs (Estes, Crooks and Holt 2001; Ripple and Beschta 2004; Côté et al. 2004). Moreover, “the absence of highly interactive carnivore species such as wolves can thus lead to simplified or degraded ecosystems” (Ripple and Beschta 2004:755).
Research evinces an association between apex predators and higher biodiversity (Sergio et al. 2006; Ripple and Beschta 2004). Biodiversity describes the quantity and quality of species richness within geographic regions and the variability between species and ecosystems. The loss of species results in the loss of biodiversity. With decreased biodiversity, the ecosystem that functions on the interdependence of all its components may suffer.

Wolves prey upon ungulates, herbivores that influence plant communities. Ungulate herbivory can affect vegetation structure, composition, diversity, productivity and succession, and can alter the habitat quality for other animals (Ripple and Beschta 2004). Without predation pressures, ungulates, such as the white-tailed deer (*Odocoileus virginianus*), may experience population explosions, sometimes beyond the carrying capacity of a habitat’s available forage (Côté et al. 2004) and, through their intense browsing, reduce the species abundance and diversity of seedlings and saplings, and shift herbaceous and shrub community structures to grasses and ferns (DeCalesta 1997). Biodiversity declines in three communities—tree, herb and shrub, and songbirds—due to the overabundance of deer (DeCalesta 1997).

In Isle Royale, balsam firs flourished with an increase in wolves and a decrease in moose (Smith et al 2003). In Yellowstone, Ripple et al (2001) found taller aspens in the high wolf-use riparian zones and wet meadows and more “vigorous” regeneration along wolf trails and other heavy use areas than in low wolf-use areas. They hypothesized that the elk changed its foraging behavior by adopting an “anti-predator strategy,” staying in open country for vigilance and safety. The presence of wolves instigated modifications
in the elk’s diet, temporal feeding patterns and spatial use, and selection of habitat patches and foraging sites. In the absence of elk, aspen regenerated.

Top predators influence lower trophic levels by direct effects on mortality and indirect effects on behavior modifications (Lima 1998; Estes, Crooks and Holt 2001; Ripple et al. 2001; Ripple and Beschta 2004). Ripple and Beschta (2004) elaborated on the concepts behind behavioral changes in response to predation: optimal foraging theory, “ecology of fear” concept, “predation sensitive food” hypothesis, prey and plant refugia and “terrain fear factor.” Optimal foraging theory describes the foraging strategy organisms adopt to maximize returns in the minimum amount of time and to balance the need for food and safety. In any given habitat, herbivores’ selection and use of space manifests from their fear of predation, the foundation for the “ecology of fear” concept. Despite fears, prey take greater risks to forage as food availability dwindles and occupy riskier sites as defined by the “predation sensitive food” hypothesis. As predation risk increases, the rate of mortality increases, thereby limiting prey population size. Prey refugia, on the other hand, are sites of prey occupation where the risk of predation is minimized. Ungulate migration outside the core wolf areas increases survival potential, but within the high wolf density areas, herbivory decreases, creating plant refugia for plants that escaped browsing. In areas with high wolf density and predation risk, prey may avoid these altogether or forage less intensely. Wolves have a higher kill rate if they can approach prey without detection or with an element of surprise, and, in response, prey develop strategies to either hide from predators by seeking forest cover or to spot predators from afar by seeking open terrain. This strategy model of space-use and forage-patterns based on features of the landscape refers to the “terrain fear factor.”
Environmental variables such as terrain, elevation, habitat structure, snowpack, weather or wildfires influence predator and prey behaviors, predation risks and survival rates (Ripple and Beschta 2004). In response to the dynamics in the environment and changing predation risks, prey possess adaptive flexibility in behavior, which in turn changes the foraging patterns and environmental effects (Lima 1998).

The science of ecology has undergone a succession of paradigms on the nature and importance of species interactions, including those between predators and their prey. (Estes, Crooks and Holt 2001: 858)

Ecologists recognize important species interactions follow the three successive pathways, often simultaneously and interactively: bottom-up forces where primary production regulates populations; competitionism where lateral forces within trophic levels regulate populations; and top-down forces where apex predators regulate populations (Côté et al. 2004). Like a complex machine with various interconnected parts working simultaneously with and against the other, the ecosystem hypothetically functions best with all parts present.

2.1.4 Economic Benefits of Red Wolf

Urban development, agriculture and resource extraction threaten to erase the natural face of the planet, and, within remaining pockets, fragmentation may cut the flow of animal and genetic movement. If habitat destruction does not cease, large carnivores will cease to exist, except in zoos (Clark et al 1996). Unfortunately, economy drives human expansion, and people assume a strict dichotomy exists between people and carnivores, between the economy and environment, and that conservation favors carnivores and the environment over people and their economies (Rasker and Hackman...
Communities, therefore, resist conservation efforts and predator persecution persists. It took 25 years to reintroduce the grey wolf into Yellowstone National Park (Weber and Rabinowitz 1996), for citizens believed wolves would harm not only the livestock industry but the states’ economies overall (Rasker and Hackman 1996).

In order to convince the public predators do not adversely affect economy, Weber and Rabinowitz (1996) and Rasker and Hackman (1996) compared the economies of wilderness and resource management counties. The research teams found wilderness counties experienced greater income and economic growth than the counties dependent on resource extraction. As shown by Rasker and Hackman (1996), between the years 1969-1992, employment and personal income increased by 93% (34,500 new jobs) and 89% ($987 million), respectively, in the wilderness counties of northwest Montana, while the resource-extractive counties in northwest Montana only increased 15% in employment (2152 new jobs) and 19% in personal income ($70 million). Furthermore, the resource-extractive counties lost more than 1300 jobs in construction, transportation and public utilities sectors, and, therefore, had higher unemployment rates than the wilderness counties (Rasker and Hackman 1996).

Yellowstone National Park serves as a prime case study for the economic impacts of carnivore, in particular wolf, conservation. Prior to wolf reintroduction, opponents expressed concern about wolves having a negative impact on big game populations and, in turn, on the local and regional economies driven by hunting revenues (Duffield, Neher and Patterson 2008). Proponents countered with a prediction that the attraction of wolves would boost park visitation and regional economies (Duffield, Neher and Patterson 2008). Investigating the economic impacts of wolf recovery, Duffield, Neher and
Patterson (2008) estimated 94,000 visitors from outside the three-state region came to Yellowstone specifically for the gray wolf and spent on average $375 per person, or a total of $35.5 million (a total, adjusted for inflation, less than the predicted revenue). Ranchers, on the other hand, experienced annual depredation losses averaging $63,818, twice the high-end of Environmental Impact Statement (EIS) predicted estimates, in 2004 and 2005. However, the EIS based their assumptions on wolf populations numbering 100, not the actual 300. The EIS had in addition projected losses of $342,000 to $890,000 in hunter expenditures; yet, no reductions for permits, numbers of harvested animals or hunter success for mule deer or moose occurred because of wolf restoration. While populations of mule deer, bison and moose had not declined because of wolf predation, herd sizes of elk had diminished as numbers of wolves increased. The number of elk permits dropped “substantially” because of wolf presence, aggressive culling policies and drought. Overall, Duffield, Neher and Patterson (2008) concluded wolf recovery netted an estimated $58 million in positive gains for the region in 2005.

In 2005, livestock predation by wolves accounted for 0.11% of cattle losses, whereas coyotes killed 22 times more cattle than wolves, dogs killed 5 times more cattle than wolves and vultures killed twice as many cattle as wolves (Defenders of Wildlife 2009). Cattle losses by theft were 5 times higher than cattle losses by wolves (Defenders of Wildlife 2009). Disease ranked highest for cause of cattle loss.

Deer, especially in high population densities, transmit diseases such as bovine-virus-diarrhea to livestock and Lyme disease to humans (Côté et al. 2004). Disease prevalence increases with high density of host species because of increased transmission rates (Stronen et al. 2006). Wolves could reduce risks of transmissions, if diseases
increase prey vulnerability (Barber-Meyer, White and Mech 2007), by reducing average group sizes of herd species (Stronen et al. 2006). After wolf restoration in Yellowstone, bovine-virus-diarrhea in elk decreased “substantially” as compared to prior values (Barber-Meyer, White and Mech 2007). Wolves therefore may play a positive role in agricultural landscapes. Wolf regulation of wild ungulate density may reduce disease prevalence and indirectly lower transmissions to livestock, a cost-value few ranchers and farmers recognize but from which they may benefit (Stronen et al. 2006).

The commercial losses from nursery, crop and ornamental garden damage by deer browsing and deer-vehicular collisions surpass the commercial losses incurred by wolves (Côté et al. 2004). Côté et al. (2004) reported in 1991 deer caused $351 million in damages to agriculture and households and $1 billion in accident-costs. By controlling deer populations, wolves may mitigate commercial as well ecological damages.

Defenders of Wildlife issued a report on wolf ecotourism in 2005. In Yellowstone, 200 visitors paid $1700 weekly for Safari Yellowstone and the opportunity for gray wolf watching. A study on red wolves in North Carolina’s ARNWR indicated a possible 19% increase in the region’s tourism, which may bring in 25,000 visitors and $37.5 million to Eastern North Carolina. In 2005, 900 visitors participated in the red wolf howling safaris.

In 2006, the Defenders of Wildlife hosted a stakeholders’ meeting to discuss strategies for developing and managing red wolf ecotourism in North Carolina. Plans included construction of a wolf education center and express hotel, providing other recreational opportunities and revenue from canoe, kayak and bike rentals, and generating income from the merchandise and the howling tours. At present howling
tours, which on average attract 1,000 participants annually, charge a nominal fee of $5 per person. Approximately 10,000 visitors toured the Red Wolf Coalition-sponsored 2007 exhibit in Colombia, North Carolina (Red Wolf Coalition 2008). The potential economic value of red wolves has yet to come to fruition, but, with appropriate planning, individuals, local communities and regional organizations may benefit along with the red wolf.

2.2 Red Wolf Recovery Program

2.2.1 The Alligator River Wildlife Refuge (ARNWR)

As outlined in the Red Wolf Species Survival Plan (RWSSP), the goals include increasing the number of genetically pure red wolves in captivity, maintaining a viable gene pool and reestablishing the captive species in the wild. Genetic viability, as surmised by biologists, requires an estimated 330 red wolves in captivity and restoring at least 220 wolves in the wild at three or more sites (Phillips, Henry and Kelly 2003).

In 1987, the USFWS released the first wolves and by 1995 reintroduced 63 captive-born wolves into the 18,218ha ARNWR in the North Carolina outer banks (Phillips 1995). The second reintroduction site in the Great Smoky Mountains National Park (GSMNP), however, failed because of the high pup mortality (USFWS 2007). Biologists identified parvovirus in one litter of pups and evidence of coyote predation on another, as well as heavy intestinal infestation and malnutrition (USFWS 2007).

ARNWR proved ideal for red wolf reintroduction with 48,562 ha of coastal plains, abundant prey, lack of coyotes, few livestock, few human settlements, adjacency to 20,639 ha of undeveloped land held by the Department of Defense and water barriers
as a natural enclosure to facilitate management of the wild populations (Phillips, Henry and Kelly 2003). An additional ideal restoration site, Pocosin Lakes, contained 45515 ha, abundant prey, only small populations of coyotes and livestock (Phillips, Henry and Kelly 2003). DeBlieu (1991) and Phillips’ field journal described the first wolf releases, replete with every nuance of the process, atmosphere and mood. As a general rule, biologists released the wolves in family groups or adult pairs between August and October (71%) (Phillips 1995). Slow-releases, where the captive-bred wolves stayed in pens and relied on biologists for food for a period of time, helped the red wolf acclimate to its new wild surroundings.

