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Space Use And Habitat Selection By Bobcats In Southeastern Kentucky

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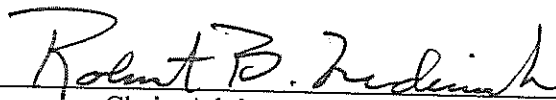
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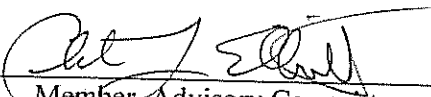
SPACE USE AND HABITAT SELECTION BY BOBCATS IN SOUTHEASTERN
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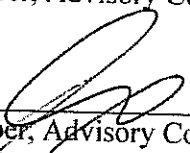
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SPACE USE AND HABITAT SELECTION BY BOBCATS
IN SOUTHEASTERN KENTUCKY

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in partial fulfillment of the requirements
for the degree of
MASTER OF SCIENCE
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DEDICATION

This thesis is dedicated to my grandma, Rose Shipley, who filled my childhood days with happy memories of flowers, animals, insects, and the rest of the natural world.

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Firstly, I thank my thesis advisor, Dr. Bob Frederick, for all of his expertise, advice, and patience throughout the last four and a half years. His numerous reviews have significantly strengthened this thesis and his mentorship helped me through some tough situations with the project.

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ABSTRACT

Population estimation and trend analyses are critically important for sustainable harvest and management of many species. The bobcat (*Lynx rufus*) plays important ecological and economic roles in Kentucky as a furbearer and mesopredator. I conducted a study of the bobcat in southeastern Kentucky as a twenty year follow-up to research conducted in the same study area. I VHF-radio-collared five (4 F, 1 M) bobcats and assessed space and habitat use patterns based on 58-65 locations per cat collected aurally approximately every 5 days over 12 months. Mean annual minimum convex polygon (MCP) home range size for all bobcats was 14.7 km² (n = 5, SE = 3.9 km²), and 22.2 km² (n = 5, SE = 7.5 km²) using the adaptive kernel (AK) method. Mean female annual home range size was 17.4 km² (MCP, n = 4, SE = 3.9 km²) and 27.4 km² (AK, n = 4, SE = 7.5). Mean female-female home range overlap was 29.1% (MCP, n = 6, SE = 8.7), and female-male overlap was 17.1% (MCP, n = 4, SE = 7.0). Mean female-female core (innermost 50% use) area overlap was 10.5% (MCP, n = 6, SE = 10.5), and female-male 12.1% (MCP, n = 4, SE = 12.1). Bobcats (all bobcats pooled) used forest in proportion to availability at the study area spatial scale, used open habitat more than expected, but avoided active mines ($P < 0.001$). Movement rate (mean = 0.12 km/hr) of a single GPS-collared male bobcat was lower during midday than during the morning, late afternoon, or nighttime periods. Also, more locations were recorded in forested habitat than expected based on habitat available within the home range, which contradicts the trend seen in the VHF data analysis, possibly indicating VHF data were not reliable in assessing habitat selection.

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ABBREVIATIONS

AK: Adaptive Kernel

ASM: American Society of Mammalogists

CG: Chest Girth

CMR: Capture-Mark-Recapture

df: degrees of freedom

ELL: Ear Length Left

ELR: Ear Length Right;

F: female

GIS: Geographical Information System

GPS: Global Positioning System

GSM: Global System for Mobile communications

Ha: Alternative Hypothesis

Ho: Null Hypothesis

HR: Home Range

hr: Hour

IACUC:

ID: Identification

KDFWR: Kentucky Department of Fish and Wildlife Resources

km²: kilometers squared

km/h: kilometers per hour

lbs: Pounds

Locs: Locations

m: Meters

M: male

M+F: male + female

MANCOVA:

MANOVA:

MCP: Minimum Convex Polygon

Min: Minute

MS: Microsoft

n: Number

NC: Neck Circumference

P: Significance level

PVB: Paul Van Booven

RFFL: Right Front Foot Length

RFFW: Right Front Foot Width

SE: Standard Error

SH: Shoulder Height

TAIL: Tail Length

TL: Total Length

TN: Trap Nights

VHF: Very High Frequency

WMA: Wildlife Management Area

X^2 : Chi-square value

ZA: Zygomatic Arch

CHAPTER 1

INTRODUCTION

BOBCAT ECOLOGY

The bobcat (*Lynx rufus*) is a mesopredator (Prugh et al. 2009) and an extant native felid found throughout most of the U.S., including Kentucky, where it is an important furbearer and ecological component. There are 36 extant species of wild felids across the world (Macdonald et al. 2010) and the genus *Lynx* is considered to be of African origin. *L. rufus* originated from the species *L. issiodorensis* but has decreased in size since first appearing in the fossil record approximately 2 million years ago (Lariviere and Walton 1997). Though the bobcat is approximately twice the size of the domestic cat and smaller than the Canada lynx (*Lynx canadensis*), this was not the defining feature that lead to the reclassification of the species under a different genus (Lariviere and Walton 1997). The bobcat was reclassified from *Felis rufus* to *Lynx rufus* in 1997 because it does not possess the P2 upper premolar, reducing the number of upper premolars from 3 to 2 as is typically found in the *Felis* (domestic cat) genus (Lariviere and Walton 1997). Bobcats have a spotted pelt in a range of colors from reddish brown to mixed grayish with a white-tipped short tail and small dark ear tufts (Howell 1997). Primarily solitary nocturnal hunters (though family groups have been known to hunt together), bobcats have a crepuscular activity cycle and typically rest during daylight hours (Lariviere and Walton 1997, Tewes et al. 2002).

As a mesopredator, the diet of bobcats is varied though hyper-carnivorous (Lariviere and Walton 1997). Fritts and Sealander (1978) found that bobcats in Arkansas consumed rabbits (*Lagomorph* species), squirrel (*Sciuridae*) species, rodents (*Rodentia* species), white-tail deer (*Odocoileus virginianus*), opossum (*Didelphis virginiana*), striped skunk (*Mephitis mephitis*), snakes, red fox (*Vulpes vulpres*), muskrat (*Ondatra zibethicus*), woodchuck (*Marmota monax*), various songbird species and upland game birds, while Tewes et al. (2002) found lagomorph and rodent species to be the dominant food sources. The variability in reported diet is matched with equally variable habitat across the bobcats range (Lariviere and Walton 1997). With the exception of Delaware, bobcat populations are stable or increasing, a testament to their adaptability (Howell 1997, Woolf and Nielsen 2000, Woolf and Hubert 2001).

In previous habitat selection studies, bobcats have been found to prefer a variety of habitat types, including black spruce, balsam fir, and white aspen (Fuller et al. 1985), lowland deciduous forests and upland coniferous forests (Lovallo and Anderson 1996b), developed and natural habitats (Riley et al. 2003), grass fields and brushy habitat types (Rolley and Warde 1985), and young regeneration forest stands and mature hardwood forests (Rucker et al. 1989). Similarly, avoidance of habitat types by bobcats was reported for birch and tamarack cover types (Fuller et al. 1985), upland conifer stands, upland deciduous forests, mixed savanna covers, and un-forested areas (Lovallo and Anderson 1996b), developed and altered open habitats (Riley et al. 2003), pine, deciduous, and mixed pine-deciduous forests (Rolley and Warde 1985), and short-leaf pine and mixed pine-hardwood habitats (Rucker et al. 1989). Bobcats have been found to

use mixed upland and young aspen cover types in proportion to their availability (Lovallo and Anderson 1996b).

A niche is the ecological role played by a species in a community, and is the range of physical and biological conditions needed for a species to maintain a stable or increasing population size (Morin 1999). This definition can be broken down into two types of niche, fundamental and realized niche. The fundamental niche are those conditions in which a species can persist in absence of competitors; the realized niche are the restricted range of conditions a species is able to exploit in the presence of competitors (Morin 1999). In areas inhabited by many species with overlapping fundamental niches, the aggregate effect of their competition can be reflected in the relative size of their respective realized niches (Morin 1999). How species compete and are able to coexist has been studied extensively (e.g. MacArthur 1958) and encompasses behavioral, morphological, and resource requirement differences between competing species (Morin 1999). Just as important as the study of competition is the related study of what happens when competitors are reduced or eliminated from an area, otherwise known as ecological release (Crowell 1961).

MESOPREDATOR RELEASE THEORY

Ecological (or prey) release describes the scenarios that could occur from the absence or negative change in the density or distribution of competitors or predators on a community, namely the direct effects of altering densities of one functional group on the next lower trophic level in a community or ecosystem (Crowell 1961, Soule 1966, Terborgh and Faaborg 1973, Kohn 1978, Sandin et al. 2010). Mesopredator release is a type of prey release where intermediate sized predators are released from the top-down

control exerted by larger predators (Brashares et al. 2010). Current definitions of mesopredators include a weight range, which has been argued to be arbitrarily limiting, placing all of the emphasis on the effects of mesopredators on prey and not enough on the interactions of apex predators and mesopredators, disjointing ecological principles from the term (Brashares et al. 2010). Mesopredator release is an important part of most trophic cascades, and mesopredator release theory hypothesizes that changes will have a negative effect on lower trophic levels (Brashares et al. 2010); trophic cascades being the phenomena of dramatic changes in ecosystem and nutrient cycling brought about by reciprocal changes in predator and prey populations when top predators are either added or removed from a food web

(<http://www.britannica.com/EBchecked/topic/1669736/trophic-cascade>, accessed 15 Oct 2012).

Ecosystems and the faunal communities contained therein are complex and by that virtue so is the theory of mesopredator release. Many factors may affect the outcome of changes in or losses of apex predators in an ecosystem, the most important being the productivity of the system (how much energy is harnessed by flora and transferred through the trophic levels), and the strength of interactions between apex predators, mesopredators, and prey (Brashares et al. 2010). Areas of high productivity should be dominated by apex predators, areas of low productivity should be dominated by mesopredators, and intermediate productivity areas should offer an opportunity for co-existence, as predicted by ecological theory (Brashares et al. 2010). So that in areas with high ecosystem productivity, the effects of mesopredator release should be strongest. The abundance of prey in productive ecosystems should allow mesopredator numbers to

increase once top-down control by apex predators is removed (Brashares et al. 2010). In less productive areas, mesopredators are more likely to be regulated by limitation of prey rather than predation, therefore the effects of mesopredator release should not be as great (Brashares et al. 2010).

Species interactions form the basis of community ecology theory and it is no surprise that these interactions can affect the outcomes of mesopredator release. When released from top-down control by apex predators, mesopredators are poised to increase in density because of their wider diet breadth and lower trophic position in the food web (Brashares et al. 2010). This also means that they may be able to suppress a variety of prey species. Because mesopredators can access prey of different size and shapes, they can have a profound impact on their ecological community when released from top-down control (Brashares et al. 2010). Many studies have found that mesopredators use a different prey base than sympatric apex predators (Estes et al. 1998, Barton 2005, Brashares et al. 2010). Apex predators usually have a more restrictive and carnivorous diet than most mesopredators, this may explain why differences in diet breadth and trophic position can lead to increased predation on lower trophic levels when apex predators are removed (Brashares et al. 2010). This difference in diet breadth, as well as foraging efficiency (mesopredators forage more efficiently), are reasons why an increase in density of mesopredators cannot be considered as simply ecological replacements of apex predators (Brashares et al. 2010). In systems with many apex predators, many mesopredators, and many prey species (i.e., high species and niche diversity), the effects of mesopredator release should not be as strong as in systems with only a few competitors (Brashares et al. 2010). Therefore the cascading impacts of mesopredator release from apex predator

removal in systems where apex and mesopredators consume many prey species should be less than in systems with prey specialists (Brashares et al. 2010). Omnivorous mesopredators can maintain high densities even in areas that have been altered because they are able to switch between different prey sources where less omnivorous predators may not (Brashares et al. 2010). It is when links between species are strong that mesopredator release should be most dramatic (Brashares et al. 2010).

While ecological theory may predict particular outcomes based upon specific criteria, measuring accuracy of these predictions is not simple. Changes in abundance or behavior of a mesopredator must be measured when an apex predator has been removed in order to test for mesopredator release (Brashares et al. 2010). Habitat variability and apex predators that are rare or cryptic can both affect the strength of the effects of mesopredator release on an ecosystem (Brashares et al. 2010). Changes in prey abundance, diversity, richness, or biomass, and increases in mesopredator abundance, can indicate mesopredator release if the changes exceed short term population variability (Brashares et al. 2010). Many studies have cited mesopredator release after apex predators disappeared (e.g., Wilcove 1985), but whether or not a release has occurred is often unknown (Brashares et al. 2010). Mesopredator release has previously been studied by attempting to quantify competition between sympatric mesopredators via the indicators previously listed (Chamberlain and Leopold 2005, Riley 2000). Determining the effects on competition between sympatric mesopredators when released from top-down control by apex predators would be difficult (Brashares et al. 2010).