Statistics showed successful releases as only 21% (Phillips, Henry and Kelly 2003). In general, 70% of the unsuccessfully released wolves that travelled far from the release area either died or were returned to captivity (Phillips, Henry and Kelly 2003). Altogether 36 captive-born red wolves died within a year from release (Phillips, Henry and Kelly 2003). The main causes of wolf mortality in North Carolina arose from vehicular accidents (30%), malnutrition and parasitism (27%) and intraspecific aggression (12%). Since 1999, the loss of red wolves from human factors included gunshot (22%), vehicle (14%), poison (3%), traps (2%) and management (13%) (USFWS 2007). Disappearance (22%) counts as another high percentage for loss, in addition to the natural causes mentioned previously.

Field data from ARNWR, though, revealed upward trends in the populations of red wolves, the number of breeding pairs and number of pups born (USFWS 2007). Preliminary USFWS (2007) analysis estimates red wolf survival rates in the wild as 78.2% overall, with adults at 80.6%, yearlings at 79.3%, and pups at 67.8%.
However, the objectives of the RWSSP, according to the 2007 Five-Year Progress Report (USFWS), remain incomplete. The USFWS still requires at least two additional reintroduction sites within the species’ historic ranges to support viable populations of red wolves. This thesis research aims to contribute to the objective of the RWSSP by identifying and evaluating potential sites within the Daniel Boone National Forest for the reestablishment of red wolves.

2.2.2 Current Challenges of Red Wolf Conservation

Red wolf recovery hinges upon the mitigation or elimination of deterministic and stochastic threats. In the five-year report, the USFWS (2007) through five-factor analysis determined that habitat fragmentation and modification, disease, interspecific aggression, hybridization with coyotes, and human-induced mortalities continue to threaten the recovery efforts.

2.2.3 Impacts of Humans on Red Wolf Recovery

During hunting season, according to the USFWS 2007 report, the red wolf in ARNWR faces a 7.2 times greater risk of dying from gunshot. Since 2004, gunshot mortality (whether illegal or accidental) has impeded the upward population trends. Indirect effects from gunshot mortality exacerbate problems of red wolf recovery by reducing the wolf population, which reduces their ability to defend territory against coyotes and increases the chance of interbreeding. Gunshots, along with vehicular strikes and disappearance, constitute the leading causes of red wolf loss. In general, human activity accounts for more than half (58%) of the losses.
Lynn (2002) explained two features of man’s attitude toward the wolf that hinders its recovery and threatens its survival: wolves and humans should not share the same space; wolves do not belong in the “humanized” landscapes. Fear of the wolf drives the antagonism between man and wolf. In rural communities, people fear wolves will prey upon livestock and pets, and prevent logging, mining and hunting opportunities by excluding human presence in the forest, putting lives and livelihoods at risk (Musiani and Paquet 2004; Ratti et al. 1999; Mech 1995; Stronen et al. 2006).

Age, occupation, and education influence values. Surveys found older, less educated citizens perceive predators as dangerous or damaging to economic welfare (Ratti et al. 1999). In rural agricultural areas, wolf survival is “disproportionately dependent on actions of people who depend on the productivity of the landscape for their livelihood” (Stronen et al. 2006: 2). Anti-wolf sentiment threatens to push wolves into remote areas devoid of all human activity (Mech 1995).

Phillips (1995) concluded red wolf recovery depends, not on partitioning undisturbed wildlands, but “overcoming the political, emotional and logistical obstacles to human coexistence with wild wolves.” History as precedent proves humans handle wolf conflicts by eradication. Resolving the conflict then depends on our values, not facts alone, and our tolerance of wolves and acceptance of coexistence with wolves, not merely a science of wildlife management (Lynn, 2002). According to Saunders (2003), the caring relationship with nature relates to the “formation of an environmental ethic.” Attitudes reflect the positive, negative, or neutral tendencies to think, feel, or act toward a particular object, place, event or person, and, in regards to an environmental ethic, attitudes dictate “how humans behave in nature and how humans care about/value nature”
Beliefs and perceptions predetermine tolerance to wolves (Stronen et al. 2006). Public education on wolf ecology and management is vital to wolf recovery (Musiani and Paquet 2004; Fritts et al. 1997; Mech 1995). Moreover, teaching empathy, rather than focusing solely on the predator’s prominent role in the ecosystem, may be more effective in changing attitudes and, thus, environmental behavior (Prokop and Krubiatko 2008).

Lack of community support undermined the initial red wolf recovery efforts in the Land Between the Lakes, a region between Kentucky and Tennessee, because officials failed to address local concerns about red wolf conflicts (Ratti et al. 1999). Whereas flexible, non-restrictive regulations enabled greater support in North Carolina, for landowners and hunters felt the government prioritized their needs (Phillips et al. 2003; Phillips 1995). Prior discussions with locals about wolf management and regulations will work to alleviate concerns. Where pressure exists to manage wolf populations, non-lethal and non-traditional methods, if cost effective, may diminish livestock depredation and benefit people and wolves in agricultural environments (Musiani and Paquet 2004); however, wolf control may call for lethal measures, despite protest from preservationists (Mech 1995).

The human factor plays an important aspect in wildlife reintroductions and should be part of the plan, not an after-thought.

2.2.4 Impacts of Coyotes on Red Wolf Recovery

The stochastic process of hybridization with the coyote (*Canis latrans*), whereby a unidirectional introgression of coyote genetic material into the red wolf genome occurs,
poses a serious threat: the possible extinction of the *Canis rufus* species (Brownlow 1996; Ratti et al. 1999). In areas with high wolf density, wolves and coyotes may coexist without hybridization (Ratti et al. 1999). In simulations performed by Roth, Murray and Steury (2008), higher numbers of coyotes, on the other hand, led to higher extinction rates for small founding red wolf populations. At present, coyotes occupy historic red wolf ranges (van Manen, Crawford and Clark 2000; Phillips et al. 2003), and the likelihood of hybridization is high, especially considering the low populations of red wolves in the wild. Murray and Waits (2007) noted the expanding coyote and the declining red wolf populations create a “disequilibrium and hybrid zone expansion.”

Controversy surrounds this issue. Some researchers believe in preserving the genetic “purity” of the red wolf, while others would rather conserve the evolutionary process of speciation, whereby hybridization produces a canid more suited to survival amongst humans and their “anthropogenically modified landscapes” (Kyle et al. 2008). Biologists even argue the genetic purity and taxonomic status of the red wolf (Wayne and Jenks 1991; Nowak 1992; Phillips and Henry 1992; Brownlow 1996; Phillips et al. 2003; Murray and Waits 2007; Kyle et al. 2008). Nowak (1992:594) described the red wolf as “an intermediate stage in the course of wolf evolution from a small coyote-like ancestor to the modern gray wolf.” Researchers hypothesized the red wolf derived from gray wolf (*Canis lupus*) and coyote interbreeding. Yet, hybridizations between gray wolves and coyotes in the northern United States and Canada do not produce red wolves (Phillips and Henry 1992). Other authorities believe the red wolf may be a subspecies of the gray wolf; however, Kyle et al. (2008) contended the red wolf was endemic to North America before gray wolf colonization. Interestingly, Kyle et al. (2008) proposed conspecificity
between the red wolf and the eastern wolf (*Canis lycaon*), adding yet another dimension to the taxonomic debate. Regardless of the red wolf’s status, the Endangered Species Act (ESA) extends protection to species, subspecies and even species with “limited genetic introgression from another species” (Nowak 1992).

Minimizing coyote hybridization currently involves removing coyotes from red wolf habitat and sterilizing or euthanizing coyotes and/or hybrids (Phillips et al. 2003; Murray and Waits 2007; Kyle et al. 2008; Roth, Murray and Steury 2008). With sufficient decrease in coyote reproductive rates, red wolf extinction rates decreased to zero in simulations (Roth, Murray and Steury 2008). Red wolves establish home range territories more quickly with absent or low-density coyote populations (van Manen, Crawford and Clark 2000). In addition, releasing red wolves as family packs may provide the stable social structure to withstand possible coyote introgressions (van Manen, Crawford and Clark 2000). If red wolf populations recover despite the threat of hybridization, the recovery program will act as a model for other wildlife restorations because “invasive species will continue to shape ecological communities and conservation biology efforts” (Murray and Waits 2007).

Interspecific competition shapes ecological communities as well. Roth, Murray and Steury (2008) examined the sympatry between red wolves and coyotes, how the two competitors share space and resources. Wolves and coyotes form family groups with territorial home ranges with exclusive core areas and overlapping edges. Home ranges for the red wolf in North Carolina average 111 km², whereas the coyote’s home range average between 2 and 20 km². In southeastern Kentucky, telemetry data suggests an average coyote home range of 18.3 km² for a male-female dyad/pair plus pups (per}
personal communication with Dr. John J. Cox, UK Dept. of Forestry). With one breeding pair per wolf and coyote pack, dispersal rates of sexually-mature young adults tends to be high, especially in low-density populations. Coyotes, though, have a higher reproductive rate than red wolves. Eighty-percent of female coyotes produced an average litter of six pups, whereas fifty-three percent of female red wolves produced an average litter between three and four pups. With a smaller body mass, 2/3 the size of a red wolf, the coyote faces higher mortality rates from interspecific aggression, which influences the space-use patterns of the smaller canid.

Wolves dominate coyotes and force coyotes from certain areas (Arjo and Peltzcher 2004). In response, coyotes may shift habitats to lower quality sites, such as closer to roads and to human habitations, which wolves tend to avoid (Arjo and Peltzcher 2004). Roth, Murray and Steury (2008) suggested the impact of coyotes on red wolf recovery will lessen if red wolves can exclude coyotes from higher quality habitats. However, red wolves reintroduced into coyote-colonized landscapes may have difficulty establishing home ranges (Roth, Murray and Steury 2008).

Abundance of prey and intra-and interspecific aggression affect habitat selection for canids (Arjo and Peltzcher 2004). Prey tends to congregate in lowlands during the winter and thus leads to wolf and coyote sympatry (Arjo and Peltzcher 2004). Riparian environments provide thermal cover and security for deer and further habitat overlap (Arjo and Peltzcher 2004). Coyote use riparian habitats for microtines (voles and lemmings) and for scavenging wolf kills, which tend to occur along rivers (Arjo and Peltzcher 2004). In North Carolina, red wolves mainly consume deer, raccoon and rabbits. With no diet estimations for North Carolina coyote, biologists assume the canid
behaves as opportunistic, generalist predator, consuming a wider variety of foods, including rodents, lagomorphs, fruits, and carrion (Roth, Murray and Steury 2008). Red wolves may even supplement their competitors’ diet by providing carrion (Roth, Murray and Steury 2008). Therefore, the coyote probably exploits resources to a greater degree than the red wolf. Roth, Murray and Steury (2008) assume exploitative competition or territorial exclusion fuels interference behaviors, which in turn depends on the pack size and the quality of the habitat. The ability of the larger red wolf to exclude the coyote might enable coexistence; otherwise, higher coyote populations have a higher chance to hinder red wolf re-colonization (Roth, Murray and Steury 2008).

It appears critical to red wolf recovery to improve the understanding of interspecific relationships—canid sympatry, competition, interference and coexistence.

2.2.5 Principles of Carnivore Conservation

One of the primary goals of conservation biology is to preserve ecosystem dynamics and biodiversity, and conservation biologists spotlighted the top predators to achieve this aim rather than explain the complexities and benefits of thriving ecosystems. Sergio et al (2006) discussed several reasons why top predators promoted higher diversity: predators choose sites based on productivity, prey density, high topographic and habitat variety, and provide spatial refugia for some species, whether by deterring other predators or interguild competitors of that species. Moreover, apex predators signal disturbance and dysfunction in an environment. Carroll et al (2001) attributed large mammalian carnivores’ sensitivity to landscape changes because of their low population
densities, low fecundity rates, limited dispersal ability across open or developed habitat and “other traits that lower ecological resilience.”

Early conservation scientists centered on carnivores for reserve design based on the belief that carnivores made good indicator species for ecosystem health (Noss et al 1996). The ideal reserve consisted of an expansive, undisturbed core with surrounding, limited-use buffer zones. Yet, biologists know little about the viability of reserves (especially in the context of global climate change), whether 1,000-10,000 km² will work for decades or if 100,000 km² will work for the long-term (Noss et al 1996). The notion of a single reserve has since evolved into a vision of a network of reserves, allowing for the dynamic movement and natural migration of prey and predators. In developed regions, small highly managed reserves with unfixed corridors might prove more effective and economically feasible.

In time, biologists decided localized species of flora and fauna indicated ecosystem status better than long-ranging habitat generalists such as large carnivores. Conservationists needed a reason to conserve carnivores though, and, hence, directed attentions to their role as a keystone species. Keystone, as the term suggests, refers to the key stone, the central wedge of an arch for example, that holds the structure together. In ecology, a keystone species helps to support and balance the community of life, and if removed, it may lead to dramatic cascading effects across trophic levels. The impact of the keystone species is much greater than expected as compared to its relative abundance or total biomass, whereby a small change in the population densities of keystone species may cause major disruptions within the ecosystem. Paine (1969) defined a keystone predator as one that feeds preferentially on the dominant competitor prey species, and by
this, predation prevents the dominant prey from excluding other species. Therefore, a higher diversity of life is maintained in a system where the keystone predator exists than when it is absent.

Habitat modification and predator persecution have removed the keystones. Without the mega-carnivore, the “top-down” control, the community structure changes. For example, the opportunistic coyote increased after the extirpation of the wolf and, consequently, the kit foxes experienced 75-90% mortality from coyote predation (Linnell and Strand 2000). Songbirds have suffered from the loss of large carnivores as well (Noss et al 1996). The songbird profits from the presence of the apex predator, which limits the presence of the mesopredator that preys upon the songbird. In addition, large carnivores may reduce the abundance of large dominant herbivores, which allows the plant communities and smaller herbivores to diversify.

Conservation of keystone predators, in effect, creates an umbrella effect, whereby protections for the predator encompass all species sharing the same space. Therefore, the predator with the largest habitat assumes the role of the umbrella species (Carroll et al 2001), but whether a carnivore functions as an umbrella species depends on the biogeographical characteristics of a region (Noss et al 1996). Not all rare or endangered habitats fall under an umbrella’s range. Caro (2003) analyzed the use of umbrella species in East Africa and concluded the need to predict which background species populations would receive long-term protection from the umbrella species. As expected, background species have higher populations within East African reserves than outside the reserves, indicative of the success of umbrella species concept in conservation.
In addition, research shows a direct link between the manipulation of flagship species and efficacy of conservation based on top predators (Sergio et al 2006). Flagship species, unlike indicator, keystone and umbrella species, do not represent ecological roles or benefits. Conservation endeavors rely on fascination and sympathy for support. Organizations rally, “Save the wolves. Save the polar bear. Save the cheetah.” Instead of graphs and charts, breathtaking photographs of power and beauty win hearts, and the countless brutal images of mutilation, skins and bones portray the direness of the situation. Non-governmental organizations (NGOs) such as World Wildlife Fund or Cheetah Conservation Fund have championed conservation for these charismatic species. Although procured on glamour, NGOs allocate donations for habitat protection, restoration, local education, and research.

Regardless of nomenclature, carnivores have represented focal species in conservation. As Noss et al (1996) poignantly noted, “Carnivores inspire people more than fungi can.” Current trends, though, suggest a shift from species-oriented conservation to ecosystem focus (Linnell and Strand 2000).

Carroll et al (2001) advocated ecosystem-level regional planning along with analysis of species-habitat association and interactions among predators, which manifest as either exploitation or interference. Exploitation refers to one species consuming more resources than another, and interference refers to the direct death of a contender as a result of the interaction. In Africa, negative relationships exist between the densities of African spotted dogs and cheetahs with the densities of lions and hyenas (Linnell and Strand 2000). Wild dogs cannot keep hyenas and other large scavengers away from kills, which has slowed their recovery. Lion predation accounts for 73% of cheetah cub
mortality. In Namibian farmlands, cheetahs survive best outside the protected areas where predator control has removed lions and hyenas. Smaller species avoid habitats of dominant carnivores, keeping spatial distance, and can succeed in areas with lower prey abundance. Therefore, intraguild predation has certain implications for carnivore conservation, such as including the coyote factor into kit fox conservation plans (Linnell and Strand 2000).

Policy and laws have neglected interactive species and have relied more on “social and economic concerns than on biological information.” However, to maintain ecological integrity, wildlife reserve planners should subsume interspecific interactions into conservation goals (Soulé et al 2003). Despite fears in Yellowstone National Park, the reintroduction of the grey wolf has also restored the ecosystem. The elk herds avoid wolf territory, areas associated with high vulnerability and risk for the elk but with recovery for the aspen trees.

In the end, our success of rehabilitating an ecologically degraded world will be judged more on the persistence of interspecies interactions than on the geographically limited persistence of populations based only on causing the least economic burden and ensuring only symbolic survival. (Soulé et al 2003).

2.3. Potential Release Sites

2.3.1 Criteria for Red Wolf Habitat

Biologists consider wolves habitat generalists, for they are “able to live in areas where prey and shelter are sufficient, so long as habitat fragmentation, disturbance or harassment by humans are minimal or do not occur” (USFWS 2007). Gray wolves in
Northwestern Montana preferred mesic forests (Arjo and Peltscher 2004). In North Carolina, restored red wolves have occupied a mosaic of landscapes—wetlands, pine forests, upland shrubs, crop land, and pocosins (wetland forests with pine tree overstory and evergreen shrub understory) and have utilized edge interfaces for travel and prey access (USFWS 2007). Fuller (1995) suggested a clustering of 2-3 packs in an area of 322 km² for viable gray wolf populations.

Researchers have described various variables affecting suitable wolf habitat. In general, wolf density correlates with prey density (Oakleaf et al. 2006). Jędrzejewski et al. (2004) reported forest cover, fragmentation, road length and distance to border as the four major factors for wolf suitability in Poland. Shaffer (2007) specified seven criteria for suitable red wolf habitat: road density less than 0.25km/km²; 1.0 km from highways; 2.0 km from towns; land cover classes comprised of deciduous, evergreen or mixed forests, shrub/scrub, grassland, woody wetland and herbaceous wetland; deer density at least 5.0 deer/km²; slopes no greater than 20º; and patch areas at least 45.6 km².

As indexes for wolf suitability, researchers focus mainly on density of prey, roads, and humans. Mech (1995) denoted road density as the “yardstick” by which agencies and recovery teams measured wolf habitat suitability. Wolves do not have an aversion to roads and travel roadways with lower traffic volumes (Wydeven at al. 2001; Mladenoff et al. 2009). Only roads with moderate to heavy traffic pose problems for wolves due to increased risk for wolf mortality from vehicular accidents (Mladenoff and Sickley 1998). In North Carolina, during the years 1987-1994, motor vehicles caused 30% of the red wolf deaths (Phillips et al. 2003). Highways and major roads with frequent traffic, not only heighten risks, but form significant barriers to wildlife
movement within the forest, and subsequent fragmentation creates potentially small patchy habitats (Heilman et al. 2002).

Wydeven at al. (2001) traced the early development of road density as a landscape feature to predict suitable habitat. In 1952, Thompson, a graduate student of Aldo Leopold, warned that road development would cause the extirpation of wolves from northern Wisconsin, and, by 1960, his prediction proved true. Thiel (1985) calculated the road density of northern Wisconsin and discovered wolves disappeared when road densities exceeded 0.68 km/km². Researchers since consider the value of 0.6 km/km² the standard threshold for maintaining wolf populations. In 1995, Mladenoff et al. utilized GIS to analyze road densities within 14 wolf-pack territories in Wisconsin, finding highly suitable habitat with road densities less than 0.45 km/km² (mean 0.23 km/km²). Fuller et al. (1992) determined road densities less than 0.7 km/km² in wolf pack areas in Minnesota. Other researchers (Shelley and Anderson 1995; Harrison and Chapin 1998; Mladenoff and Sickley 1998; Mladenoff et al. 1999; Ratti et al. 1999; Corsi et al. 1999; Unger 1999; Frair 1999; Houts 1999; Kohn et al. 2000) incorporated road density in habitat studies and determined it the best predictor for suitable wolf habitat.

Mladenoff et al. (1999) tested road density within wolf habitats by defining six probability classes. In an area with road density values between 0.37 and 0.45 km/km² (50% probability class), the habitat has a 50-75% chance of supporting wolves. With this probability class index, Mladenoff and Sickley (1998) estimated 52,804 km² of potential habitat in western and eastern Maine and northern New Hampshire for eastern wolf recovery (Harrison and Chapin 1998).
Road density approximates human activity and the potential for human-caused mortality (Harrison and Chapin 1998; Mladenoff et al. 2009). Highest natural mortality for wolves occurred in habitats with road density values between 0.63 and 0.84 km/km² and highest human-induced mortality occurred in habitats with road densities between 0.84 and 1.14 km/km² (Wydeven at al. 2001). Some researchers combine road and human densities into the wolf habitat analysis. Wydeven et al. (2001) reported sustainable thresholds as road densities less than 0.7 km/km² and human densities less than 4 humans/km². In Wisconsin, human densities within wolf habitats averaged 1.52 humans/km² and within nonpack areas averaged 5.16 humans/km² (Harrison and Chapin 1998).

In mountainous terrain, Wydeven at al. (2001) suggested road density may not provide as useful an index because of “very patchy” ungulate distribution. Forested areas managed for wolves should maintain sustainable road densities (0.45km/km² at core wolf habitat and overall 0.6 km/km²) by closing logging roads after operations, avoiding increasing road density or changing traffic levels, and keeping land 100 m within den sites undeveloped (Wydeven at al. 2001). Wydeven at al. (2001) believed as human attitudes toward the wolf improve road densities may not factor as much into wolf habitat selection. Until then, researchers will continue to apply the “yardstick” measure of road densities to wolf habitat suitability studies.

2.3.2 Daniel Boone National Forest

Road analysis completed by the Daniel Boone Forest Office shows the Forest Development Road System consists of 2147.67 km of roads—717.768 km of arterial and
collector roads and 1429.10 km of local roads, 965.6 km of which the Forest Service maintains annually. Forest arterial roads serve as major access routes to and through large areas, and Forest Collector roads serve as connecting roads to smaller areas of land. Forest Local roads, generally of short length, dead-end at a terminating facility. Traffic on Forest Arterial and Forest Collector roads do not differ from types of traffic expected on public roads, whereas traffic on Forest Local roads is limited to specific users or activity. At present, however, traffic volumes need to be accessed and mapped.

DBNF has one inventoried roadless area within the 1,133 ha Wolfpen area adjacent to the 5044 ha Clifty Wilderness.

With 2.1 million proclamation acres, of which the National Forest Service (NFS) manages 27,479 ha, the DBNF (Fig. 2) in the Appalachian foothills in eastern Kentucky appears promising as a restoration site for red wolves.