BOBCATS, LAND USE AND MESOPREDATOR RELEASE

Bobcats are sympatric and compete with several different mesopredators such as coyotes, foxes, raccoons, skunks, and fishers for resources throughout the United States; in the southeast bobcats are sympatric with coyote (*Canis latrans*) and gray and red foxes (*Urocyon cinereoargenteus*, *Vulpes vulpes*), and in areas lacking larger predators such as mountain lions (*Puma concolor*) or black bears (*Ursus americanus*), coyotes may be considered the apex predator over bobcats (Lariviere and Walton 1997, Gilbert and Keith 2000, Riley 2000, Woolf and Hubert 2001, Tewes et al. 2002, Chamberlain and Leopold 2005). Prior studies have reported home range overlap for bobcats and coyotes (Chamberlain and Leopold 2005), bobcats and red or gray foxes (Riley 2000, Chamberlain and Leopold 2005), and bobcats and fishers (*Martes pennanti*; Gilbert and Keith 2000).

There are several different types of interspecies competition that can occur within a community and usually simultaneously as a way to partition resources. Encounter competition occurs when one species gains access to limited resources by interfering with the ability of its competitor to secure the same resources (Paine 2010). Consumptive competition occurs when some quantity of a resource is consumed by an individual thereby depriving others of it (Gilbert and Keith 2000). Territorial competition is usually documented by spatial segregation of species or individuals; areas where two individuals have > 10% overlap are considered to not be experiencing territorial competition (Gilbert and Keith 2000). Conclusions are often inferred from impacts to species demography while in the presence of a competitor or from changes in the use of a resource (Brashares et al. 2010). Strong competition can affect survival, reproduction, or spatial distribution

of the weaker, usually smaller species (Riley 2000). The smaller species can live in sympatry by finding physical refugia, using resources that are not available to the larger, stronger species, or using common resources more efficiently; this relies on spatial and temporal heterogeneity (Riley 2000).

Gilbert and Keith (2000) found that in areas where bobcats and fishers are sympatric, they exhibit encounter competition, with the bobcat as the dominate species as indicated by a constriction of diet by fishers in areas where bobcats were present and a diet broadening in areas where bobcats were absent. Similarly, Riley (2000) found that encounter competition between bobcats and gray foxes may have been likely since bobcats were the abundant, larger carnivore in the study (Riley 2000). Diet overlap between bobcats and gray foxes was very high, though no consumptive competition was found because the main prey source (meadow voles) were seemingly very abundant during all seasons (Riley 2000).

Bobcats, gray foxes, and coyotes are sympatric throughout their respective ranges in the United States and they partition habitats and prey; felids and canids generally hunt in different habitats though their diets overlap extensively (Chamberlain and Leopold 2005). Consumptive competition may be likely between bobcats and fishers because fisher predation on bobcat young reduced kitten survival rates (Gilbert and Keith 2000). Bobcats and fishers did not display territorial competition, but perhaps exclusion occurs at a finer scale for these competitors (e.g., exclusion from specific parts of the home range; Gilbert and Keith 2000). While gray fox, bobcat, and coyote home ranges overlapped substantially, their respective core areas remained fairly exclusive indicating exclusion may have occurred at this finer spatial scale (Chamberlain and Leopold 2005).

Likewise, gray fox and bobcat home ranges overlapped extensively in rural and urban areas, but core areas did not in urban areas (Riley 2000). However, urban area use by foxes may also be to utilize other food resources such as trash, ornamental fruit, pet food, etc. (Riley 2000).

In Riley's (2000) study, voles were important to bobcat diets in the urban areas and less so for fox diet in the rural areas since those areas had more forest habitat, which may have provided more cover for foxes and other food resources than in the urban areas, thereby reducing consumptive competition in the rural areas. Chamberlain and Leopold (2005) found that bobcats and gray foxes displayed variance between their diets, and that coyotes had a more varied diet than bobcats. Gray foxes likely maintained sympatry with bobcats and coyotes by using core areas in preferred habitats with low coyote use (Chamberlain and Leopold 2005). Competition may not have been so intense between bobcats and gray foxes as to extirpate either species, but use of urban areas by gray foxes may allow the species to have higher densities in areas of sympatry with bobcats (Riley 2000).

AFFECTING BOBCAT DEMOGRAPHIC FACTORS

The bobcat has been found to live for up to 32 years in captivity and up to 15.5 years in the wild (Lariviere and Walton 1997). Bobcats are polygamous and females are seasonally polyestrous (Rolley 1985); females that fail to become pregnant in early spring may come in heat again later in the spring or early summer (Lariviere and Walton 1997). Most females breed during their second spring (Rolley 1985), and males remain reproductively active until death (Lariviere and Walton 1997). The breeding season can vary with latitude, longitude, altitude, and climatic variations, and although breeding is

possible throughout the calendar year it mainly occurs from December to July (Lariviere and Walton 1997). Births generally occur from late April through June; reported gestation periods range from 50 to 63 days and average litter size is 2.4 (Howell 1997, Lariviere and Walton 1997). Young are raised by the female and the male provides no parental care; the nursing period lasts an average of 2 months and at 3 months of age juvenile bobcats will begin accompanying the female outside of the den site (Howell 1997, Lariviere and Walton 1997). Juveniles will begin to travel alone, near the den site, at about 6 months of age, and will disperse before the next litter is born (Kitchings and Story 1984).

Survivorship of bobcats varies throughout the U.S.; in the northeast annual survival of adults was 0.62, but declined to 0.49 and 0.19 when subject to heavy harvest and poaching (Fuller et al. 1995). In unexploited populations survivorship of adults can be high, while juvenile survival is typically low; in exploited populations juveniles tend to have a higher rate of survival, likely due to increased food availability from decreased numbers of adults (Howell 1997). A study of an unharvested population in California found that predation, disease, and starvation accounted for 35%, 15%, and 10% of deaths (Lembeck and Gould 1979). In the Midwest, survival of adults ranged from 0.53 to 0.66 and juvenile survival was found to be 0.30 (Rolley 1985). Assuming bobcat reproduction is a density-dependent function in a resource limited environment, an excess of adult males in relation to adult females may indicate an unexploited or oversaturated population (Tumilson and McDaniel 1988). One study found a high number of adult male bobcats in high density populations and a high number of adult female bobcats in low density populations (Lembeck and Gould 1979). Having a male-skewed population

could effectively stabilize the population growth rate by reducing the number of breeding females in the population and limiting use of those resources that are limited (Tumilson and McDaniel 1988). Sex ratios of bobcat populations may become skewed because of greater mobility of males, lesser mobility of females, increased activity of either sex during the breeding season, seasonal differences due to maternal care of young, degree of hunting pressure, the density of the population, or be interpreted as skewed because of inaccurate sexing or differential attractiveness to baits (Tumilson and McDaniel 1988). Fuller et al. (1995) found that other non-harvest causes of mortality accounted for up to 53% of deaths and recommended obtaining better indices of bobcat abundance, food, and disease in order to better manage populations, in addition to monitoring illegal exploitation.

Anthropogenic sources of mortality have the greatest influence on bobcat populations, especially exploitation and land use changes (Woolf and Hubert 2001). In a survey by Woolf and Hubert (2001) of state wildlife management agencies across the U.S., several biologists attributed increasing population growth to improved habitat. In states with stable bobcat populations interspecific competition with coyotes and habitat limitations were cited as the limiting factors to population growth (Woolf and Hubert 2001). Beginning in the 1970's bobcat pelts began to experience a dramatic price increase due to international restrictions on wildlife trade and demand from outside markets such as Russia and China, and bobcat pelts fetched an average of \$143.00 dollars/pelt at that time (Lariviere and Walton 1997, Fuller et al. 2005). This demand caused an increase in North American bobcat harvest from an estimated 14,230 individuals in 1970 to almost 76,000 in 1984 (Lariviere and Walton 1997). Hunting

pressure has been suggested to be responsible for skewed ratios in bobcats and harvest season timing is likely an important factor in the interpretation of sex ratios; extended harvest seasons can increase the proportion of kittens in exploited populations (Knick et al. 1985, Tumilson and McDaniel 1988). Previous studies have found that males are often favored in the younger age classes (Bailey 1972, Knick et al. 1985), and it has been suggested that later starting or extended seasons will likely produce more males (Tumilson and McDaniel 1988).

Some land use changes that can have a negative impact on bobcat populations include habitat and cover type changes, increased road densities, increased negative human activities, and extractive industry (Nielsen and Woolf 2000). Nielsen and Woolf (2000) assessed the impact of human land use in areas with high human density (17.8 persons/km²) and found that bobcats selected core areas as refugia from human activities and not for preferred habitat. This avoidance was hypothesized to adversely impact bobcat populations if humans continue to inhabit areas with preferred bobcat habitats (e.g., forested habitats, Nielsen and Woolf 2000). Land use changes are likely to transform preferable bobcat habitat cover types to less preferable if human occupation continues to increase (Nielsen and Woolf 2000). Land use changes can also affect prey densities and denning site availability, increase road densities and human activities that can negatively affect bobcat behavior (Nielsen and Woolf 2000). Public policy shapes management decisions and human activities are expected to increase; human dimension aspects will likely outweigh ecological considerations when formulating management strategies in the future (Nielsen and Woolf 2000).

BOBCAT CONSERVATION AND MANAGEMENT

Wild felids play an important regulatory role in the ecosystems they inhabit (Loveridge et al. 2010). There are several limitations to population persistence including habitat loss or fragmentation, habitat quality variation, forest/habitat successional stage, habitat proximity, habitat size, environmental conditions, and species reproductive rates (Ruggiero et al. 1994, Nielsen & Woolf 2002, Riley et al. 2003). Wild felid densities have been closely linked to prey abundance and loss of prey species from unmitigated harvest or habitat loss (Loveridge et al. 2010). Habitat loss, from land use changes such as increased agriculture and other human activities, is one of the prominent factors affecting felid perpetuation (Loveridge et al. 2010). Anthropogenic sources of mortality (e.g., hunting or trapping, road kill, poaching, and problem animal control) have been found to be additive to natural mortality levels and sources, substantially increasing the levels of mortality in areas of exploitation (Loveridge et al. 2010). The viability of a population is the probability that a population will not go extinct in a particular number of years given the current population size (Ruggiero et al. 1994, <http://warnercnr.colostate.edu/~gwhite/pva/index.htm>). Population viability analysis is used to predict species extinction in addition to management strategy comparisons and possible effects of habitat loss on long-term viability (Ruggiero et al. 1994, <http://warnercnr.colostate.edu/~gwhite/pva/index.htm>). Several variables can be used when measuring the population viability of a species, such as population size, habitat use, space use, genetic variability, and the population demographic ratio (Ruggiero et al. 1994).

The North American model of conservation and management is based on use of hunting and trapping by the general public as controlled by the state governmental system, otherwise known as harvest management (Connelly et al. 2005). Economics and wildlife management are deeply entwined and play important roles in the conservation of wildlife and their habitats (Woolf and Hubert 2001). Equitable harvest opportunities remain a major objective of harvest management programs that are based on the presumption that harvesting will have a small impact on populations (Connelly et al. 2005). Determining the level of harvest for populations within a particular state depends on the population dynamics and the long-term management goals (Connelly et al. 2005). An inventory of populations, identification of population and harvest goals, and the development of goal-meeting regulations are the three concepts/steps for developing and maintaining harvest management programs (Connelly et al. 2005, Strickland et al. 1994). Harvest programs generally aim to keep population levels stable or to increase or decrease populations based upon inventory data (Connelly et al. 2005).

Wildlife management is based on eight principles that assess how harvest affects mortality rates of a population in subsequent breeding cycles: additive mortality, compensatory mortality, diminishing returns, doomed surplus, harvestable surplus, inversity, opening day phenomenon, and threshold of security (Connelly et al. 2005). Loveridge et al. (2010) reported that harvest tends to be additive to natural sources of mortality for wild felids, such that total mortality ends up being greater than if harvesting did not occur (Connelly et al. 2005). As a wild felid and a furbearer that is harvested throughout the United States, the bobcat may certainly fall under this umbrella principle of additive mortality, making monitoring of populations important for accurate

population inventories and subsequent management. Uncertainty of harvest impacts on the status of wildlife populations can impede meeting population management goals (Connelly et al. 2005). Environmental variation, uncertainty of how harvest will affect future population size (structural uncertainty), and partial observability (i.e., indirect sign methods of observation), and management control all affect management decisions and their subsequent outcomes (Connelly et al. 2005). Only through thorough research and effective enforcement of regulations can wildlife populations be properly managed (Connelly et al. 2005). Bobcats have withstood dramatic changes in land use and a period of high exploitation during the 1970's and 1980's which attests to the species resiliency and management effectiveness (Woolf and Hubert 2001).