![Figure 2: Map of the Daniel Boone National Forest](http://www.littlewolf.org/preserve/daniel-boone-national-forest/)

The loss of keystone species in combination with the loss of the dominant forest canopy species, the American chestnut (*Castanea dentata*), resulted in impoverished
biotic communities in Kentucky (Maehr, Grimes and Larkin 1999). By 1850, people had extirpated red wolf, timber wolf, black bear (Ursus americanus), mountain lion (Puma concolor), bison (Bison bison) and elk (Cervus elaphus), all large mammals save the white-tailed deer, leaving Kentucky with the fewest large mammals than any other state in the southern Appalachians (Maehr, Grimes and Larkin 1999). Maehr, Grimes and Larkin (1999) viewed restoration of the large mammals to historic ranges as a conservation priority. Kentucky, through reintroduction of native species, especially large mammals and particularly large carnivores, may recover the integrity of its ecosystems.

After an absence of 150 years, elk returned to southeastern Kentucky in 1997 (Maehr, Grimes and Larkin 1999) and the initial herds have grown upwards to 7,500, the project’s goal. Wildlife biologists selected the restoration zones based on the low human population and distance from row crops and major urban centers (Larkin et al. 2004) and the suitable habitat of forest, grasslands and shrublands for elk (Maehr, Grimes and Larkin 1999).

Larkin et al. (2004) provided a detailed description of the 14-county restoration zone for elk that covers 1.04 million ha of in the Cumberland Plateau. In eastern Kentucky, mountain-top coal removal has transformed the rugged topography of winding ridges, steep slopes and narrow valleys into “gently sloping grasslands.” Herbaceous vegetation in the strip-mined reclamation areas includes Kentucky 31-tall fescue (Festuca arundinacea), bush clover (Lespedeza spp.), perennial ryegrass (Lolium perenne) and orchard grass (Dactylis glomerata), which supply 573 kg of forage with the capacity to support 0.28-.083 domestic calf-cow units/ha. Secondary and tertiary growth forests
represent a mixed-mesophytic forest, which occur on moist, well-drained sites. American beech (*Fagus grandifolia*), yellow-poplar (*Liriodendron tulipifera*), basswood (*Tilia spp.*), sugar maple (*Acer saccharum*), northern red oak (*Quercus rubra*), white oak (*Quercus alba*), eastern hemlock (*Tsuga Canadensis*) and yellow buckeye (*Aesculus octandra*) exemplify several of the 30 overstory species contained within the restoration zone forests. On ridge tops, southwestern facing slopes and sites with shallow soils, Oak-Hickory (*Quercus-Carya*) and Oak-Pine (*Quercus-Pinus*) dominate. Common understory species include eastern redbud (*Cercis Canadensis*), flowering dogwood (*Cornus florida*), spicebush (*Lindera benzoin*) and pawpaw (*Asimina triloba*). The three release sites in the Cyprus-Amax Wildlife Management Area (Cyprus) in Perry County, the Orr mining property (Orr) in Harlan County and the Redbird Wildlife Management Area (Redbird) in Leslie County represent a landscape dominated by active and reclaimed-surface mines, the DBNF and private forests. Cyprus, a 7,400 ha coal mine, contains 1,000 ha of active mining, 2,000 ha of reclaimed grasslands and the remainder in deciduous forests. Closed to the public, Orr experiences little human activity except for mining of an active coal mine (3% of the property) and contains a mosaic of secondary and third growth forests (89%) and reclaimed grasslands (7%). Redbird comprises 9,200 ha of the DBNF with approximately 95% forest cover and less than 2.5 ha of open habitats and experiences the least impact from human activity and settlement of the two other release sites.

Situated in the eastern portion of the Cumberland Plateau, Redbird lies southeast of DBNF, the narrow 225.3 km strip of forest oriented north-south along the western edge of the Cumberland Plateau (Daniel Boone Forest Office). The Cumberland Plateau,
a tableland with relatively low relief, ranges in elevation from 200 m in the Ohio River floodplains to 1,200 m in the Black Mountains. Sandstone cliffs, bluffs, ridges, hills, natural bridges, waterfalls and winding narrow-valley streams “deeply” dissect the terrain, and the forest composition varies.

As topography varies, soil types vary. Soil scientists have classified 90 soils within the DBNF (USDA 2008); although, most of the soils are Udults, Ultisols characterized by strongly leached acidic soils with a uric moisture regime and by high forest productivity. The USDA generalizes the soil textures within the DBNF as deep, fine-loamy and fine-silty on alluvial terraces (4%), moderately deep to deep fine-silty and fine-clay on ridge tops and crests (23%), and moderately deep to deep fine-loamy, coarse-loamy and loamy-skeletal on gently sloping to steep sloping sites and on coves (73%). On shallow loamy-skeletal soils, tree growth averages 0.2832 m$^3$/0.4049 ha/yr, whilst on deep well-drained soils found on flood plains, terraces, benches, toe slopes and coves, tree growth ranges up to 3.77 m$^3$/0.4049 ha/yr. Upland hardwood species such as white oak, chestnut oak, northern red oak, black oak, scarlet oak, hickory and pine constitute 49% of the DBNF (USDA 2008). Cove hardwoods (24%) such as northern red oak, white oak, basswood, yellow poplar, hemlock, sugar maple, beech and occasional black walnut and black cherry, pine (15%) and mixed hardwood-pine (12%) constitute the remainder of the mixed mesophytic forest (USDA 2008). In addition to the woody understory described in the elk restoration zones, rhododendron or fern-ephemerals occupy moist sites and mountain laurel or blueberry-huckleberry occupies dry sites within the DBNF (USDA 2008).
According to the 30-year climate record, the humid-temperate DBNF region has an average annual temperature of 13ºC (mean winter temperature of 4ºC and mean summer temperature of 24ºC) with evenly distributed precipitation throughout the year averaging 117 cm annually (mean annual snowfall of 51 cm) (Larkin et al. 2004). Evaporation from ponds and lakes averages about 88.9 cm, 27.9 cm less than the annual precipitation (USDA 2008). The DBNF contains four Army Corp of Engineer reservoirs, Cave Run Lake, Buckhorn Lake, Lake Cumberland and Laurel River Lake, with 258 km² of water at normal pool level and 347 km² at maximum pool level (Daniel Boone Forest Office 2003). Intense rainfall activates the flows of many small ephemeral streams and cause flooding in areas with shallow soils and steep slopes (Daniel Boone Forest Office 2003). With 1931.2 km of branches and streams (USDA 2008), a dendritic drainage pattern forms from the combination of flat-topped ridges and rolling hills, and the DBNF crosses the drainage basins of three large rivers, the Cumberland River, Kentucky River, and Licking Rivers (Daniel Boone Forest Office 2003).

Brine disposal from oil and gas drilling and acid mind discharges reduce water quality in the streams (Daniel Boone Forest Office 2003). Riparian ecosystems provide quality aquatic and terrestrial habitats for a myriad of species, flood control, removal of sediments from surface run-off, ground water recharge, retardation of erosion, nutrient cycling and other vegetation, soil and hydrologic functions. In the DBNF, the Forest Service in cooperation with other agencies have established restoration projects and improved maintenance within the watershed to protect the riparian zones and water quality (USDA 2008).
Boating, fishing and water sports attract visitors to the lakes, rivers and streams. On an annual basis, 5 million tourists visit the DBNF for the scenery and the recreational activities such as boating, fishing, water sports, hunting, hiking, camping, rockclimbing and wildlife viewing (USDA 2008). The DBNF provides habitat for 23 endangered and threatened species and manages 5 wildlife areas: the 3739.7 ha pioneer Weapons Wildlife Management Area, the 5486.7 ha Mill Creek Wildlife Management Area, the 7020.1 ha Beaver Creek Wildlife Management Area, the 2700.1 ha Crane Creek Wildlife Management Area, and the 10331.2 ha Redbird Wildlife Management Area (USDA 2008).

van Manen et al. (2000) created prediction models for red wolf release successes in historic ranges in the southeastern United States. Three categories of independent variables potentially related to red wolf success included the biological condition of the release sites, the human influences on the release sites and the characteristics of the release. Red wolf population size, coyote density and large and medium prey densities accounted for the biological conditions, while human, road and livestock densities as well as the amount of agricultural land characterized the human influences of the release sites. For the third variable, the characteristics of the release, the researchers considered the age and sex of the red wolf, its previous environment, time in captivity, period of acclimation, group size and release season. In general, they found negative correlations between road, coyote and human densities and longer captive and acclimations periods with red wolf success. Survival improved with low human density, large prey density and by using wild-reared wolves and shorter acclimation periods. For the eastern Kentucky-northern Tennessee region, the composite index of red wolf release success resulted in 0.667.
(values greater than 0.8 indicated high success potential). If these models prove reliable, the release of red wolves in the DBNF (unknown whether included in or adjacent to study region) has the potential for greater than average success.

2.4 Methodological Approaches

2.4.1 Logistic Regression for Wolf Restoration

In order to predict the probability of wolf release success, van Manen et al. (2000) used a logistic regression model. Simply stated, logistic regression statistically analyzes the probability of an occurrence of an event, generally explaining the outcome of predictor variables (risk factors, for example) and a dichotomous dependent variable (success/failure or absence/presence).

Mladenoff et al. (1995) developed the logistic regression model for potential wolf habitat in the Northern Great Lakes region. The researchers applied five spatial landscape-scale variables: human population density, prey (deer) density, road density, land cover, and land ownership. Because of the intercorrelation of the variables associated with human land-use, Mladenoff et al. (1995) disregarded all but the road density variable. An additional variable, the fractal dimension, improved the logistic model’s performance. The fractal dimension, denoted as $D$, represents an index of land cover patch boundary complexity, basically a rough or fragmented geometric shape. In the study, wolves inhabited simpler patch shapes than those within the nonpack areas, indicative perhaps of the wolf’s preference for habitats with less fragmentation.. Radio-collared wolves provided information about movement through the landscape and enabled the researchers to verify habitat locations and types. In the Great Lakes region,
pack areas had greater proportions of mixed conifer-hardwood, forested wetlands and public lands and lower proportions of agricultural land, deciduous forests and large lakes. Mladenoff et al. (1995) incorporated the favorable habitat into the logistic model to predict and map wolf distribution into new territory using GIS and derived two model equations:

1. \( \text{logit}(p) = -6.5988 + 14.6189R \)

where road density is calculated by dividing the road length within an area by the cell area.

2. \( \text{logit}(p) = -49.550 + 19.854R + 26.861D \)

where \( p \) represents the probability of wolf occurrence, \( R \) the road density, \( D \) the fractal dimension index.

Researchers have since applied this approach for wolf-habitat predictions (Mladenoff et al. 1995; Mladenoff and Sickley 1998; Mladenoff, Sickley and Wydeven 1999; Brito, Crespo and Paulo 1999; Ratti et al. 1999; Glenz et al. 2001; Houts 2003; Keating and Cherry 2004).

While popular, the logistic regression for habitat modeling carries the risk of misapplication based on inadequate understanding concerning the model, its interpretations, and sampling design (Keating and Cherry 2004). Brito, Crespo and Paulo (1999) compared logistic regression to overlap analysis and determined logistic regression more “elucidative and precise,” however, presence/absence data (generally based on telemetry data) present accuracy problems. Researchers may misinterpret non-use as unsuitable habitat for a particular species, whose actual presence may go undetected by deficient sampling techniques. Ratti et al. (1999) noted other limitations to
the logistic model. The model’s small sample of wolf packs makes “comparisons to other areas and wolf packs tenuous.” With wolf density below carrying capacity, recolonizing wolves select optimal habitats, whereby less-optimal but suitable habitat remains unoccupied and available. In addition, the relative homogeneity of the land cover types and ungulate density of the study area may render the model inappropriate for diverse areas. Moreover, logistic regression predictive models have failed to predict wolf re-colonization in Wisconsin “apparently because it failed to consider the adaptability of wolves” (Mech 2006).

Mladenoff et al. (2009) revisited their logistic model predictions for the Great Lakes region and found wolves, in reality, had begun to occupy the lowest quality habitats but at very low rates. In an attempt to better describe wolf occupancy, Mladenoff et al. (2009) created a new model, which incorporated landscape composition. They found that the presence of wolf negatively correlated with road density and agricultural lands and positively correlated with mixed forests. By introducing agricultural land use, the performance of the logistic regression model increased. This model is expressed as

\[ \text{Logit (} p \text{)} = 5.0018 - 11.7095A - 2.5655R, \]

where \( p \) represents the probability of wolf presence, \( A \) the percentage of agricultural lands and \( R \) the road density (km/km²). In summary, Mladenoff et al. (2009) concluded their original 1995 model is “the best conservative indicator of preferred, most-critical habitat” and the new model should be seen as “more descriptive of wolf occupancy, and more dependent on complex population dynamics, than [the] simpler 1995 model.”
Chapter Three
Methodology

3.1 Study Area

With 849,841.3 proclamation hectares, of which the National Forest Service (NFS) manages 282,310 ha, the DBNF (Fig.3) in the Appalachian foothills in eastern Kentucky appears promising as a restoration site for red wolves.

Abundant prey, mainly contiguous mixed-mesophytic forest and successful elk reintroduction factored into site selection within historic red wolf range. The DBNF remains one of the few national forests remaining in the southeast region of the United States with sufficient wildlands area for wolf occupation (Fig. 4). Other sizable national forests in the southeastern region that warrant habitat analysis include the Ozark National....
Forest in Arkansas, Shenandoah National Park in Virginia, and Nantahala National Forest in North Carolina, which lies south of the Great Smoky Mountains National Park where red wolf reintroduction failed. However, this analysis settled on the DBNF for its nearby location to the University of Cincinnati.

Figure 4: National Forest Land of the Southeast United States. Map courtesy of the USDA Forest Service. (http://www.fs.fed.us/r8/sitemap.php)

White-tailed deer populations within the state number well over a million, and a gross estimated deer density, extrapolated from DBNF deer harvest data (Ratti et al. 1999), of 10 deer/km² exists within the DBNF. Requiring a minimum of 5 deer/km² (Shaffer 2007), red wolf populations would not be limited by prey density. The estimate, while regarded with caution, may underestimate the deer population and density.

Hunting in Kentucky is regulated by zones, and, in the DBNF counties and state park, zone 3 and zone 4 regulations apply and place more restrictions on the amount of deer harvest.

Overpopulation, rather, has increased susceptibility to disease. In 2007, the lethal epizootic hemorrhagic disease, or “bluetongue,” affected the deer in Kentucky, which the
USFWS called the “worst outbreak in 30 years” (USFWS report 2007). The USFWS cautioned hunters from consuming meat from emaciated or weak deer in case of secondary infections. The restoration of the red wolf may mitigate disease within the deer populations by reducing the number of deer more effectively than managed culling. With a reduction in deer population, the incidences of car-deer collisions (Fig. 5) and crop-ornamental garden damage may decrease as an additional benefit.

![Annual DBNF County Deer Strikes](image)

Figure 5: Annual Daniel Boone National Forest County Deer Strikes
Data source: [http://www.kentuckystatepolice.ky.gov/deerauto.htm](http://www.kentuckystatepolice.ky.gov/deerauto.htm)
Kentucky State Police

The 21 Kentucky counties containing portions of the DBNF include Bath, Clay, Estill, Harlan, Jackson, Knox, Laurel, Lee, Leslie, McCreary, Menifee, Morgan, Owsley, Perry, Powell, Pulaski, Rockcastle, Rowan, Wayne, Whitley and Wolfe (Fig. 6). Table 1 synthesizes the county information.
Figure 6: Daniel Boone National Forest County Map (courtesy of http://www.fs.fed.us/r8/boone/aboutus/map.shtml)
## Table 1: Daniel Boone National Forest County Demographics

<table>
<thead>
<tr>
<th>County</th>
<th>Population</th>
<th>Size (acres)</th>
<th>Pop. Density (per km²)</th>
<th>2008 Deer Harvest</th>
<th>Forest Acres</th>
<th>% Total Acreage</th>
<th>2007 Farms (acres)</th>
<th>2008 Head of Cattle</th>
<th>Market % Livestock</th>
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</thead>
<tbody>
<tr>
<td>Bath</td>
<td>11,750</td>
<td>178,854</td>
<td>16</td>
<td>522</td>
<td>19,300</td>
<td>10.8</td>
<td>129,057</td>
<td>24,700</td>
<td>58</td>
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<tr>
<td>Clay</td>
<td>23,930</td>
<td>301,440</td>
<td>20</td>
<td>585</td>
<td>77,946</td>
<td>25.9</td>
<td>51,194</td>
<td>3,000</td>
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<tr>
<td>Estill</td>
<td>14,948</td>
<td>162,560</td>
<td>23</td>
<td>586</td>
<td>5598</td>
<td>3.4</td>
<td>64,780</td>
<td>8,000</td>
<td>73</td>
</tr>
<tr>
<td>Harlan</td>
<td>30,783</td>
<td>298,880</td>
<td>25</td>
<td>347</td>
<td>803</td>
<td>0.3</td>
<td>3,034</td>
<td>155</td>
<td>45</td>
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<tr>
<td>Jackson</td>
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<td>221,440</td>
<td>15</td>
<td>445</td>
<td>58,601</td>
<td>26.5</td>
<td>82,614</td>
<td>10,500</td>
<td>69</td>
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<tr>
<td>Knox</td>
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<td>248,320</td>
<td>17</td>
<td>845</td>
<td>74</td>
<td>0.03</td>
<td>51,115</td>
<td>3,900</td>
<td>74</td>
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<tr>
<td>Laurel</td>
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<td>279,040</td>
<td>51</td>
<td>846</td>
<td>64,255</td>
<td>23.0</td>
<td>102,489</td>
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<tr>
<td>Lee</td>
<td>7,414</td>
<td>134,400</td>
<td>14</td>
<td>484</td>
<td>8587</td>
<td>6.4</td>
<td>29,419</td>
<td>2,100</td>
<td>50</td>
</tr>
<tr>
<td>Leslie</td>
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<td>258,560</td>
<td>11</td>
<td>282</td>
<td>52,142</td>
<td>20.2</td>
<td>5,642</td>
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<tr>
<td>McCreary</td>
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<td>273,920</td>
<td>16</td>
<td>441</td>
<td>142,669</td>
<td>52.1</td>
<td>15,056</td>
<td>2,100</td>
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<tr>
<td>Menifee</td>
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<td>13</td>
<td>415</td>
<td>46,857</td>
<td>35.9</td>
<td>43,110</td>
<td>3,800</td>
<td>65</td>
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<tr>
<td>Morgan</td>
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<td>1104</td>
<td>13,089</td>
<td>5.4</td>
<td>136,303</td>
<td>12,225</td>
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<tr>
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<td>206</td>
<td>16,570</td>
<td>13.1</td>
<td>35,857</td>
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<td>340</td>
<td>2151</td>
<td>0.98</td>
<td>10,661</td>
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<tr>
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<td>445</td>
<td>15,972</td>
<td>13.9</td>
<td>32,763</td>
<td>2,400</td>
<td>35</td>
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<tr>
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<td>423,680</td>
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<td>986</td>
<td>38,381</td>
<td>9.1</td>
<td>231,781</td>
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<td>90,435</td>
<td>18,500</td>
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<tr>
<td>Rowan</td>
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<td>179,840</td>
<td>31</td>
<td>695</td>
<td>62,648</td>
<td>34.8</td>
<td>49,963</td>
<td>5,400</td>
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<tr>
<td>Wayne</td>
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<td>293,760</td>
<td>17</td>
<td>630</td>
<td>1174</td>
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<td>142,827</td>
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<td>Whitley</td>
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<td>281,600</td>
<td>34</td>
<td>880</td>
<td>46,517</td>
<td>16.5</td>
<td>73,414</td>
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<td>Wolfe</td>
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<td>12</td>
<td>353</td>
<td>16,615</td>
<td>11.6</td>
<td>57,701</td>
<td>3,100</td>
<td>67</td>
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</tbody>
</table>

Data sources:

US Census Bureau [http://quickfacts.census.gov/qfd/states/21000.html](http://quickfacts.census.gov/qfd/states/21000.html)
Kentucky State Data Center [http://ksdc.louisville.edu/kpr/popest/est.htm](http://ksdc.louisville.edu/kpr/popest/est.htm)

### 3.2 Data

Bill Luhn, the GIS Coordinator for the DBNF, supplied the vector road and land classification data for the spatial analysis in GIS. The road layer was originally projected to North America 1983 Geographic Coordinate System (GCS). In order to make
coordinates compatible with the land classification layer, the road layer was transformed to NAD 1983 State Plane Kentucky FIPS 1600 feet coordinate system. Maps are displayed in Lambert conformal conic projections.

Deer harvest data were obtained from the Kentucky Department of Fish and Wildlife. Daniel Boone National Forest statistics were obtained from the National Forest Service, and county statistics were obtained from the United States Census Bureau, Kentucky State Data Center, and the United States Department of Agriculture. Landsat 7-TM remote sensing imagery of the DBNF was downloaded from the United States Geological Survey (USGS).

3.3 GIS Analysis

3.3.1 Use of GIS in Habitat Selection

The role of GIS in conservation biology and ecological modeling has expanded and enhanced conservation efforts for endangered species (Bishop et al. 2002, Akçakaya 1994). Bishop et al. (2002) listed several applications of conservation GIS: habitat conservation planning, land use management, environmental impact assessment, mapping of the spread of invasive species, species richness, or diversity, prediction of species distribution, and habitat selection for the reintroduction of rare species.

As noted by Bishop et al. (2002), GIS is a useful tool in habitat selection. It involves the combination of digital spatial map layers with abiotic and biotic data for habitat criteria. Habitat analysis considers many factors, such as habitat use and habitat availability, the location where species spend most of their time, the selection of required resources, and the amount of energy animals expend foraging or hunting. For instance,
modeling fish habitat requires hydraulic modeling with data on the stream flow conditions and biological sampling statistics on the conditions in which fish live (Merwade et al. 2004). For larger animals such as wolves, radio-collars and GPS monitoring can be used to track not only the geographic extent of distribution but also the frequency and intensity of habitat use. These data can then be incorporated into GIS for further analyses.

Wydeven et al. (2006) stated “Wolves lend themselves well to examining of their habitat selection using GIS. Wolf packs occupy fairly discrete areas that are maintained as territories, and represent the breeding potential of a wolf population.” With clear boundaries and specific home range sizes, wolf territories can be transformed into distinct polygon vectors to determine areas. With the assumption that a larger area can support a larger wolf population and a higher breeding potential, GIS analysts will be able to calculate the carrying capacity for wolves and population growth.

Habitat selection studies often require many biophysical data, such as vegetation cover. With GIS, biologists may collect these empirical data more efficiently (Akçakaya 1994). Traditional vegetation mapping with aerial photographs interpretation accrues high costs in labor and time. However, with the integration of ecological, statistical and data models into GIS, vegetation mapping can be more accurate and realistic, and it can be acquired with reduced costs (Accad and Neil 2006). Whether studies focus on distribution patterns, densities or predictions, spatial statistics in GIS facilitate data collection and quantitative analyses.