In the U.S., bobcat management falls to state wildlife management agencies, but because of their elusive nature monitoring may be difficult at best despite being the top predator in some ecosystems (Conner et al. 2001, Roberts and Crimmins 2010). Forty out of 48 states had no management program nor protection for bobcats as recently as 1971 (Roberts and Crimmins 2010). Harvest rates of 20% have been reported as sustainable, though if reproductive rates are low or natural mortality rates are high, harvest rates below 20% may cause population decline (Knick 1990, Fuller et al. 1995, Roberts and Crimmins 2010). Of course this suggested rate is only useful if a population estimate is already known and if population affecting factors such as poaching, disease, and prey abundance can be estimated (Fuller et al. 1995). Currently the United States holds approximately 71% of the suitable bobcat habitat, and bobcat populations are more widely distributed than in previous decades (Roberts and Crimmins 2010). Increases in abundance and distribution are likely attributable to factors including habitat availability,

increased prey density, changing land-use practices, and intense harvest management (Fuller et al. 1995, Roberts and Crimmins 2010). Most states have reported increasing or stable populations from monitoring via methods such as population models, archer surveys, hunter surveys, harvest data, field studies, scent-post surveys, sign-station surveys, public sightings, and detection dogs; harvest data analysis and surveys were most commonly used for monitoring because of the relatively low cost for large geographic scale data (Roberts and Crimmins 2010). As reported by Roberts and Crimmins (2010) the U.S. bobcat population estimate ranges from 2,352,276 to 3,571,681 and is likely an underestimation. The population increase country-wise is likely due to changing agricultural and land-use practices, range expansion, and habitat improvement programs as well as improved state monitoring and management programs (Roberts and Crimmins 2010).

THE BOBCAT AND KENTUCKY

Wild felids can be considered economic assets when used sustainably through trophy hunting, commercial exploitation, or tourism (Loveridge et al. 2010). Since the inception of CITES in the 1970's, when trade in threatened wild felids was restricted, the market for non-threatened felid fur experienced a boom (Fuller et al. 2005). Traders in the U.S. turned to bobcats as a source of legally obtainable fur for the growing market (Woolf and Hubert 2001). While the fur market offers a form of livelihood for economically depressed regions such as those found in southeastern Kentucky (<http://quickfacts.census.gov/qfd/states/21000.html>, accessed 9 October 2012), it also created an opportunity to expand upon research in order to meet the criteria set forth by CITES. Bobcats are currently listed in Appendix II of CITES, which requires proof that

trade is controlled and not detrimental to populations

(<http://www.cites.org/eng/resources/species.html>, accessed 9 October 2012). Black market trade in restricted spotted wild felid fur is of concern since it is difficult to discern between species when viewing a pelt. Bobcats primarily fall under Appendix II because of their "look-alike" status (Woolf and Hubert 2001). Only five wild felids of least concern conservation status exist worldwide and they are all spotted in phenotype, yet many more threatened Appendix I categorized spotted wild felids persist in all regions of the world (<http://www.wildcatconservation.org/conservation-status-of-wild-cats>, accessed 9 October 2012; <http://www.iucnredlist.org/details/12521/0>, accessed 20 October 2012). In the last 8 years harvest numbers of bobcats in Kentucky have followed an increasing trend from 1437 in 2004 to 2417 in 2011, for a total of 15,511 bobcats harvested statewide (<http://fw.ky.gov/app1/bobcatresults.aspx>, accessed 9 October 2012), indicating a growing economic sector in the state.

Incorporating human dimensions into ecosystem management has been recognized as important for developing appropriate wildlife management plans (Maehr et al. 1999). The people of southeastern Kentucky are considered to be independent and distrustful of governmental programs (Maehr et al. 1999). This region tends to be economically depressed, with a majority of the land being privately owned (Maehr et al. 1999). The average median household income for southeastern KY from 2006 to 2010 was \$21,046.00 and the percentage of persons below the poverty level was 28%, while the percentage below the poverty level for the state was only 18% for the same time period (<http://quickfacts.census.gov/qfd/states/21000.html>, accessed 9 October 2012). The biggest parcels of privately owned land are held by coal mining companies. Strip mining

and timber harvest continues to be the prevalent industry in this section of Kentucky (Maehr et al. 1999). These industries have extensive landscape influences by clearing forests and creating topographically simpler grasslands from reclaimed mines surrounding successional growth forest islands (Maehr et al. 1999). Bobcat pelt prices for 2012 ranged from \$68.00 to \$380.00, with an average price of \$224.00 USD per pelt (<http://trappingtoday.com/index.php/category/fur-prices>, accessed 9 October 2012). The potential total average income from pelt sales for the state, based on the average pelt price, could have been over \$540,000.00 USD for 2012, assuming all pelts trapped in the 2011 harvest season were sold. For an area with a very high rate of poverty, pelt sales have the potential to provide income to struggling families. This can become a motivator for illegal harvest, which is one of the reasons bobcats remain under CITES Appendix II, despite requests to down-list and remove the requirement of federal permits for trade (<http://www.cites.org/eng/resources/species.html>, accessed 9 October 2012). As previously mentioned, Nielsen and Woolf (2000) found that bobcats seek refugia away from human activities that may include extractive industry as well as harvesting activities. Given the reported additive effect of harvest on subsequent breeding populations, having an appropriate monitoring and management plan for bobcats becomes critical for species perpetuation and continued economic benefits of bobcat harvest to southeastern Kentucky.

Roberts and Crimmins (2010) performed a survey of all 50 state wildlife management agencies regarding their respective bobcat population monitoring and management plans. Kentucky reported using only harvest analysis to monitor bobcat populations and reported a habitat extent of 102,896 km², with an increasing population status from 1981

to present (Roberts and Crimmins 2010). Though the study authors reported a current statewide population estimate of 14,000 for Kentucky (Roberts and Crimmins 2010), the department does not support that reported estimate (L. Patton, 2012, Wildlife Biologist, Kentucky Department of Fish and Wildlife Resources [KDFWR], personal communication). Additionally, the state department does not currently have a management plan for bobcats or for any other furbearer species (L. Patton, 2012, personal communication). Bobcats are managed in 37 states and all but one state controls harvest via combinations of season length, bag limits, harvest quotas, and harvest method restrictions (Woolf and Hubert 2001). In 1987, an experimental harvest quota of 400 (less than 10% of the estimated population for the state) was implemented though was not met during this initial season (Frederick et al. 1989). Harvest limits in Kentucky are currently set at five per licensed individual, three of which may be taken by gun; the harvest season begins 12 November and runs through the end of February, and bobcats may be trapped throughout the diel period, but may only be hunted during daylight hours (<http://fw.ky.gov/pdf/deerguide1213smallgame.pdf>, accessed 28 Sept. 2012). General interest in bobcats as a predator and their economic value as a furbearer are what initially attracted research (Woolf and Nielsen 2000) and its economic value and CITES listing are what continue the need for research and development of a management plan for Kentucky.

PROJECT OBJECTIVES

Population estimation and trend analyses are critically important exercises for sustainable harvest and management of many game species. Animals that occur at low densities, exhibit elusive behavior, or that are wide-ranging pose economic and logistical

challenges to wildlife managers attempting to monitor them (Loveridge et al. 2010).

While animal surveys of various types (e.g., track counts, scent stations) have traditionally provided data on species occurrence and relative abundance, newer tools such as genetic analyses and radio collars incorporating Global Position Systems (GPS) and cell phone technology provide greater insight into questions regarding animal ecology and population management.

Whitaker (1988), Penry (1988), and Painter (1991) used VHF radio-collars to examine bobcat space use patterns and habitat use in both eastern and western Kentucky. These studies examined the potential for a harvest season in light of known bobcat populations in Kentucky. As a result, an experimental quota season began in 1987 (Frederick et al. 1989). Subsequent harvest data indicated bobcats had continued to increase and expand statewide (L. Patton, 2009, personal communication). Other than harvest statistics, no data on bobcats in Kentucky had been collected since those collected in the 1980's, as reported by Frederick et al. (1989). By 2008, the population dynamics of bobcats in Kentucky needed to be re-evaluated to improve and update current management strategies (K. Waldrop, 2008, Wildlife Director, KDFWR, personal communication).

I used both VHF radio collars and GPS collars with cellular texting technology to better understand bobcat ecology and spatial dynamics within the eastern Kentucky mixed mesophytic ecosystem. The high resolution of GPS data was the foundation for a number of planned analyses designed to address key questions about bobcat space use patterns and habitat selection.

The objectives of this study were to: 1) estimate bobcat home range sizes, assess core use areas, and estimate the amount of spatial overlap between individual bobcats, 2) investigate habitat selection at different spatial scales (e.g. home range and study area scales), and 3) compare the results generated to those of a prior study (Whitaker 1988) to evaluate potential changes in space use over time.

MARK-RECAPTURE AND ANALYSES

The Mark-Recapture method for estimating population size and other population variables such as home range and core use area size, and ultimately population density, involves capturing and marking a sample of individuals from a population, releasing, and subsequent recapture at a future point in time (Dinsmore and Johnson 2005).

There are two types of marking methods that are commonly employed for post animal-capture: permanent and non-permanent (Silvy et al. 2005). Permanent methods of marking include any technique that will last for the lifetime of the individual being marked (e.g. branding, tattoos, toe clipping, ear notching; Silvy et al. 2005). Non-permanent methods of marking are those that do not last for an individual's lifetime, such as ear tags, PIT tags, and neck collars (Silvy et al. 2005). Because marks can influence animal behavior, and may reduce survivorship, it is important to select a marking method which will supply the information needed to answer the question being investigated without negatively affecting the individual to a significant extent (Silvy et al. 2005). Neck collars are considered to be a non-invasive marking technique (Silvy et al. 2005), and can be equipped with VHF and GPS transmitters to provide a non-invasive method for collecting recapture data (White and Garrott 1990). Animals marked with radio-

collars can be more frequently and consistently observed (relocated) than those marked using different methods (Fuller et al. 2005).

TELEMETRY TECHNIQUES AND CAVEATS

Traditional VHF radio tracking has been extensively used to study bobcats for the purpose of understanding their habitat use patterns, predicting their occurrence, and calculating relative abundance (Nielsen and Woolf 2002, Woolf et al. 2002, Benson et al. 2004, Diefenbach et al. 2006). Radio-telemetry can provide data on movements, dispersal, migration, space use, habitat selection, resource use, population abundance, intra- and inter-specific relationships, and estimate fecundity and survival (White and Garrott 1990, Fuller et al. 2005). VHF collars require radio tracking from aircraft or rely on triangulation, the process of estimating the location of a transmitter by using two or more compass bearings obtained by using directional antennas at known locations remote from the transmitter's position (White and Garrott 1990), whereas GPS collars use a satellite-based system to obtain animal coordinates. There have been many published studies on aspects of bobcat ecology where one or both of these methods were used with mixed success throughout the United States (Burton et al. 2003; Chamberlain et al. 2003; Cain et al. 2003; Moen et al. 2003; Godbois et al. 2004; Constible et al. 2006; Plowman et al. 2006; Kiawah Island Conservancy 2007; Pruess and Gehring 2007; Lynch et al. 2008; Tucker et al. 2008).

The advent of GPS collars coupled with the spatial analysis capabilities of Geographical Information Systems (GIS) has created unique opportunities to observe animals at shorter time intervals and analyze results at finer geographic scales than previously attainable. The intensive, fine-scale data reveal animal resource use patterns at

a new level of both spatial and temporal detail. GPS collars have not been widely used on bobcats primarily because large transmitter size has restricted their use to larger mammals until recently (Fuller et al. 2005). Unlike VHF radio-telemetry, GPS telemetry systems offer a more accurate method for gathering location data (Millspaugh and Marzluff 2001, Fuller et al. 2005). GPS coupled with GSM (Global System for Mobile communications) cellular technology uses cell phone towers to transmit data from the collared individual directly to a designated computer, reducing the need for on-the-ground triangulation.

Radio-telemetry can produce biased results if the marking causes increased mortality, stress, or non-normal behavior (Fuller et al. 2005). Location error can affect the accuracy and precision of animal locations via the variability in equipment performance, animal movement, variability in radio-wave propagation, and equipment operation (Fuller et al. 2005). Objects can block radio waves, which normally remain vertical over water or flat ground, causing horizontal polarization obstructing the gain and directionality of the signal (Fuller et al. 2005). Similarly, signal strength of the radio frequency is negatively affected by distance; the greater the distance from the radio-collar the weaker the signal (White and Garrott 1990). Elevation and topography are the greatest affecting factors of frequency reception, topography can cause bounce in signal which creates inaccuracy in relocations (White and Garrott 1990, Fuller et al. 2005). It is because of these variability-causing factors, and the difficulty in quantifying their effect, that VHF radio-telemetry usually provides only an estimate of the actual location of the animal (Fuller et al. 2005).