3.3.2 GIS Models and Software for Habitat Selection
As GIS in conservation science becomes more prominent, computer engineers have designed software and extension tools specifically for ecological modeling. Akçakaya (1994) analyzed population viability and produced metapopulation models in GIS using RAMAS, a software package, which integrates “landscape data on habitat requirements with demographic data to analyze risks of extinction, evaluate management options, and assess human impact on wildlife populations.” ESRI also provides several extension tools (such as XTools for ArcGIS) and GIS-linked software (such as Hawth’s Analysis Tools, Patch Analyst 4.0, EcoSim, EM4), which are available for download.

In this research, traditional spatial analyst tools were chosen to identify suitable habitats for red wolf reintroduction because wolves are habitat generalists. The finer-scaled patch analysis is not necessary. With no wolves present in the DBNF, this research also lacked the data to predict animal distributions and population growth with the more sophisticated tools.

3.3.3 Kernel Density Estimation in Habitat Selection

In previous wolf habitat GIS analyses, road density served as proxy for human activities. It is used as a delimiting factor for wolves as high road densities equate to higher mortality risks. In North Carolina, vehicular and gunshot incidents account for more than half of the red wolf deaths. Road density is therefore an important predictor for suitable wolf habitat.

In this research, the vector road data included all roads, paved and unpaved, public and private, within the DBNF. In total, 3691 line segments existed, representing the highway, arterial, collector, and local roads. They were classified by road type:
undefined highway, free flowing mixed traffic, congested during heavy traffic, flow interrupted/limited use and slow flow or blocked. The latter two types of roads consisted of forestry, fire service and closed secondary roads and were removed from the analysis due to their assumed negligible risk to wolves. Roads extending beyond the forest boundary were clipped.

Earlier models, such as those by Mladenoff et al. (1995) and Shaffer (2007), only considered simple road density. Simple density is calculated by adding the lines that fall within an area and dividing the sum by the area size. Many of these models neglect to weight the risk factor associated with different road types. To enhance the performance of these models, this analysis attempted to capture the weighted risks by adopting the more sophisticated kernel density estimation. In this estimation, the road density was calculated as a weighted function based on the assumption that asphalt highways posed the greatest threat to wolf mortality and local gravel roads the least. A new field of “Rank” was added to the road layer’s attribute table according to traffic volume and associated risk to wolves. Kernel density calculations then summed all the values of the kernel surfaces—the smoothed curved radius around a line, with density greatest on the line and diminishing outward—and produced smoother results than the simple density function. In GIS spatial analyst, the kernel density estimation is calculated with the Epanechnikov $K$ estimator, an optimal smoothing function determined by $K(t) = \frac{3}{4} (1-t^2)$, where $|t| = d/h \leq 1$ ($d =$ the distance between the cell and the line in the dataset). Kernel density estimations, unlike other interpolation techniques, such as kriging-cokriging, trend surface, and regressions, aim to produce smooth commutative density functions.
(Amatulli et al. 2007), producing the “hot spots.” This weighted function may produce a more realistic picture of road density within a space.

Worton (1989) used kernel density estimates for home-range studies. He described the kernel process as a probability density function \( k \) placed over a data point or line, and the addition of \( n \) components constructs the estimator. The kernel represents a density. Therefore, the estimation reflects a “true probability density function.” The smoothing parameter controls the variation in each component, with direct correlation existing between size of the bandwidth \( h \) and the scope and scale of detail in the data observations. The probability function is expressed as

\[
\hat{f}_h(x) = \frac{1}{Nh} \sum_{i=1}^{N} K \left( \frac{x - x_i}{h} \right)
\]

where \( K \) represents the kernel and \( h \) represents the bandwidth. The bandwidth defines the radius of the circle of each grid cell, and, in GIS, the default bandwidth measure is based on the geographic extent of the point or line patterns. While the selection of the bandwidth is important, the process of selection is more art than science.

With their smoothing parameters \( (h) \) and more efficient use of locational data, kernel density estimators outperform histograms. In fixed kernel methods, the smoothing parameters are fixed values. Adaptive kernel methods allow the smoothing parameter to vary, such that low concentration areas have higher \( h \) values than areas with high concentration of points as such they help to produce a smoother estimate. Using two data sets, Worton (1989) illustrated the application of the kernel density estimation for home-range analyses and compared the results between a least-squares cross-validation and an ad hoc approach for selecting \( h \) values. Worton defined the ad hoc method of choosing the value of \( h \) as the use of “the optimum \( h \) value obtained for some standard distribution,
such as the normal distribution.” The least-squares cross-validation demonstrated greater clarity in the “utilization distribution” (UD) density than the ad hoc approach, which over-smoothed the estimate. The accuracy of the UD estimator, therefore, depends on the value of $h$. However, he iterated that the ad hoc fixed kernel density estimation, if accuracy does not matter as much, provided all the pertinent details to UD density, such as the location of the modes. Worton (1989) also reported that kernel density estimators successfully explained an animal’s movement within home-range territories.

Seaman and Powell (1996) evaluated the accuracy of kernel density estimation for home-range analysis and found fixed kernel performed better than the adaptive kernel. In general, the density estimate will be high in areas with many observations and, conversely, low in areas with few observations (Seaman and Powell 1996).

Though viewed as the “most reliable contouring method in ecology” (Hemson et al. 2005), kernel density estimation has only been applied to home-range analysis, animal movements and resource use, and measurements of overlap areas of species distribution, and other properties of the location such as soil, temperature and photosynthetic rate (Seaman and Powell 2005) but not road density. Road analysis with kernel density estimation has been limited to networks in the urban environment, such as traffic monitoring, accidents, and bus stops. This research aims to demonstrate the effectiveness of the kernel density estimation for predicting habitat suitability based on the weighted road density risk-factor.

3.3.4 Kernel Density Output
As a general cartographic rule, maps for presentation purposes have 4 to 6 classes; therefore, the kernel density output values for road density were reclassified from 9 to 5 classes, which generated a better spatial pattern. Natural breaks classification was used rather than GIS’s equal interval default because natural breaks bases itself on the frequency in the data and breaks the values into natural or intuitive classes. While this does not reflect wolf suitability, it allows the analyst to understand the intensity and frequency of road density calculated by the kernel density estimation, especially in comparison to the simple density method.

In order to set the range of wolf suitability according to previous studies (Mladenoff et al. 1995; Mladenoff et al. 1997; Wydeven 2001; Paquet et al. 2001), wolf suitability was then rated as highly suitable for core areas where den sites would occur (road densities less than 0.23 km/km²), high (0.23-0.45 km/km²), medium (0.45-0.6 km/km²), low (0.6-0.84 km/km²), and unsuitable (more than 0.84 km/km²).

3.3.5 Criterion for Selecting Potential Sites

With a low probability of wolf persistence in areas with road density greater than 0.68 km/km² or 0.7 km/km² (Thiel 1985; Fuller et al. 1992; Wydeven et al. 2001), this threshold was used in the raster calculation as one of the main variables in the final wolf habitat suitability model. The second variable followed Mladenoff et al.’s (2009) adjustment of the original logistic regression model to exclude agricultural lands.

Color values were assigned for the land cover classification data to illustrate a more natural composite: green colors for forest, blues for water and wetlands, reds for human development and yellow-oranges for agriculture. The layer was then reclassified
according to wolf suitability. Areas with crop and pasture were reclassified as null for unsuitability, and the remaining classes were reclassified as positive for suitability. Human development need not have been reclassified for unsuitability as it was already factored into the road density calculations and thresholds.

With the use of the raster calculator, areas with low road density and forested, non-agricultural landscapes were determined and merged. The red wolf habitat suitability was then converted from raster to vector in order to highlight habitat patches with a minimum area of 50km² and potential restoration sites.

3.3.6 Assessment of Methodology

In general, non-parametric estimations of density distribution such as the kernel density estimation perform more robustly than parametric estimations, especially if the probability density function does not follow a normal distribution curve. Non-parametric estimations rely on fewer assumptions, which may result in fewer misinterpretations. Since this research ranks the roads by mortality risk, the use of the non-parametric kernel density estimation is more appropriate.

However, to assess the kernel methods, road density was recalculated with the simple density function. Three red wolf habitat suitability models were created. The first model based the simple road density off the shape lengths. Logarithmic and exponential transformations were then applied to the road lengths to normalize the skewness in the density probability and to generate “goodness-of-fit” models because non-parametric density estimations may perform “poorly” if the densities are too far from the Gaussian shape (Wand, Marron and Ruppert 1991). Instead of logistic regression and less rigorous
statistical assumptions, Corsi, Dupré and Boitani (1999) used the logarithmic transformation to normalize the density data on dump sites and ungulate species to obtain similar results.

An evaluation of the influence of the weighted roads within the model was determined by using the simple density function with the weighing factor and by using the kernel density estimation without the weighing factor. Outputs were compared to the weighted kernel density estimation model.

3.3.7 Field Verification

To ascertain the effectiveness and viability of the model and methodology, further investigation was conducted on a few selected potential restoration sites generated from the kernel density estimation. Field work, which involved personal observation, was used to verify whether the sites are actually suitable for reintroduction. Besides proximity to streams and roads were measured in GIS, slope analysis was performed on these sites using the Digital Elevation Model (DEM) layer of the DBNF. Since the literature indicated two different gradient thresholds (20° and 40°), the resultant values were reclassified into three ranges: 0-19.9, 19.900000001- 39.9, and > 39.900000001. Moreover, vegetation cover, serving as a surrogate for prey availability, was determined by first running a Tasseled Cap Transformation of a Landsat 7 image of the DBNF. The Tasseled Cap Transformation optimizes data viewing of vegetation by weighting the reflectance of the pixels to produce a composite index of brightness, greenness, and wetness. In GIS, the habitat patch layers were overlain with the Tasseled Cap
Transformed image, and the extent of greenness was measured. Deer density was then estimated using the equation

\[ \text{Deer/km}^2 = 2.2789 + 0.0533 \times \text{greenness} \]

where greenness represented the vegetation cover (Carroll et al. 2001).

While efficient, modeling habitat selection in GIS is to be used as a guide and not meant as a replacement for fieldwork. GIS data might not portray the heterogeneity, richness, or quality of the environments that impact the fitness and survival of a species, and only through site verification can a more accurate assessment be made.
Chapter Four

Results

The amount and distribution of favorable red wolf habitat was mapped in a stepwise fashion.

4.1. Road Density

The spatial distribution of road density (Fig. 7) shows the highest densities are associated with Interstate 75, major state byways, the Red River Gorge geological area, and the large lake recreation areas in the north (Cave Run Lake) and south (Cumberland Lake). Wolf habitat ranges from optimal to unsuitable (Fig. 8) with the suitable wolf habitat patches with road densities less than 0.7 km/km² (Fig. 9).
Figure 7: DBNF Road Density: Kernel Density Estimation with Weighing Factor. Road density as the predictor model in GIS. Dark green represents low road density areas and suitable wolf habitat.
Figure 8: DBNF Road Density: Red Wolf Habitat Suitability Ranges
The wolf habitat suitability ranges indicate optimal core habitat in darker green, good habitat in bright green, fair habitat in yellow, poor habitat in orange and unsuitable habitat in red.
Figure 9: Road Density of the DBNF: Red Wolf Habitat Suitability
Map results of raster calculation for more suitable and less suitable habitat based on road density. Potential “optimal” patches have road densities between 0 and 0.7 km/km².

4.2 Forest Composition and Land Cover

Forest dominates the landscape composition of the DBNF (Fig.10). Within the forest, the extent of pasture and crop land (Fig.11) appears more concentrated along the
southeastern edge of Stearns District, the Morehead District and the western and northwestern region of the Red Bird District.

Figure 10: Daniel Boone National Forest: Landcover and Landuse Classification. Map shows natural color composite of the different landcover classes.
Figure 11: Red Wolf Habitat Suitability Model: Natural vs. Agricultural Lands. Results of raster calculation for the landcover/landuse classification.

4.3 Red Wolf Habitat Suitability Model: Kernel Density Estimation Methodology

The red wolf habitat suitability model based on road ranks and kernel density estimation (Fig.12) reflects patches with low road density and mainly natural landscapes,
while the map of potential restoration sites (Fig. 13) includes an additional requirement for suitability and delineates patches with areas greater than 50 km². As a result, this study identified nine patches as potential sites for red wolf reintroduction.

Differences between the kernel density estimation of road densities with and without the ranking system appear evident (Fig. 14). With higher density probabilities in areas where road type would have carried a lower mortality risk in the weighted kernel methods, the suitability model without the ranking system (Fig. 15) predicted very few, smaller habitat patches and only one potential restoration site for the red wolf.
Figure 12: Red Wolf Habitat Suitability Model: Kernel Density with Weighing Factor. Road density based on road ranks and kernel density estimation and landcover rasters merged. Red designates potential habitat suitable for red wolves: areas of low road density and mainly natural landscapes.
Figure 13: Potential Red Wolf Restoration Sites: Kernel Density with Weighing Factor. Map of the nine potential red wolf restoration sites. Patches have areas greater than 50 km², low road density and suitable habitat.
Figure 14: Road Density of the DBNF: KDE Overlay with and without Weighing Factor. Map of the overlay of the kernel methods with and without the weighing factor. Shadowed grey areas represent the road density without the ranking system.
Figure 15: Red Wolf Habitat Suitability Model: Kernel Density without Weighing Factor. Map of suitability model based on kernel density without weighing factor. Fewer patches with much smaller areas resulted. This method produced one potential restoration site at the southern border.

4.4 Red Wolf Habitat Suitability Model: Simple Density Methodology
On the other hand, the red wolf suitability model based on the simple density function (Fig.16) returned a limited number of patches and none with areas greater than 50km². The simple density function performed more like the kernel density estimation when the weighted roads were incorporated into the calculations. The model based on the simple density and weighing factor (Fig. 17) produced fewer reintroductions sites with smaller areas as compared to this research methodology’s model. Table 2 compares the areas of the potential restoration sites of both models.

<table>
<thead>
<tr>
<th>KDE Habitat Patch</th>
<th>Area km²</th>
<th>SD Habitat Patch</th>
<th>Area km²</th>
<th>Difference</th>
</tr>
</thead>
<tbody>
<tr>
<td>1</td>
<td>65</td>
<td>1</td>
<td>55</td>
<td>10</td>
</tr>
<tr>
<td>2</td>
<td>86</td>
<td>2</td>
<td>60</td>
<td>26</td>
</tr>
<tr>
<td>3</td>
<td>112</td>
<td>3</td>
<td>82</td>
<td>30</td>
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<tr>
<td>4</td>
<td>127</td>
<td>4</td>
<td>112</td>
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</tr>
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<td>8</td>
<td>131</td>
<td>8</td>
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</tr>
<tr>
<td>9</td>
<td>391</td>
<td>9</td>
<td>369</td>
<td>22</td>
</tr>
</tbody>
</table>

Table 2: Habitat Patch Comparison of the Different Model Results.
Potential restoration sites derived from the kernel density methods were larger. In total, the kernel density model predicted 1207 km² of suitable sites for red wolf reintroduction, whereas the simple density model only predicted a total of 1019 km². With home ranges of 25-130 km², the 188 km² difference between the models could support 1.45 to 7.52 red wolf packs, or approximately up to 37.6 individuals.

The Logarithmic Transformation (Fig. 18) produced more suitable habitat patches than the kernel density and simple density estimators; however, the suitability is questionable as suitable patches appeared in unsuitable high-road density areas such as in Red River Gorge, Cave Run Lake and Cumberland Lake. On the other hand, the Exponential Transformation (Fig.19) produced fewer suitable habitat patches than the kernel density methods but more than the simple density estimator. It still seems as if the kernel methods performed more robustly, and its use seems justified. However, the
question remained if the model based on the kernel density and weighing factor could facilitate information about the actual suitability of the habitat patches.

Figure 16: Red Wolf Habitat Suitability Model: Simple Density without Weighing Factor
Map of the red wolf habitat suitability model based on the simple density calculations and unranked roads shows limited habitat patches. None measure more than 50 km².
Figure 17: Potential Red Wolf Restoration Sites: Simple Density with Weighing Factor. Map of the potential red wolf reintroduction sites based on ranked roads and simple density. Patches have areas greater than 50 km², low road density and suitable habitat.
Figure 18: Red Wolf Habitat Suitability Model: Logarithmic Transformation. Logarithmic transformation produced many suitable but questionable habitat patches. Suitable habitat appears in unsuitable areas such as Red River Gorge, Cumberland Lake and Cave Run.
Figure 19: Red Wolf Habitat Suitability Model: Exponential Transformation. Exponential transformation produced fewer suitable habitat patches than the Log T; however, it does produce more than the simple density estimator.
Figure 20: Potential Red Wolf Reintroduction Sites in the Red Bird District. Habitat Patch 1 and Habitat Patch 2 represent the two potential reintroduction sites considered for further investigation.
Out of the nine potential sites identified by the model, two sites seem to be more appropriate because of their forest buffers, connectivity to other patches and lack of agriculture. Referred to as HP1 and HP2, these patches will be further analyzed. Details such as proximity to streams, roads, and frequently visited areas in HP1 and HP2 (Fig. 20) were investigated. Measurements in GIS confirmed area of patch, proximity to streams, essential for thriving ecosystems and wolf survival, and the distance from roads (Table 3). Road densities at the cores ranged from 0.07 km/km² to 0.12 km/km² with road densities increasing toward the edges. With home ranges for red wolves between 25 km² and 130 km² (Phillips, Henry and Kelly 2003), HP1 and HP2 fulfill the minimal requirements. HP1 has an approximate area of 65 km², and the distance from the core to the Red Bird River is estimated at 12 km. HP2 has an approximate area 86 km², and distances from the core to the large stream networks range from 12 km to Kentucky River in the west, 6 km to Greasy Creek in the south, and 7 km to Grassy Fork in the east. However, observations of small streams and creeks did not appear in the GIS data, and it is assumed water within the habitat patches is adequate for wolf survival and may limit their need to travel outside the boundaries to access the larger streams. HP1 contains the Red Bird Wildlife Refuge as well as a significant portion of the Red Bird Crest Trail that allows hiking, biking, horseback riding, and ATV activities.

<table>
<thead>
<tr>
<th>Patch</th>
<th>N</th>
<th>E</th>
<th>S</th>
<th>W</th>
</tr>
</thead>
<tbody>
<tr>
<td>HP1</td>
<td>4.0 km</td>
<td>8.0 km</td>
<td>3.0 km</td>
<td>3.0 km</td>
</tr>
<tr>
<td>HP2</td>
<td>4.65 km</td>
<td>2.8 km</td>
<td>5.4 km</td>
<td>4.8 km</td>
</tr>
</tbody>
</table>

Table 3: Road Proximity to Potential Reintroduction Sites.  
Approximate distances from northern, eastern, southern, and western borders to roadways.
Field observations of the rugged HP1 and HP2 indicated a need for slope analysis. Three major streams, the Kentucky River, Greasy Creek, and Grassy Fork, cut deep, steep-sided ravines that bordered HP2. With no roads traveling into the patch, access was restricted by the steep slopes. Slope analysis of HP1 (Fig. 21) and HP2 (Fig. 22) revealed hilly terrains in both patches, more so in HP2. The literature disagrees on the slope gradient threshold (20° or 40°), but Paquet, Wierzchowski and Callaghan (1996) state wolves prefer valley bottoms and elevations less than 1,850 meters. However, wolf migration corresponds to vertical migration of prey species (Paquet, Wierzchowski and Callaghan 1996). Snow depths greater than 40-50 centimeters restrict wolf movement below 1,700 meters, but wolves will use plowed roads and snowmobile trails to access higher elevations (Paquet, Wierzchowski and Callaghan 1996).
Figure 21: Elevation and Slope Gradient in Habitat Patch 1. Slope analysis of Habitat Patch 1. Red indicates steeper slopes that wolves may not use.
Figure 22: Elevation and Slope Gradient in Habitat Patch 2. Elevation and slope analysis of Habitat Patch 2. Red indicates steeper slopes and yellow-orange indicates higher elevations that wolves may not use.

In HP1 (Fig. 23), the extent of greenness value was 69.618, and vegetation cover represented 99.3% of the total land area. HP1 could, in theory, support an estimated deer density of 5.9895 deer/km². Vegetation cover in HP2 (Fig. 24) was determined to be
84.6% of the total land area, given its extent of greenness value of 76.121, whereby HP2 may have the capacity to support 6.336 deer/km². In regards to adequate prey base, both habitat patches meet the minimum requirement of 5 deer/km² for red wolf. Photographs of HP1 and HP 2 (Fig. 25, 26, 27 and 28) show a fraction of the suitable wildlands environments for the red wolf.
Figure 23: Extent of Greenness in Habitat Patch 1.
Remote sensing image of Habitat Patch 1 showing extent of greenness.
Figure 24: Extent of Greenness in Habitat Patch 2.
Remote sensing image of Habitat Patch 2 showing extent of greenness.
Figure 25: Photograph of Habitat Patch 1.
Meadow off one of the forestry service roads in Habitat Patch 1.

Figure 26: Photograph of Habitat Patch 1.
Stream running alongside forestry service road in Habitat Patch 1.
Figure 26: Panoramic Photograph of the Eastern Side of Habitat Patch 2.
Figure 27: Photograph of Habitat Patch 2.
Small stream running through southern border of Habitat Patch 2.
Chapter Five

Discussion

For the reintroduction of rare and endangered species such as the red wolf, identifying high-quality, suitable habitat is vital. Poor pup survival from parvovirus further imperiled the wild population and led to the discontinuation of the red wolf recovery project in the Great Smokey Mountains National Park (USFWS 2007). This demonstrates the need to model as many abiotic and biotic influences on species’ survivorship as possible.

While unfragmented and undisturbed habitat represents the optimal habitat for wolves (USFWS 2007), the extent of historic red wolf range encompasses the populated southeast of the United States (Nowak 1992). Pristine wildlands as the criterion for restoration of viable wolf populations may be an unattainable ideal. As documented in ARNWR, certain packs hunted on farmland and thrived in this capacity alongside humans (Phillips, Henry and Kelly 2003). Mladenoff et al. (2009), on re-examination of their original prediction model, discovered wolf occupation in habitats that were assumed unsuitable. This showed a lack of knowledge about the wolf’s adaptability. The assumption that wolves do not belong in humanized landscapes may be unjustified (Lynn 2002). By permitting wolves use of fragmented habitats, wildlife biologists and ecologists may broaden their knowledge about the species’ ability to persist in the human-disturbed wildlands. This knowledge may lead to the development of more effective management strategies to mitigate conflicts between man and wildlife in shared spaces.
The original logistic regression model introduced by Mladenoff et al. (1995) predicted the probability of wolf distribution and occupation in regions already inhabited by wolves. However, red wolves have not been present in Kentucky since the early 1900’s (DeBlieu 1991), and the potential for red wolf restoration in one of the few remaining contiguous forests in the Southeastern United States had yet to be assessed.