Both Adrados et al. (2002) and D'Eon and Delparte (2005) looked at the accuracy of data from GPS radio collars. They found that 3-D differential data are more accurate

than non-differential data (Adrados et al. 2002) and that collar position (antenna angle from vertical, relative to the ground) can affect the accuracy of relocation data (D'Eon and Delparte 2005). GPS location data are no longer scrambled by the military making raw (non-differential) data very precise. D'Eon and Delparte (2005) found data to be most accurate when the collar is positioned between 0 and 90 degrees from vertical (antenna facing the sky), with terrain affecting success at these optimal angles. A significant potential source of observational bias is habitat type (Rodgers 2001). In two studies that used GPS radio telemetry, the location acquisition success for GPS was lower in forested habitats due to signal blockage by canopy cover (Land et al. 2008; Pellerin et al. 2008). Pellerin et al. (2008) noted that auto-correlated data sets from both VHF and GPS transmitters may provide better estimates of true home range size than data sets from either type independently. It is therefore important to select the telemetry technique that is most appropriate for the study species, the project objectives and inferences to be made, and for the study site, and it may be appropriate to use more than one data collection method.

SPATIAL ANALYSIS TECHNIQUES AND CAVEATS

Two of the most commonly used analysis methods for estimating home range and core use areas are the Minimum Convex Polygon (MCP) and Kernel analyses. Minimum Convex Polygon is the oldest and most used home range estimator and is a type of polygon estimator; this method creates polygons based on a percentage of the innermost locations in a data set (Kernohan et al. 2001). The kernel home range estimator is a type of probabilistic model that uses the utilization distribution (i.e., the density of locations) to accurately characterize an animal's probability of occurrence in space, over a period of

time, with contours (Worton 1989, Kernohan et al. 2001). Both MCP and kernel analyses can be used with nonparametric data sets, removing the need to meet the assumptions of normally distributed data sets (Kernohan et al. 2001). Kernel analyses are less sensitive to outliers than MCP analysis and this sensitivity can lead to misrepresentative convex polygons when an animal strays from its "normally" used areas (Kernohan et al. 2001). Additionally, the kernel estimation method is able to account for multiple centers of activity, which may be more representative of space use for a species (Kernohan et al. 2001). Different estimators can produce very different estimations of home range size (White and Garrott 1990, Kernohan et al. 2001); therefore it is important to use an estimator that is more comparable when comparing home range sizes between studies. Use of more than one estimator with a dataset, one of which should be MCP because of its comparability, is recommended (Harris et al. 1990).

Both MCP and kernel estimators are subject to many caveats and assumptions that can affect the accuracy of the result. Location data collection methods, time between locations, and total number of locations can each affect home range estimate accuracy. Sample size is one of the most important caveats for both MCP and kernel estimations. For MCP estimations a sample size of 100 to 200 locations is recommended since this estimator will underestimate home range size at small location sample sizes (Fuller et al. 2005); kernel estimates should be based on at least 30 location points, but 50 is ideal, because this estimator will overestimate home range size with a sample size of less than 30 locations for the sampling period (Kernohan et al. 2001). If sufficient numbers of locations have not been met, MCP home range estimates for animals based on data sets with different numbers of locations may not be directly comparable (Fuller et al. 2005).

Biological sample size (e.g., the number of particular time-of-day locations when a species is most active) is important for representing the movements and activities over the period from which inferences are to be drawn. Therefore it is imperative to consider a species' biology when determining appropriate data collection methods (Kernohan et al. 2001). Both MCPs and kernel methods are robust to autocorrelated location data such that statistical independence between locations becomes less important than the sampling scheme that best represents a species activities over the study duration (Kernohan et al. 2001). Additionally, the MCP estimator has no underlying assumption of point distributions and therefore may include areas unused by an individual in the estimate (Fuller et al. 2005).

Two techniques of kernel home range estimation have been developed, fixed and adaptive, and these techniques are based on how the bandwidth (smoothing) is applied to the data set (Worton 1989, Kernohan et al. 2001). Fixed kernel estimation applies the same bandwidth value to the entire data set, whereas with adaptive kernel estimation a local bandwidth is applied to each observation (location; Kernohan et al 2001). The local bandwidth becomes larger in areas with few locations and smaller in areas with many locations, producing more smoothed tails (i.e. areas with few locations) than peaks (areas with many locations; Worton 1989, Seaman and Powell 1996). Fixed kernel estimates have the advantage of being more accurate at the outer contours of the home range, while adaptive kernel estimates are more accurate at the inner contours of the utilization distribution (Worton 1995, Seaman et al. 1999). Adaptive kernels can attach uncertainty to locations at the outer contours of the data set, but the outer contours are where the least amount of locations are sequestered and are of the least biological significance for an

animal (Kernohan et al. 2001). Determining which type of kernel estimator method to use should be based on the project objectives, the species being studied, and the sampling of the distributions of locations (Kernohan et al. 2001).

Bandwidth selection for kernel space use estimation is a major caveat of this analysis method and there are many methods for selecting bandwidth (Kernohan et al. 2001). This bandwidth is considered an overall disadvantage to the kernel method of estimation as it controls the width of the kernel (Fuller et al. 2005). In kernel analysis the contour smoothing method will determine the estimator's sensitivity to outliers in a data set (Kernohan et al. 2001). The amount of smoothing that is applied to a data set is determined by the bandwidth value, which is the critical component in density estimation (Worton 1995). Use of the appropriate bandwidth value is of utmost importance because using too small of a bandwidth will cause kernels to break apart and become over-exaggerated in areas of dense and sparse-to-no use; too large of a bandwidth will over-smooth the kernels, disintegrating the contours into a flat surface as opposed to distinguishing between areas of varying use density (Kernohan et al. 2001, Fuller et al. 2005). Two of the most commonly used bandwidth selection methods for kernel analysis are optimum bandwidth and least squares cross validation (LSCV) selected bandwidth (Worton 1995, Kernohan et al. 2001). The optimal bandwidth is an estimate of the assumed true distribution if it is the bivariate normally distributed (Kernohan et al. 2001). The LSCV bandwidth selection method does not assume any type of distribution of the data set and estimates the bandwidth by minimizing the discrepancy between the estimated density and the true density based on the mean integrated squared error (Kernohan et al. 2001). The major drawback to the LSCV bandwidth method is that in

areas where locations occur at or near the same point, the bandwidth degenerates and is unable to smooth the area and may not accurately represent the distribution of locations (Kernohan et al. 2001). LSCV is the recommended bandwidth selection method as it does not over-smooth multimodal distributions of locations and therefore does not overestimate the home range estimate as the optimal bandwidth selection method does (Worton 1995, Seaman et al. 1999).

RESOURCE SELECTION ANALYSIS AND ASSUMPTIONS

Resource Selection Defined

Resource selection is the process in which an animal makes a choice from the resources that are available and this choice is considered selective if the resource is used disproportionately to its availability (Johnson 1980, Alldredge and Griswold 2006). Garshelis (2000) has criticized this definition of resource selection, stating that the equal availability of equally abundant resources may not be true in all cases. The use of a more nuanced definition of resource selection is dependent on the project objectives, data collection and analysis methods, and data quality (Alldredge and Griswold 2006).

Compositional Analysis and Assumptions

Neu et al. (1974) developed a type of compositional use-availability method for resource selection analysis, which compares the observed and expected locations for an animal to the available composition of resources or habitat types. This test method has been well established in the literature and can be used with caution on studies with small

sample sizes (Byers et al. 1984, White and Garrott 1990, Alldredge and Ratti 1986, Alldredge and Griswold 2006).

There are several assumptions that apply to resource selection studies in general, and most of these assumptions apply to the Neu et al. (1974) habitat selection method (Alldredge and Griswold 2006). Starting with the most basic assumption, resource selection studies assume a random sample of individuals. This assumption can be violated if pooling across individuals or times (Alldredge and Griswold 2006). The assumption of no spatial or temporal autocorrelation of relocations can be violated if locations are close together in time (White and Garrott 1990). If locations are not independent of each other in time then Type I errors may result, where the null hypothesis is rejected when it should be accepted (White and Garrott 1990). By performing the Bonferroni correction along with the chi-square test, the likelihood of Type I errors may be reduced by applying a more stringent set of significance values to the test (Byers et al. 1984, White and Garrott 1990). Byers' (1984) adjusted Bonferroni correction uses a 95% confidence coefficient, which is equivalent to an alpha value of 0.05. Values falling outside of the calculated confidence interval only have a 5% probability of happening by chance (Whitlock and Schuler 2009), and if the chi-square test was already significant in value (rejecting the null) then disproportionate habitat use can be reasonably inferred (White and Garrott 1990). The selection of resources made by one individual is assumed to be independent of selections made by other individuals, but this assumption can be violated if animals are territorial (Alldredge and Griswold 2006). Bobcats are known to be territorial at particular spatial scales, as denoted by areas of exclusive use (e.g., 50% or 25% core area), where habitat that is readily available for one

individual may not be available for the others (Hansen 2007). This territoriality may affect the inferences that can be made from the compositional analysis.

Available habitat is assumed to be the same for all individuals and it is assumed to be known without measurement error when partitioning a map into habitat types (White and Garrott 1990, Alldredge and Griswold 2006). By using a digital ortho-photo file comprised of mosaics of digital ortho-photo quarter quads of counties, with a very small ground sample distance (e.g. 1-m x 1-m), any measurement error associated with the habitat classification can be minimized to meet this assumption. Habitat availability is assumed to be constant over time, but this assumption can be violated if availability changes over seasons, or if habitat availability is increased or decreased during the study period (Byers et al. 1984). Any seasonal changes during the study duration that are not quantified would violate this assumption. Used habitats are assumed to be classified correctly (Alldredge and Griswold 2006), but radio-telemetry error can cause classification errors and the effects of telemetry error on habitat classification ought to be quantified for use in making inferences (Erickson et al. 2001).

CHAPTER 2

THE HOME-RANGE CONCEPT – LITERATURE REVIEW

The concept of home range has been frequently used by wildlife biologists since its formal definition in 1940 (Burt 1940). Though a concept, today home range is defined as the extent of area with a defined probability of occurrence of an animal during a specified time period (Feldhamer et al. 2007). Home range size is dependent on habitat/resource quality and animal size (Kernohan et al. 2001, Feldhamer et al. 2007). The definition has been subdivided into more specific terms, such as core area, territory, and utilization distribution. Utilization distribution is the probability distribution defining an individual's use of space over a period of time (Millsaugh and Marzluff 2001, Feiberg and Kochanny 2005). The core area of a home range is the area most heavily used by an individual, is an area with a nonuniform pattern of use, and a type of utilization distribution of heavy location density (Kernohan et al. 2001, Feldhamer et al. 2007). Territory is an area of exclusive occupation by an individual that is defended by overt aggression or advertisement (Feldhamer et al. 2007). Food, mating areas, and other competitively sought resources may be guarded within a territory (Feldhamer et al. 2007). Home range size and sizes of these other more-specifically defined areas have been used to estimate population density (Frederick et al. 1989), and in conjunction with other measures of space and habitat use studies, are fundamental to making informed management decisions. In order to understand the importance of the home range concept in modern wildlife biology it is imperative to delve into its history. With this in mind, I will outline the home range concept, methodologies developed to apply it, and how its application has in turn brought about an evolution in the concept itself over time.

In 1940, W.H. Burt published a manuscript (Burt 1940) that clarified the use of home range by defining it as follows: “the home range of an animal, as here defined, is that area about its established home which is traversed by the animal in its normal activities of food gathering, mating, and caring for young. It excludes those areas traversed by vagrants or other individuals in search of home sites.” At the time, mark-recapture studies were prevalent for determining species occurrence and deriving population estimates. Burt (1940) recognized the issues of error when trying to estimate home range size from trap data, calculating area from irregularly shaped ranges, and dealing with seasonal changes in home ranges. In his follow-up publication (Burt 1943), he distinguished between territory and home range, which previously had been used synonymously. He recognized two types of territories (sections of the home range), those for denning or food resources, both of which an animal may behaviorally defend with aggression (Burt 1943). He also included in his definition of home range the possible variances in home range size for different sexes, ages, seasons, population densities, and degree of home range overlap (Burt 1943). The idea of an “amoeboid” shaped home range was introduced in this publication, highlighting the unlikelihood of geometrically shaped home ranges (Burt 1943).