Descriptions of red wolf habitats in ARNWR (Phillips, Henry and Kelly 2003; USFWS 2007) and habitat selection criteria specified by Shaffer (2007) provided the guidelines for this thesis research. Like ARNWR, with its mosaic of landscapes and mixed mesophytic forest, the DBNF contains favorable red wolf environments and adequate prey populations (USDA 2003; USFWS 2007). Road density as the best predictor for suitable wolf habitat was the main focus of this research’s habitat selection (Mech 1995; Mladenoff et al.1995; Mladenoff and Sickley 1998; Harrison and Chapin 1998; Mladenoff et al. 1999; Ratti et al. 1999; Corsi et al. 1999; Unger 1999; Frair 1999; Houts1999; Kohn et al. 2000; Wydeven et al. 2001; Shaffer 2007; Mladenoff et al. 2009).

This thesis methodology, though, adopted a different and unique approach to evaluating road density in a study area. Instead of using the simple density calculations for road density, the red wolf habitat selection model incorporated the kernel density estimation as a smoothing method and a ranking system as a weighing factor for mortality risks based on road type. The type of road made a difference in the inquiry. Frequency and speed of travel is higher on asphalted highways and state byways than on gravel “country” roads, and, therefore, the chances of vehicular strikes with wildlife will increase on roads more often travelled. Rowan County in the northern Morehead District had the highest deer-car collisions, and the model did not predict any suitable habitat in
the region. On the contrary, Leslie and Clay Counties in the Red Bird District had very low deer-car collisions, and the model predicted four potential restoration sites. It would seem the assumption built into the methodology is realistic. In the model, the highways and state routes received the highest mortality-risk rank, and, in reality, the higher deer-car collisions occurred in the northern and southern regions of the DBNF, areas associated with State Route 60, Interstate 64, and Interstate 75.

Without the weighing factor, the road type does not make as much of a difference in the density probability, and the density is, therefore, greater in areas than where the type of road carries a theoretically lower mortality risk. The difference in the intensity and frequency of high density was greater in the simple density than even the kernel density without the weighing factor, indicative of the kernel density estimation being a weighted density probability function in and of itself. The weighing factor, therefore, had the greatest influence on the simple density model’s performance. Without weighted roads, the simple density method resulted in a red wolf suitability habitat model with limited habitat patches and no potential restoration sites. The simple density in conjunction with the weighing factor, on the other hand, produced a habitat selection model more similar to the weighted kernel method model by identifying eight potential restoration sites. The red wolf suitability habitat model integrating the kernel methods and ranking system still performed more robustly than the weighted simple density model by identifying nine potential restoration sites with larger areas.

The final model displayed distinct spatial patterning of the nine potential restoration sites. Clusters occurred in the Red Bird District (Habitat Patches 1, 2, 3 and 4) and in the Somerset District (Habitat Patches 6 and 7), while random isolated patches
occurred in the London (Habitat Patch 5), Stearns (Habitat Patch 8) and Stanton Districts
(Habitat Patch 9). In terms of connectivity, the patch clusters would better facilitate wolf
movement and genetic flow. The corridors, however, are areas of high road density, and,
while wolves do cross roads, the risk of mortality increases (Mladenoff et al. 1995).

Though adequate in acreage and forest cover, the DBNF is fragmented by human
settlement and activity. Moreover, human lands use appears to cause minimal
fragmentation in five of the nine potential restoration sites (Habitat Patches 3, 4, 6, 7 and
9). Wolf survivorship in such habitats depends upon the attitudes and behaviors of the
people present as Mladenoff et al. (1995:282) noted, “As long as wolves are not killed,
they appear to have the ability to occupy areas of greater human activity than previously
assumed.”

Demographics in the DBNF may preclude the co-existence with wolves. Studies
showed that lower educational and economic status foster anti-wolf sentiments (Ratti et
al. 1999). In the DBNF counties, average median household income ($27,736) and
poverty levels (30%) fall below the average median household income ($40,299) and
poverty levels (17.2%) of the state (US Census Bureau). 57% of county residents
compared to 74% of state residents obtained a high school diploma, and 9% of county
residents compared to 17% of state residents hold a Bachelor’s degree or higher (US
Census Bureau).

With public education and community involvement in decision-making about red
wolf reintroduction and management, tolerance and acceptance may be fostered, and any
existing negative attitudes may change towards the positive (Musiani and Paquet 2004;
Fritts et al. 1997; Mech 1995; Phillips 1995). Conducting a survey of those living and
working in region and even of those visiting the DBNF would provide insights into the attitudes toward the red wolf and its restoration.

Despite this reservation, the findings of this research appears concordant with the elk restoration project in that optimal habitat occurs in the Red Bird District. Four potential patches (Habitat Patches 1, 2, 3 and 4) show more promise as core habitat ranges for red wolves, with Habitat Patches 1 and 2 the most promising with forest buffers and no agriculture. Further evaluation of Habitat Patches 1 and 2 suggested their viability for red wolf reintroduction.

Major streams such as the Red River, Kentucky River, Grassy Creek and Greasy Fork flow roughly parallel to roadways that border the habitat patches, but smaller streams and creeks flow within patch boundaries and cores. Abundant water and prey resources, such as deer and raccoon, indicate ecosystems in which the red wolf would thrive. In theory, HP1 and HP2 can support one or two packs, and gene flow between the two is possible. Smaller suitable habitats to the west and southwest and the larger suitable habitats to the northwest may provide home ranges for future generations of dispersing wolves.

With portions of the Redbird Wildlife Management Area and Redbird Crest Trails in Habitat Patch 1, an advantage includes the facilitation of red wolf monitoring for wildlife biologists and viewing for wildlife enthusiasts. The hunting, off-road vehicle traffic, horseback riding, biking, and hiking may seem disadvantageous for red wolf occupation and survival. However, red wolf populations in ARNWR and Pocosin Lakes National Wildlife Refuge have persisted in similar recreational areas.
Limited road access into Habitat Patch 2, on the other hand, makes it more isolated and remote than Habitat Patch 1. This isolation may increase suitability, despite the evident barrenness in the north and higher elevation taken from remote sensing imagery. Whether clear-cutting from logging, mountain-top removal from coal mining, or tree-stand death from disease or forest fire, the cause for the bright spot determined by the Tassel-Cap Transform is unknown.

In southeastern Kentucky, old-growth logging in the southern half of the DBNF has ceased due to the forest legacy program and partnerships in watershed restoration, and coal mining has been abandoned due to Federal Surface Mining, Reclamation and Enforcement Act in 1977. Conservation of the forests and watershed is fostered through the financial assistance to private landowners from programs such as the Forest Stewardship Program, the Conservation Reserve Program, the Environmental Quality Incentives Program, the Wetland Reserves Program, the Wildlife Habitat Incentive Program, the Kentucky Heritage Land Conservation Fund, and the Urban and Community Forestry Program.

Conservation of the forest, the watershed basin, and the 23 threatened and endangered extant species in the DBNF would benefit from the red wolf’s return. In addition, the public should be educated about the values of wildlife and the importance of ecosystem diversity. The 74.2% of Kentuckians who opposed logging of public lands (UK Survey 1994) may support red wolf reintroduction, especially if the public understands the ecological role of the red wolf in maintaining ecosystem health and diversity.
Chapter Six

Conclusion

This thesis research identified nine potential red wolf restoration sites in the DBNF. The results from this research, however, are not final. Despite the potentiality, comprehensive knowledge is lacking on how well the red wolf will establish home ranges in coyote-inhabited territories and if coexistence of the two canids can happen without hybridizations. Further research on the intraspecific relationships between the two canids and data on the coyote density and distribution within the DBNF is necessary.

Other uncertainties include the actual effects of red wolf reintroduction on prey species and the ecosystem. For example, the elk reintroduction to the DBNF has raised some concerns on its effects on the plant community and existing white-tailed deer populations. The elk may drive the deer from the forest into farmland, and, if red wolves were present and dependent on the deer, then the wolves may follow the deer and come into conflict with landowners. Detailed studies on the ecological consequences of wolf reintroductions in other regions could provide a guide to the possible outcomes in the DBNF.

In the study region, highways and state byways, rather than gravel roads, carry the greater threat to wolf survivorship, and, by weighting the road class by mortality risk and road density function, the resultant map portrays a more realistic portrayal of the road influence on the potential of wolf habitat. The simple density function, even if roads are ranked by mortality risk, has a tendency to underestimate the extent of suitable habitats.

This thesis research improved the methodology in habitat selection models for wolves and contributed to RWSSP by the identification of reintroduction sites in the
DBNF and the evaluation of habitat viability. Further fieldwork, such as collection of ground truth data, vegetation sampling, observations of herbivore and mesopredator habitat use, and local climate variations, is recommended. However, only the restoration of experimental populations will confirm suitability.

To ascertain the validity of the model, the built-in assumptions need to be tested. A future study requires traffic volume data of the road network in the DBNF and gunshot mortality distribution points from ARNWR to determine proximity to the type of roads most associated with incidental or intentional shootings of red wolves. If the data are available, a follow-up study in North Carolina to test the model’s predictive performance may help to further validate the model.

Revisiting previous predictions models with the weighted functions may also elucidate more on the effectiveness of the amended methodology. The original logistic regression model created by Mladenoff et al. (1995) may have predicted suitable habitat and wolf occupation with more accuracy if the road density variable had been determined by weighting the road class and performing the kernel density estimation. This thesis research methodology is not limited to the DBNF and could be applied to other study areas for predicting habitats where density is an important variable.
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http://www.dnr.state.wi.us/Org/land/er/publications/wolfplan/appendix/appendix_c.htm
Appendix

Kentucky Department of Fish and Wildlife

Appendix B: Daniel Boone National Forest Deer Harvests.
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Appendix C: Bath County Deer Harvests.
Data source: http://fw.ky.gov/harvest/deerharvest.asp
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Appendix D: Clay County Deer Harvests.
Data source: http://fw.ky.gov/harvest/deerharvest.asp
Kentucky Department of Fish and Wildlife
Appendix E: Estill County Deer Harvests.
Kentucky Department of Fish and Wildlife

Appendix F: Harlan County Deer Harvests.
Kentucky Department of Fish and Wildlife
Appendix G: Jackson County Deer Harvests.
Data source: http://fw.ky.gov/harvest/deerharvest.asp
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Appendix H: Knox County Deer Harvests.
Data source: http://fw.ky.gov/harvest/deerharvest.asp
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Appendix I: Laurel County Deer Harvests.
Data source: http://fw.ky.gov/harvest/deerharvest.asp
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Appendix J: Lee County Deer Harvests.
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Data source: http://fw.ky.gov/harvest/deerharvest.asp
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Appendix Q: Powell County Deer Harvests.
Kentucky Department of Fish and Wildlife

Appendix R: Pulaski County Deer Harvests.
Kentucky Department of Fish and Wildlife
Appendix S: Rockcastle County Deer Harvests
Data source: http://fw.ky.gov/harvest/deerharvest.asp
Kentucky Department of Fish and Wildlife

Appendix T: Rowan County Deer Harvests
Data source: http://fw.ky.gov/harvest/deerharvest.asp
Kentucky Department of Fish and Wildlife
Appendix U: Whitley County Deer Harvests
Kentucky Department of Fish and Wildlife

Appendix V: Wolfe County Deer Harvests
Kentucky Department of Fish and Wildlife