Mohr (1947) used Elton’s (1932) definition for economic density, which is the number of individuals per acre within their territories, in conjunction with the definition for home range from Burt (1943). Mohr (1947) reported on several home range estimation methods, all from mark-recapture trapping and all of which, except for the method of Burt (1940), employed a grid strategy. The home range definition was refined with the addition of minimum home range, which is the area within the convex polygonal

boundary (the polygon created by drawing straight lines from one trap to another of the outermost traps) of the trap array (Mohr 1947). The grid-estimated home range was the area covered by grid sections where individuals were caught. Grid size, which is dependent on the distance between traps, may have an effect on the estimated home range such that smaller grid sizes may be more accurate in estimating home ranges than larger grid sizes. However, grid estimates of home range may be less accurate than minimum home range estimates due to over-estimation (Mohr 1947).

Mohr (1947) suggested that habitat availability may have an effect on home range size, but did not go beyond the idea of habitat uniformity. Hayne (1949) covered different methods used to calculate home range size from mark-recapture data. It was here that the idea of intensity-of-use was correlated with estimations of home range by describing the implicit assumption of equity of habitat use (Hayne 1949). Hayne (1949) discussed 3 classes of methods for calculating home range size from mark-recapture data: 1) methods that used the area enclosed by trapping points, 2) methods that added a boundary area to compensate for the lack of traps within the boundary, and 3) methods that incorporated the greatest distance between trap locations as the diameter of a circle, or axis of an ellipse, as the home range (Hayne 1949). The polygon formed from the first class of methods can be measured for area, which has been called the minimum home range size, and was considered to be a conservative estimate (Burt 1943, Hayne 1949). The cons to this method purported at the time were: a) the unlikelihood of an individual's home range being solely located within the trap polygon, and b) the home range estimation can only be made for individuals that were trapped on >2 nights, in non-straight trap-lines (Hayne 1949).

The second class of methods for calculating home range size from mark-recapture data employed a buffer of half the distance between trap points as the home range boundary to meet the assumption that an individual's home range is more likely to extend beyond the trap area (Hayne 1949). The tradeoff of this method acknowledged by Hayne (1949) was the uncertainty of an animal's actual home range extending beyond the minimum home range estimate. The greatest distance-ellipse method assumed that the home range is elliptical in shape and that the greatest distance between trap locations is the actual maximum distance traveled by the individual (Hayne 1949). All three methods assumed that the animal did not use areas of the trap grid in which it was not captured (Hayne 1949).

As an alternative to the preceding methods, the "center-of-activity" calculation as suggested by Hayne (1949), used the average geographic center of all recapturing locations. The center-of-activity was not a home range estimate, but was used to find the distance from the center of recapture points to the most distant location where the animal was trapped. Locations where individuals were captured more than once were weighted with greater importance (Hayne 1949). Hayne (1949) found a greater percentage of recaptures at the same trap site occurred at the center than at the periphery of the trap area. This was the earliest description of utilization distributions. Home range was redefined by this method as the area wherein an individual is captured with varying frequency. Unlike previous methods, the center-of-activity procedure viewed the home range as variable and not discrete (Hayne 1949).

Although centered on bird ecology, the publication by Odum and Kuenzler (1955) discussed issues of home range measurement and standardization of measurement

methodology, as well as the testing of home range concepts previously developed under Burt (1943). The common method for determining bird home range and territory size was from direct observations over time; plotting location points on a map during the breeding season (Odum and Kuenzler 1955). The authors made a distinction between maximum territory (home range) and utilized territory, where maximum territory is the polygon created by the outermost observational points; and utilized territory is some portion of the area bound by the polygon lines, the size of which is dependent on various important habitat variables (Odum and Kuenzler 1955). For home range size (maximum territory) two methods of measurement were discussed: 1) observation-area curve, and 2) activity radius (Odum and Kuenzler 1955). The observation-area curve is an adaptation of the species-area curve, which determines the number of observations needed for measuring home range size before diminishing returns (Odum and Kuenzler 1955). The authors suggested that this method may make interspecies comparisons obtainable via standardization and that one may be able to track changes in home range size over time (Odum and Kuenzler 1955). The activity radius method is essentially the same as the center-of-activity method offered by Hayne (1949, Odum and Kuenzler 1955).

In one of the first publications on tracking wildlife by radio-telemetry Cochran and Lord (1963) developed the telemetry system and tested it on several different small mammal species. Most radio-telemetry relies on triangulation, the process of estimating the location of a transmitter by using two or more compass bearings obtained by using directional antennas at known locations remote from the transmitter's position (White and Garrott 1990). The suggested accuracy verification method at the time was direct animal flushing from the location estimated from the bearings taken; the original

investigators reported an accuracy of 10-m for collared rabbits (Cochran and Lord 1963). Effects of stationary versus moving animals on bearing acquisition for a location were also discussed and the authors suggested that two receivers/operators are required with moving individuals, while one receiver is sufficient for a stationary animal (Cochran and Lord 1963). While movements were mapped, the authors did not attempt to estimate home range size from continuous tracking data. This technology allowed for the direct observation of animal locations, leading eventually to estimating home range size and shape from radio location data.

Jennrich and Turner (1969) used capture data to assess popular home range estimate methods; both minimum convex polygon (previously termed: minimum home range) and activity radii methods were evaluated, and a new method, the covariance determinant method, was explored. The authors suggested the use of either the determinant method or the minimum convex polygon method (with large enough sample size); but objected to the use of the activity radii method for determining home range size (Jennrich and Turner 1969). The covariance determinant method estimates home range size by using capture-mark-recapture (CMR) data with an equation that does not assume a circular home range shape, but, instead, an elliptical shaped boundary. They claimed the covariance procedure significantly reduced bias in the home range estimate (Jennrich and Turner 1969). Circular normal estimates were suggested to overestimate home range size if space use was not unimodal. The minimum home range method was renamed the minimum convex polygon (MCP) method for estimating home range size; it was suggested the MCP only be used with large sample sizes because of the extensive bias incurred from small sample sizes (Jennrich and Turner 1969). Jennrich and Turner

(1969) refined the utilization distribution definition to be the smallest elliptical area that describes 95% of an individual's use of the habitat. This was one of the first mathematically characterized definitions of home range published that used the still popular CMR method of data sampling.

McNab's (1963) paper was one of the first to link body size to home range size, creating a basis for resource use to influence the study of home range in wildlife research. The energy required for an animal to live, which is a function of body size (body weight), may dictate its home range size. The author postulated that a larger food requirement will necessitate a larger home range (McNab 1963). McNab (1963) concluded body weight does indeed determine home range size, when looking at a log transformed slope of weight to home range size. McNab (1963) also predicted that environmental conditions, population density, and habitat and resource quality, availability, and distribution as a function of metabolic rate will affect home range size.

CHANGING THE CONCEPT: RESEARCH OF THE 70'S AND 80'S

The early 1970's was marked by the continued use of CMR data to estimate home range size, as exemplified in Koepl et al. (1975) and Van Winkle (1975); it was not until later in the decade that radio-telemetry became the preferred method of data collection. Koepl et al. (1975) used the traditional Burt (1943) definition of home range. Van Winkle (1975) redefined utilization distribution as the bivariate relative frequency distribution of positional location data for an individual over time. While Koepl et al. (1975) explored a modification of the determination of covariance method (Jennrich and Turner 1969) and the center of activity with confidence ellipses for bivariate normal data method (Hayne 1949, Koepl et al. 1975); Van Winkle (1975) introduced univariate and

bivariate home range models of utilization distribution by the theoretical probability of locations. Van Winkle's (1975) models are essentially an extension of Hayne's (1949) publication on center of activity home range estimation and a test of Jennrich and Turner's (1969) elliptical bivariate home range model. Resource use was not considered in the calculation of home range size or shape by Koepl et al. (1975). And although Van Winkle (1975) assumed a homogeneous habitat, he acknowledged that animals are not likely to move at random within habitats, and highlighted the need for statistical analyses of these parameters.

By the late 1970's radio-telemetry was gaining ground within the wildlife literature; Dunn and Gipson (1977), Harestad and Bunnell (1979), Damuth (1981), Byers et al. (1984), Samuel et al. (1985), and Worton (1987, 1989) all used radio-telemetry as the main method for location data collection. However, the way home range was defined in each of these publications varied greatly. Dunn and Gipson (1977) used Jennrich and Turner's (1969) definition; while Harestad and Bunnell (1979) and Schoener (1981) used Burt's (1943) original interpretation of home range. Damuth (1981) and Byers et al. (1984) did not explicitly define home range, but Anderson (1982) and Worton (1987, 1989) used utilization distribution when characterizing the concept; only Samuel et al. (1985) focused solely on a sub-area of the home range concept by assessing core areas.

The previously developed methods for analyzing CMR data for home range properties are inappropriate for radio-telemetry location data because of a lack of independence between locations (Dunn and Gipson 1977). McNab's (1963) work linking home range size with metabolic rate was continued by Harestad and Bunnell (1979) and Damuth (1981), who agreed with McNab's conclusion that home range size (using MCP)

is proportionate to body mass (and metabolic rate) and is dependent on the distribution of food resources. However, Damuth (1981) pointed out an exception for herbivores in which an individual's home range estimate may be affected by the way the entire herd utilizes the available resources.

Anderson (1982), Schoener (1981), and Worton (1987, 1989) all considered how varying use of habitat within a home range can affect estimates of home range size. Both CMR and radio-telemetry methods of data collection were employed, while MCP, regression, chi-square, kernel, and Fourier transformation methods were used for assessing relationships between home range size and habitat use (Schoener 1981, Anderson 1982, Worton 1987, 1989). Schoener's (1981) work was an extension of Jennrich and Turner's (1969) regression method for assessing utilization distribution for home range shapes other than bivariate normal or unimodal circular. The MCP method was then evaluated by using the r^2 statistic, which the author called the "R-method," concluding an $n \geq 25$ would eliminate sample size bias; and suggesting that MCP estimates are close to the true home range at that sample size level (Schoener 1981). Schoener (1981) attempted to resolve issues of home range size and habitat utilization by improving previously used methods of home range estimation. This was an important paper that redefined home range shape and examined the effects of heterogeneous habitat use on size estimates. Home range sizes derived from the R method were found to be $\leq 50\%$ of previous estimates from older methods, especially those using utilization distributions.

Anderson (1982) used a Fourier transformation nonparametric procedure for estimating home range size based on the utilization distribution and a density function,

eliminating sample size bias and without assuming a particular home range shape. Results suggested that this method more accurately described the actual spatial distribution than previous methods like MCP; which includes areas of the home range not necessarily used. Anderson (1982) suggested using MCP as a quick and easy method for estimating home range size. He highlighted the importance of the assumption of statistical independence of observations that is inherent in all home range size estimation methods, noting that there is no current method for testing for independence of successive observations (Anderson 1982). Worton (1987) compared several home range estimation methods and concluded that Anderson's (1982) Fourier transformation method and the kernel method were the most appropriate.

Nonparametric kernel methods, fixed and adaptive, for home range estimation were first described in Worton (1987) and later expanded upon (Worton 1989). A kernel is a probability density, which is applied over a location when conducting a kernel analysis to determine space use by an individual (Seaman and Powell 1996). Kernels are related to utilization distribution in that they incorporate the density of locations to evaluate the relative probability of finding an animal at a given location, such that around each location there are regions that contain a likelihood of animal presence. In areas with high kernel densities, more locations are present and contributing to that density, and areas with low densities have fewer locations present (Seaman and Powell 1996). Kernels include probability density contours to depict different percentages of the home range area, as well as core use areas, without relying on parametrically distributed locations for kernel shape (Worton 1987, 1989). Fixed kernel methods use the same kernel width

between locations; whereas adaptive kernel width increases as the distance between kernels increases (Millspaugh and Marzluff 2001).

Byers et al. (1984) attempted to link home range space use, but not size, with habitat type by assigning each location a vegetation type. A chi-square test was used to test between expected and actual vegetation type use, where expected was based on the availability of vegetation types within a defined area (Byers et al. 1984). This test assumes that animals observed (aerial locations) in a vegetation type are actually using (foraging, resting, etc.) that vegetation, and temporal independence of location data.

Samuel et al. (1985) investigated the relationship between core use area size and methods of estimating home range size. The term “core area” was coined by Kaufmann (1962) as the areas within a home range receiving concentrated use by resident animals, but Samuel et al. (1985) modified the definition to the spaces of a home range that show a greater than equal use pattern. Comparisons between observed space-use and expected use based on the assumption of a uniform pattern of use led to the insight that the size and location of core areas are dependent on the methods of home range size estimation; which can change the boundary and distribution of core areas within the home range (Samuel et al. 1985).

CHANGING THE CONCEPT: RESEARCH IN THE 90’S AND BEYOND

Beyond the 1980’s a shift in the technology used to assess the home range concept took place, away from CMR to an almost exclusive use of telemetry. The use of Burt’s (1943) definition of home range and its extension in utilization distribution, however, is still prevalent. Most studies focused on habitat relationships with home range, such as habitat use effects on home range size or shape (Aebischer et al. 1993), sample size

effects on home range size (Girard et al. 2002), methods of estimating home range size (Borger et al. 2006), and the usefulness of auto-correlated location data for analyzing habitat-use within a home range (Kie et al. 2010).

Aebischer et al. (1993) supported analysis of proportional habitat use by independent individuals with MANOVA/MANCOVA compositional analyses, and found that small sample size and non-independence of locations are important sources of bias. Habitat use was estimated by the proportion of radio-locations within each habitat compared to the proportion of home range area occupied by each habitat type, i.e., the available habitat (Aebischer et al. 1993). They concluded that home range estimates are not a true representation of the available habitat, but are useful in delimiting utilization (Aebischer et al. 1993).

Girard et al. (2002) assessed the minimum number of locations (sampling effort) needed for MCP as well as kernel home range estimates when using GPS telemetry location data. The effect of increasing number of locations on home range size estimates identified the need for 100 to 300 locations annually, and 30 to 100 locations seasonally per individual to reach an asymptotic value (Girard et al. 2002). Borger et al. (2006) assessed both fixed and adaptive kernels, at 10 locations per month over a standardized number of days, for variations between individuals and study sites. Most of the variance in home range size came from inter-individual variation when performing a repeated measures analysis of home range size. Either of the kernel methods were statistically unbiased when radio-sampling was standardized over time and, unlike past claims (Kernohan et al. 2001), were shown to be efficient and robust (Borger et al. 2006). Long-term studies with increased sample size (n = individuals, not locations) and number of

study sites were recommended in order to limit potential sources of bias that are prone to clouding other, more interesting, related biological questions (Borger et al. 2006).

The idea of using data to answer more interesting biological questions was expanded upon in Kie et al. (2010), where they redefined home range as an area of some value that an animal knows and remembers when traversing a landscape of varying resources.

When considering kernel estimates, areas that may not be used much (or at all) may be included in estimates if close to greatly used areas of the home range (Kie et al. 2010).

To overcome this issue, the authors suggest the use of mechanistic models for home range analysis which take into account an animal's movement patterns as a function of the spatial distribution of limiting resources (Kie et al. 2010). These mechanistic results can then be used to predict how an animal's space use will change with changing population size or resources, or the identification of important resources (Kie et al. 2010). Context continues to play an important role in determining biological significance of data generated by this or any other space use methodology. This suggests that when analyzing fine scale data where many location points are taken, it may be more useful to look at paths of movement for identifying significant areas or features within the home range.

CHAPTER 3

METHODS

STUDY REGION

The Cumberland Plateau is partially located in eastern Kentucky and is characterized by steep and rugged terrain containing second and third growth deciduous forests (Bowling 2009). There are several watersheds within the Cumberland Plateau including the Licking river, Ohio river tributaries, the Kentucky river and its tributaries, and the Upper Cumberland river (Simpson and Florea 2009), forming dendritic drainages with serpentine narrow ridges and steep valley walls (side slopes ranging from 35 to 90 percent), with narrow stream bottoms, in the eastern portion of the plateau (Coltharp and Springer 1980, Hurst and Lacki 1997). The plateau is part of the broader Appalachian Mixed Mesophytic ecoregion, and the forests of the Cumberland Plateau are some of the most diverse in the continent (Dale et al. 2009). The mixed mesophytic forests of this area supported dominant overstory species which included American beech (*Fagus grandifolia*), yellow-poplar (*Liriodendron tulipifera*), American basswood (*Tilia americana*), sugar maple (*Acer saccharum*), northern red oak (*Quercus rubra*), white oak (*Q. alba*), eastern hemlock (*Tsuga Canadensis*), and yellow buckeye (*Aesculus octandra*) (Braun 1950). Understory trees included dogwood (*Cornus florida*), magnolia (*Magnolia sp.*), sourwood (*Oxydendrum arboretum*), striped maple (*Acer pennsylvanicum*), redbud (*Cercis canadensis*), ironwood (*Carpinus caroliniana*), hop-hornbeam (*Ostrya virginiana*), holly (*Ilex opaca*), and service-berry (*Amelanchier arborea*) in moist, well drained sites (Larkin et al. 2003, Dahl 2008). Oak-hickory (*Quercus carya*), oak-chestnut

(*Q. castanea*) and oak-pine (*Q. pinus*) climax communities were supported in areas of rocky soil, ridge tops, and upper southwestern slopes (Larkin et al. 2003, Dahl 2008). Though eastern Kentucky has a low human population density with most of the human development being concentrated in lower elevations near river bottoms, approximately 90% of the region is privately owned (Maehr et al. 1999, Dahl 2008).

Land use in southeastern Kentucky has impacted the forests of the Cumberland Plateau, creating a mosaic of forest, shrub- and grasslands, from what was once continuous forest (Maehr et al. 1999, Dale et al. 2009), as a result of state and federal mining reclamation laws (Maehr et al. 1999). The largest areas of privately held land are owned by commercial mining companies and it is their activities which has changed the landscape to a more simplistic grassland (Maehr et al. 1999). Surface mining accounts for approximately 10% of the southeastern Kentucky landscape, while agriculture, secondary successional forest, and urban areas account for the remaining 90% (9%, 80%, and 1% respectively, Bowling 2009). Road densities remain low in this section of the state (Maehr et al. 1999).

STUDY SITE

The Paul Van Booven (PVB) wildlife management area (WMA) is a reclaimed surface mine characterized by deep v-shaped valleys and steep slopes typical of the eastern section of the Cumberland Plateau, with forested ravines and some forested hilltops (Whitaker 1988, Dahl 2008). Past surface mining for coal eliminated many ridge-tops from the area and the elevation ranges from 225 to 470 meters, creating areas with flat-topped mesas and gently sloping grasslands (Whitaker 1988, Larkin et al. 2003). The dominant vegetation on the site is a mix of grain annuals, legumes, and trees that

were planted as part of the post-mining restoration including Kentucky-31 tall fescue (*Festuca arundinacea*), autumn olive (*Elaeagnus umbellata*), bush clover (*Lespedeza* spp.), birdsfoot trefoil (*Lotus corniculatus*), crown vetch (*Coronilla varia*), perennial ryegrass 11 (*Lolium perenne*), orchardgrass (*Dactylis glomerata*), black alder (*Alnus glutinosa*), and black locust (*Robinia pseudoacacia*) (Whitaker 1988, Larkin et al. 2003, Dahl 2008). The WMA is part of the Starfire Complex and much of the surrounding properties are privately held and sections are active, inactive, or reclaimed surface mines (e.g. the International Coal Group property and the Big Elk Mining Company property; see Dahl 2008 Fig 2.1).

The climate is characterized as humid temperate continental with warm summers and cool winters (Hill 1976, Bowling 2009). Annual temperature during the study averaged 57 °F, with a range from 28 to 77 °F (www.ncdc.noaa.gov/cdo-web/results). Total precipitation averaged 56 inches with a range of 1.92 to 7.34 inches; snowfall accounted for an average of 17.8 inches of the total annual average precipitation (www.ncdc.noaa.gov/cdo-web/results, accessed 9 October 2012).

TRAPPING AND MARKING

Bobcats were trapped according to the protocol outlined by Whitaker (1988). Trapping began in March 2010 and continued through April 2010, for a total of 59 days. Bobcats were captured by using padded number 2 double-coil spring steel leg hold traps (Softcatch #2, Oneida Victor Inc. Ltd., Euclid, OH; Schemnitz 1994). Various lures and baits (e.g., beaver [*Castor canadensis*] castor, muskrat [*Ondatra zibethicus*] carcasses) were used to increase trapping efficiency (Frederick et al. 1989). Trapping in the study area continued until the trapping success decreased over time, to an extent that indicated

the majority of adult bobcats had been captured. Traps were checked twice daily when possible, once in the morning and once before dusk, in order to reduce potential stress to trapped individuals (Gannon et al. 2007). I used Telazol^R to immobilize trapped animals (5 mg drug/kg body weight), delivered by intramuscular injection; the dosage was based on the estimated in-field weight (Kreeger 1996, Lovallo and Anderson 1996a, Shindle and Tewes 2000). Handling of bobcats followed American Society of Mammalogists (ASM) guidelines (Gannon et al. 2007) and standard individual measurements, including sex, weight, total body length, ear length, and tail length were recorded (Knick 1990). Each cat was marked with an individually-numbered, glass-encapsulated, passive integrated transponder (PIT model 1400 or 1406, Biomark, Inc., Boise, ID; Gannon et al. 2007). Procedures used in this study were reviewed by the Eastern Kentucky University Institutional Animal Care and Use Committee and approved as Protocol 10-2009.

Depending on the size of the animal, I placed either a global positioning system (GPS) or very high frequency (VHF) radio collar on a bobcat for subsequent tracking by radio-telemetry. Lotek Wildcell SL GPS/GSM radio collars (Lotek Wireless, Inc., Newmarket, Ontario, Canada) were attached to 2 male bobcats, based on the collar weight being < 4% of the cat's bodyweight (Gannon et al. 2007). Five female bobcats were collared with lighter weight VHF units (Lotek Wireless, Inc.). GPS radio-collars were programmed to record GPS coordinates once every 6 hours, producing 4 fixes (locations) per day transmitted to a computer at Eastern Kentucky University via GSM text message. The GPS collars also included VHF transmitters. Bobcats were located approximately every five days over the course of 12 months (May 2010 through May 2011) by using VHF aerial telemetry techniques (Whitehouse and Steven 1977). I

recorded the geographic coordinates, date, time, and cover type for each bobcat location obtained aurally. Both VHF and GPS data sets were randomly sub-sampled, to the mean number of VHF locations, in order to ensure congruent temporal sampling between individual bobcats (J. J. Cox, 2011, Adjunct Assistant Professor, University of Kentucky, personal communication).

ACCURACY TESTING AND DAYTIME BIAS

I conducted a simple accuracy test of aerial telemetry error in March 2011, on the PVB WMA study area. I placed three VHF-only collars and one GPS-with-VHF collar on a ridge line close to a stream restoration site. The collars were attached to trees with various levels of canopy cover, at approximately 33 cm from the ground with the antenna at the 0° position (D'Eon and Delparte 2005). I recorded the GPS location of each collar by using a handheld GPS unit, and called these the “true” locations. I considered this a best case testing scenario because the ridgeline placement would reduce the amount of bounce experienced while tracking from an aircraft (White and Garrott 1990).

I aurally tracked the stationary collars on one occasion from an airplane at 762 m (2500 ft) above ground level, which was the altitude used for this study. True and aurally located (observed) collar locations were uploaded into ArcGIS 9.3.1 (ESRI, Redlands, CA), and I measured the straight-line distance (meters) from each observed location to the true location (White and Garrott 1990). I created a buffer with a radius of half the mean straight-line distance around each observed location and called this my error radius (White and Garrott 1990); this buffer was then placed around the aurally determined locations for each bobcat to account for telemetry error. However, because

testing was not blind the telemetry error should be considered an underestimate (White and Garrott 1990).

In March 2010, I attached four GPS collars to a tree in the PVB WMA to determine GPS accuracy. The collars were attached approximately 1.5 m above the ground, with the antenna at the 0° position; I recorded the true location of each collar with a handheld GPS unit (D'Eon and Delparte 2005). I programmed the collars to record locations at 30 minute intervals; and left the units affixed to the tree for a full 24 hour period. Both the true and observed locations were then uploaded into ArcGIS 9.3.1 and I measured the straight-line distance from each collar's observed location to the true location. A buffer was created by using the same method as described for determining error in the VHF aerial telemetry system.

To test the effects of daytime sampling on home range size estimates versus home range estimates that utilized locations from the entire diel period, I randomly selected 40 mid-day (1200 hr) locations from the GPS collared male bobcat and plotted them in an ArcGIS map (J. J. Cox, 2011, personal communication). I then used BIOTAS software (Ecological Software Solutions, LLC, Hegymagas, Hungary) to get the 95% MCP and AK home range estimates (White and Garrott 1990). These data were compared to calculated annual 95% home range estimates based on 40 locations randomly selected from throughout the diel period, and a percentage difference calculated to determine the effects of daytime sampling bias.

HOME RANGE AND CORE AREA SIZE

Both minimum convex polygon (MCP) and adaptive kernel (AK) bobcat home ranges (95%) and core use areas (50% and 25%) (Seaman and Powell 1996, Seaman et al. 1999)

were calculated from GPS and VHF data by using BIOTAS Software and Hawth's Tools (Spatial Ecology, LLC, <http://www.spatial ecology.com>) for ArcGIS 9.3.1 (Benson et al. 2006, Diefenbach et al. 2006, Riley 2006, Tucker et al. 2008). Previous studies found that bobcat's contract and expand their home ranges seasonally, which can affect the annual estimate (Whitaker 1988). To circumvent this potential bias in comparing home range estimates, I randomly sub-sampled the male's GPS data set to the mean number of female VHF locations used for generating space use estimates (J. J. Cox, 2011, personal communication). Borger et al. (2006) found that both fixed and adaptive kernel methods were comparatively robust and efficient at estimating home ranges, so I chose to use the adaptive kernel method. I defined core areas as those areas in which 50% of all locations occurred. For the MCP method, 50% of locations furthest from the arithmetic mean center point were eliminated before calculating the MCP (Seaman and Powell 1996, Seaman et al. 1999); for the AK core it was the area within the 50% contour. I defined the "central core" for the MCP method as the area containing the central-most 25% of all locations (Millspaugh and Marzluff 2001, Whitaker 1988). For the AK method, the central core was the area within the 25% contour (Millspaugh and Marzluff 2001). Kernel bandwidth was selected by using the Least Squares Cross-Validation method (Hemson et al. 2005, Horne and Garton 2006 a, b). Sample size limitations severely restricted the use of statistical analyses due to test assumption violations; in all cases only the standard error (SE) for the estimates was generated (D. Mundfrom, 2011, Chair and Professor, Department of Mathematics and Statistics, Eastern Kentucky University, personal communication).

OVERLAP ANALYSIS

Percent Overlap

Maintenance of areas of exclusivity may indicate intraspecific competition and an increased population density when there is a high level of home range overlap (Kernohan et al. 2001). I used a type of static interaction analysis, which measures animal interactions throughout a specific time interval and does not take into account the temporal nature of spatial relationships, to measure space use sharing by calculating the percent area overlap between home ranges for pairs of bobcats (Kernohan et al. 2001). I used the Polygon-In-Polygon tool (Hawth's Tools, Spatial Ecology, LLC) to obtain the area measurement of the two individuals' home ranges that intersected. To calculate the percent overlap of the two home ranges, I divided the area of the intersection by the sum of the areas of the two home ranges minus the area of the intersection, and the product multiplied by 100; i.e., $\{(\text{Intersection Area } 1,2) / [(\text{HR1} + \text{HR2}) - \text{Intersection Area } 1,2]\} \times 100$. The percent overlap was calculated at all spatial scales (95, 50, and 25%) for both space use estimators (MCP and AK).

Co-Occurring Locations

In order to determine if two individuals were using the same space, I calculated the percent of co-occurring locations. If two bobcats whose home ranges overlap, but whose locations do not may indicate spatial avoidance which has been identified as an indicator of intraspecific competition (Kernohan et al. 2001, Brashares et al. 2010). For each pair of bobcats, I looked at the percentage of individual points (telemetry locations) in each overlap area to determine if overlapping areas represented "spatially co-occurring

locations." I used the Point-Count-In-Polygon tool (Hawth's Tools, Spatial Ecology, LLC) to obtain the number of points for each individual within the polygon created by the intersection of the two individuals' home ranges (or core areas). Any points that were intersected by the border of the polygon were counted as inside the polygon. To calculate the percent of co-occurring locations, I divided the sum of points that fell within the overlap polygon for two individuals, by the sum of the total number of points within a particular spatial scale for both individuals and multiplied by 100. Additionally, I calculated the mean M-M and F-M percent co-occurring locations. The percent co-occurring location calculation was performed at all 3 spatial scales (90, 50, and 25%) for both space use estimators (MCP and AK).

HABITAT SELECTION ANALYSIS

I followed the "Design III" type study design, which states that all individuals are identifiable and the habitat/resource units that are being studied are recorded as used or unused for either all or part of the sample of individuals (Erickson et al. 2001, Alldredge and Griswold 2006). I assessed habitat selection by bobcats following Neu et al.'s (1974) use-availability habitat selection method. Habitats were broadly classified into three categories that best represented the dominant land cover types in the study area: 1) forested, 2) open (no tree canopy cover, but including areas with shrub cover), and 3) active mine (Neu et al. 1974, Erickson et al. 2001, Alldredge and Griswold 2006). To determine study area boundaries, I plotted trap site locations in ArcGIS and calculated half the mean maximum distance traveled by all bobcats (greatest distance between two locations of all locations for an individual bobcat) to create a buffer around each trap site (White and Garrott 1990, Kelly et al. 2008). The buffers were then dissolved (a tool in

ArcGIS that merges all of the selected polyforms into one shapefile) to generate the study area boundary, which was overlaid onto a 2010 orthographic image map of the area (<http://datagateway.nrcs.usda.gov>, accessed 2010). The study area encompassed all but one of the bobcat home ranges; in order to be inclusive, I dissolved the study area boundary with the home range boundary and called this inclusive boundary the study area. I clipped the ortho file to the study area boundary and performed an unsupervised reclassification of the clipped raster, starting with 100 classes and reclassifying down to the three aforementioned habitat classes (Erickson et al. 2001).

The proportional area of each habitat class was calculated in GIS from the raster count values and the raster cell size in the following equation: (cell size x) x (cell size y) x (class count value) (Chamberlain et al. 2003; <http://forums.arcgis.com>, accessed May 2011). The cell size of the 2010 ortho file was 1 m X 1 m, and areas were expressed in square kilometers. The observed locations for each bobcat were then overlaid onto the reclassified map image and a point count of locations within each habitat class was taken by joining the attribute tables of the ortho-raster values to the locations attribute table.

To determine if bobcats were selecting for or avoiding a particular habitat, I performed a chi-square goodness-of-fit test [as outlined in Neu et al. (1974)] by using MS Excel (Microsoft Corporation, Redmond, WA), with a null hypothesis that bobcats would use all three habitat classes in proportion to their availability and an alternative hypothesis that the habitats would be used disproportionately from available (Byers et al. 1984, White and Garrott 1990). I calculated the expected number of locations for each class based on the estimated availability of each class (as described previously) and the total number of bobcat locations in the sample (Erickson et al. 2001, Whitlock and

Schulter 2009). I calculated adjusted 95% Bonferroni confidence intervals for the proportion of habitat used only if the null hypothesis was rejected (Byers et al. 1984, Alldredge and Griswold 2006). If the upper bound of the Bonferroni confidence interval for habitat use was lower than the proportion available for that habitat class, then the habitat type was considered as “avoided;” and if the lower bound of the confidence interval for use exceeded the proportion of habitat available, the habitat type was considered as “selected.” If the proportion of available habitat fell within the confidence interval bounds, the habitat selection was considered "proportionate." If the null hypothesis was not rejected, all habitat classes were assumed to be used proportionate to their availability. To exclude bias due to uneven sampling of bobcats or times of day, I randomly removed the appropriate number of points from individual bobcat locations and time-of-day categories, until all bobcats had the same number of locations in their sample (J. J. Cox, 2011, personal communication). I pooled telemetry locations for all bobcats in order to meet the assumptions of the chi-square analysis; and I assumed that all locations were independent (Whitlock and Schuller 2009). Pooling allowed me to change the sample unit from the bobcat to the location (Erickson et al. 2001). To determine if pooling had an effect on the likelihood of Type I or II errors, I applied the preceding procedure to each bobcat individually to determine if overall patterns of habitat selection varied among individuals (White and Garrott 1990).

I performed the habitat selection analysis at the study area and the 95% home range, as well as the 50% and 25% core area scales (as defined by the MCP estimates only, due to issues in calculating the habitat availability using the adaptive kernel contours in ArcGIS), because bobcats may select habitats at small spatial scales and patterns of

resource use may be present that do not occur at larger spatial scales (Erickson et al. 2001). To determine the amount of available habitat for the pooled data set, I totaled the area of each habitat class from all individual bobcats' home range or core areas. Analyses for the 95%, 50%, and 25% MCP scales of availability were the same as those conducted when the study area was used to estimate habitat availability.

The Chi-Square Goodness-of-Fit Test and Bonferroni Correction

For the chi-square analysis, I defined habitat selection as the disproportionate use of one or more habitats compared to expected use based on the availability of each habitat type (White and Garrott 1990). Habitats that are considered avoided are those that are used less than would be expected based on availability (White and Garrott 1990). The chi-square tests the goodness of fit of the observed locations to the expected distribution of locations where the individual uses each habitat proportionately to its availability, or, thus, at random (White and Garrott 1990). White and Garrott (1990) recommend not pooling data unless faced with a situation of having few observations on many individuals. In my study, I have been presented with the issue of having many observations on only a few individuals. However, the chi-square test requires a minimum sample size of 5 for each category (Whitlock and Schulter 2009). If each bobcat in this study was tested for habitat preferences, many expected values would have been below 5 locations within a particular habitat category; therefore I pooled the data.

Although Cherry (1996) showed that the adjusted confidence interval calculation outlined by Neu et al. (1974) was inferior to calculations presented by Bailey (1980), I decided to use Neu et al.'s (1974) original calculation because I pooled the location data from a small sample of individuals. While this may decrease the power of the analysis,

the calculations presented by Bailey (1980) were intended for large sample sizes and may not be applicable for the small sample sizes of this study or for pooled data.

CHAPTER 4

RESULTS

TRAPPING AND MARKING

I trapped bobcats from February through April 2010, for a total of 448 trap nights (TN) and a capture rate of 1.8/100 TN. I captured and collared a total of seven bobcats (5 with VHF collars and 2 with GPS collars; 2 M, 5 F); 1 GPS collar failed and 1 VHF collar either slipped off or was removed from a female bobcat, both after approximately 1 month of deployment. Although both GPS collars were also equipped with VHF transmitters, the VHF unit also failed simultaneously with the GPS unit on the failed collar; only two GPS collars were deployed because other bobcats were not large enough to carry the heavier GPS units. These two individuals were not included in any of the space use or habitat use-availability analyses.

Adult males ($n = 2$) had an average weight of 7.14 ± 1.12 (SD) kg, and adult females ($n = 4$) averaged at 5.90 ± 0.56 kg (Table 1). Three of the 4 females were pregnant at the time of capture. All bobcats collared were characterized as adults.

SPACE USE

Accuracy Testing and Daytime Bias

Aerial telemetry locations had an estimated error radius of approximately 200 m; while GPS test locations resulted in an error radius of 8 m. The aerial error radius estimate was considered an underestimate because placement of the test collars was not

blind and relocation on multiple dates was not possible due to weather and road access interference.

Table 1. Standard measurements of bobcats trapped in Breathitt and Perry counties, Kentucky, in February - March 2010.^a

ID	Weight		TL (cm)	SH (cm)	ELR (cm)	ELL (cm)	RFFW (cm)	RFFL (cm)	RHFL (cm)	TAIL (cm)	NC (cm)	CG (cm)	ZA (mm)
	(kg)												
F1	5.22	76.5	37	6.5	6.4	4.4	3.9	4.2	4.2	12	19	31.4	58.2
F2	6.58	69.5	42	7	8	4	4.5	16.5	12.5	21.5	35	35	78.1
M1	7.94	73.5	45	9.5	9	5.25	4.5	18	15	23.5	38.5	38.5	87
F3	5.90	80	34.5	9.5	8.5	4.2	4.5	17.2	14.3	19.7	35.5	35.5	82.2
F4	5.90	72	40	8	8.5	4.8	4.5	16.5	15.5	21.5	N/A	N/A	83.3
M2	6.35	76	42	8.5	7.5	4.8	3.5	16	11.5	20	33	33	91

^aAbbreviations: TL, Total Length; SH, Shoulder Height; ELR, Ear Length Right; ELL, Ear Length Left; RFFW, Right Front Foot Width; RFFL, Right Front Foot Length; TAIL, Tail Length; NC, Neck Circumference; CG, Chest Girth; ZA, Zygomatic Arch.

Daytime samples resulted in a 95% MCP home range size that was similar in size to that produced from locations sampled from all times of day (Table 2). The daytime 95% AK home range size, however, was reduced by 88% when compared to the annual 95% AK home range produced from locations from the entire diel period (Table 2).

Home Range and Core Area Size

Bobcats were radio-tracked from 1 May 2010 through 31 May 2011 during 66 flights, producing a maximum of 65 points (minimum of 58 points) for the year and averaging 63 locations per bobcat. Due to collar malfunctions, the GPS collared male had a much smaller VHF data set compared to the females, in both number of locations and time span sampled. Annual 95% MCP home range (HR) size, for both male and female bobcats, varied from 4.9 to 25.3 km² (n = 5, SE = 3.9, mean = 14.9 km²; Table 3), with a mean female annual HR size of 17.4±3.9 km² (n = 4). The 95% AK home range estimate varied among all bobcats (1.3 to 47.3 km²), with a mean female HR of 27.4±7.5 km² (n = 4). Annual MCP core area size varied from 1.3 to 4.5 km² (n = 5, SE = 3.0, Figure 1); and from 0.01 to 6.3 km² (n = 5, SE = 3.4) for AK estimates. Mean female core areas were 3.4±0.6 (MCP) and 4.2±1.1 (AK) km² (n = 4). MCP annual central core area (25%) size ranged from 0.5 to 1.2 km² (n = 5), with a mean female central core area of 0.9±0.2 km² (n = 4). AK central core areas ranged from 0.003 to 2.2 km² (n = 5), with a mean female central core area of 1.6±0.5 km² (n = 4).

Table 2. Estimated 95% minimum convex polygon (MCP) and 95% adaptive kernel (AK) home ranges (HR) when based on 40 randomly sampled daytime (1200 hr) locations versus those based on 40 locations randomly selected from all time of day.

Location Type	MCP HR Size (km ²)	AK HR Size (km ²)
Daytime	6.03	0.93
Annual	5.56	7.51
Difference	8.5%	87.6%

Table 3. Home range, core area, and central core area estimates, based on two methods [minimum convex polygon (MCP) and adaptive kernel (AK)] for 4 females (F1-F4) and one male (M1) bobcat radio-tracked in eastern Kentucky, May 2010 - May 2011.

Home Range and Core Area Size Estimates (km ²)							
	F1	F2	F3	F4	M1	Mean	SE
95% MCP	9.23	22.41	25.29	12.48	4.92	14.87	3.89
95% AK	19.11	26.47	47.27	16.8	1.27	22.18	7.50
50% MCP	2.65	3.8	4.51	2.6	1.26	2.96	0.56
50% AK	5.19	3.22	6.29	2.22	0.01	3.39	1.11
25% MCP	0.79	1.14	1.23	0.47	0.63	0.85	0.15
25% AK	2.54	0.76	2.17	0.93	0.003	1.28	0.47

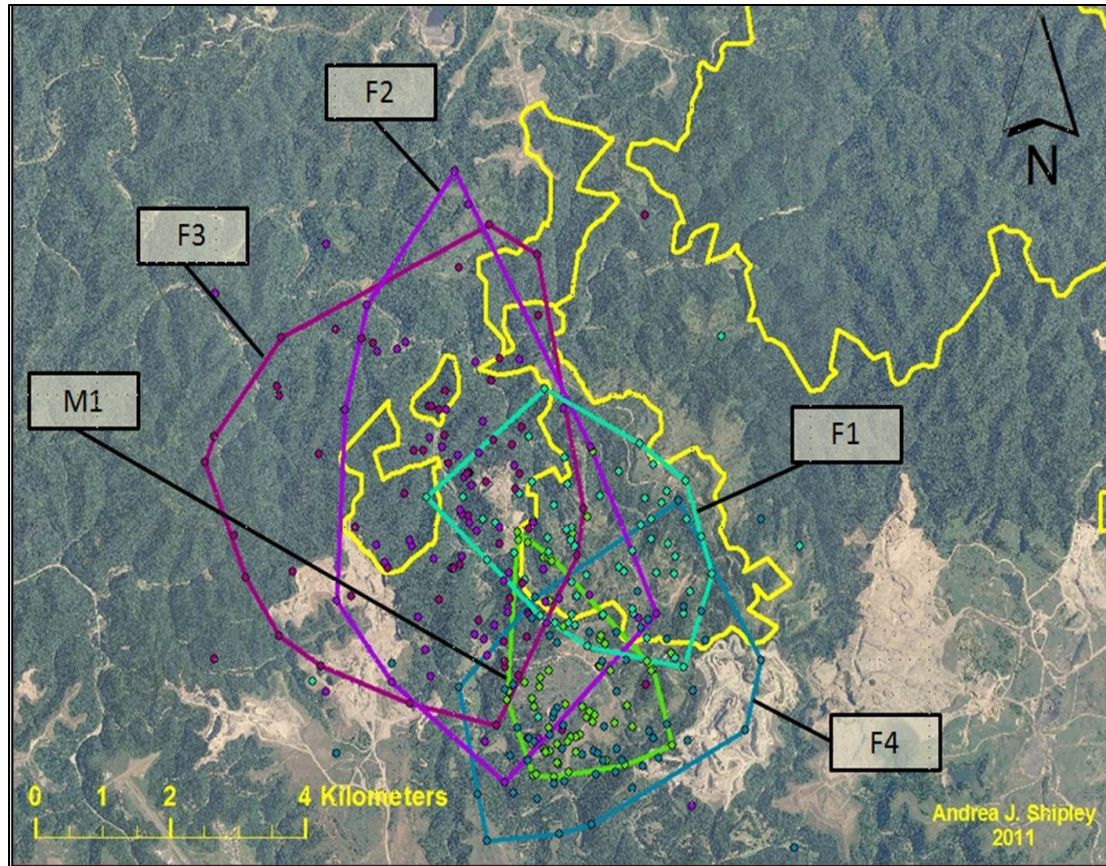


Figure 1. Home ranges [95% minimum convex polygon (MCP)] of bobcats in Breathitt and Perry counties, KY, May 2010 – May 2011.

F1 - F4 designate female bobcats; M1 is a male. Matching colors are used for corresponding polygons and telemetry locations: F1 is represented by light blue, F2 is represented by purple, F3 is represented by dark pink, F4 is represented by dark blue, and M1 is represented by green. The Paul Van Booven Wildlife Management Area and Robinson Forest boundaries are represented in yellow.

Overlap Analysis

There was considerable intrasexual and intersexual annual area overlap for bobcats in this study at the home range scale; but 50% and 25% core areas remained mostly exclusive. For both estimation methods, locations from only 1 individual were present in the overlapping areas in some instances. Female-female (F-F) MCP annual home range overlap ranged from 2.8 to 61.2% ($n = 6$ adjacent pairs, mean = 29.1%, SE = 8.7, Table 4); while female-male (F-M) HR overlap was from 3.1 to 34.4% ($n = 4$ adjacent pairs, mean = 17.1%, SE = 7.0; see Figure 2). When generated by adaptive kernel analysis, the F-F home range overlap was higher than when estimated by MCP, from 12.4 to 72.2% ($n = 6$ adjacent pairs, mean = 38.3%, SE = 9.4, Table 4), but F-M overlap was lower, from 2.4 to 6.7% ($n = 4$ adjacent pairs, mean = 3.9%, SE = 1.0). MCP 50% core area overlap for F-F ranged from 0 to 62.9% ($n = 6$ adjacent pairs, mean = 10.5%, SE = 10.5), and F-M 0 to 48.5% ($n = 4$ adjacent pairs, mean = 12.1, SE = 12.1). AK core area overlap for F-F was 0 to 44.3% ($n = 6$ adjacent pairs, mean = 9.3%, SE = 7.1), and F-M 0 to 0.5% ($n = 4$ adjacent pairs, mean = 0.11, SE = 0.1, Figure 3). F-F 25% central core area overlap by MCP analysis varied from 0 to 28.1% ($n = 6$ adjacent pairs, mean = 4.7%, SE = 4.7%), and F-M 0 to 12.2% ($n = 4$ adjacent pairs, mean = 3.1%, SE = 3.1). For AK analysis F-F overlap of central core areas was 0 to 18.6% ($n = 6$ adjacent pairs, mean = 3.1%, SE = 3.1) and F-M 0 to 0.2% ($n = 4$ adjacent pairs, mean = 0.05, SE = 0.1).

For each pair of bobcats, I determined the percentage of individual points in each overlap area to determine if overlapping areas represented spatially co-occurring locations. Of those areas that overlapped for two individuals, the annual home ranges had varying percentages of co-occurring locations, while core areas and central core areas

Table 4. Percentage of minimum convex polygon (MCP) and adaptive kernel (AK) home range overlap between bobcats in Breathitt and Perry counties, Kentucky, May 2010 – May 2011, for both home range (95%) and core area (50% and 25%) estimates. "F" represents female bobcats, and "M" a male.

ID	Overlap Estimates (%)												SE
	F1-F2	F1-F3	F1-F4	F1-M1	F2-F3	F2-F4	F2-M1	F3-F4	F3-M1	F4-M1	Mean		
95% MCP	41.19	36.50	18.31	8.85	61.15	14.51	21.95	2.80	3.07	34.36	24.27	8.41	
95% AK	72.19	40.43	28.57	3.29	57.56	18.74	3.01	12.38	2.38	6.80	24.54	11.09	
50% MCP	0	0	0	0	62.94	0	0	0	0	48.46	11.14	10.61	
50% AK	3.19	8.20	0	0	44.31	0	0	0	0	0.45	5.62	6.19	
25% MCP	0	0	0	0	28.11	0	0	0	0	12.24	4.04	4.16	
25% AK	0	0	0	0	18.62	0	0	0	0	0.21	1.88	2.63	

^a Abbreviations: MCP, Minimum Convex Polygon; AK, Adaptive Kernel.

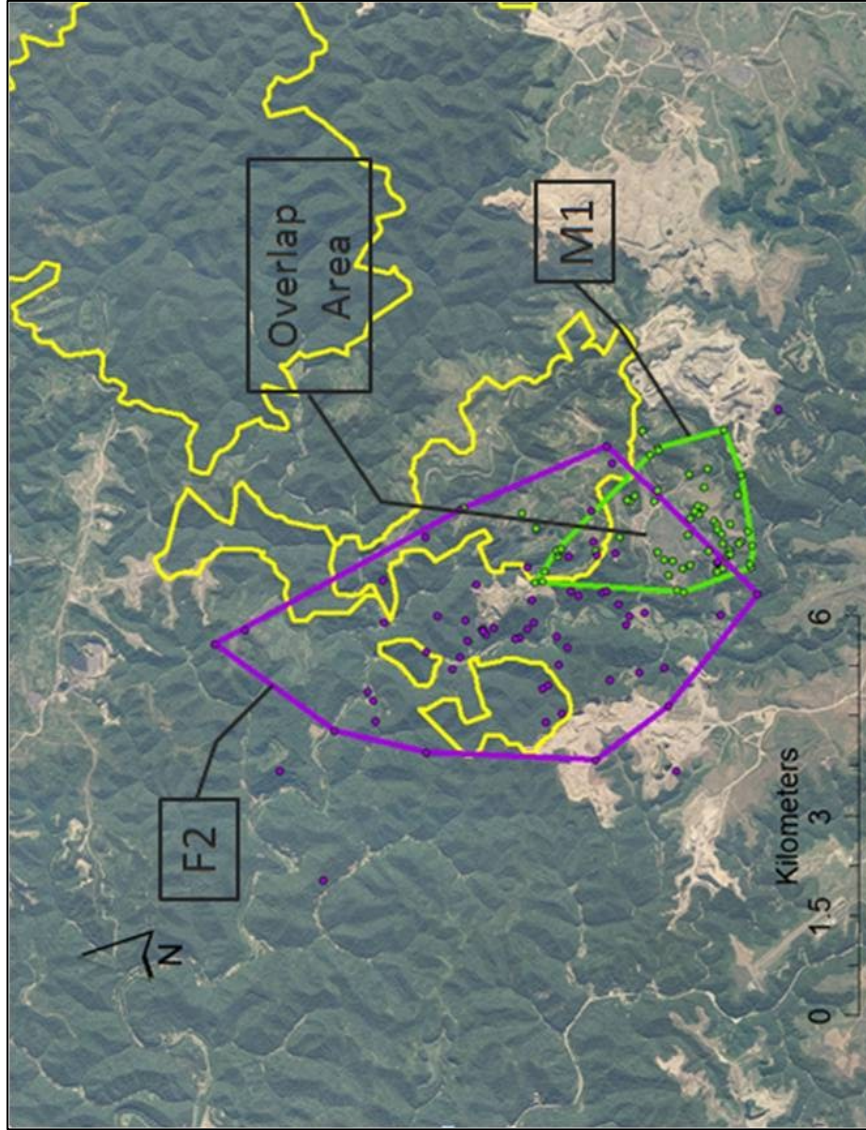


Figure 2. An example of overlapping home ranges for male (M) and female (F) bobcats in eastern Kentucky in 2011. The solid lines are the 95% minimum convex polygon home range boundaries; dots represent all locations of bobcats estimated by radio-telemetry.

remained mostly exclusive; i.e., few or no co-occurring locations (Table 5a-c). For 95% MCP and AK home range overlap areas, percentage of co-occurring locations was 39% and 46%, respectively; mean F-F for 95% MCP was 42% and F-M 34%; while F-F and F-M means for 95% AK were 53% and 36%, respectively. Core area overlap areas had average co-occurring location percentages of 17% and 11%, for MCP and AK, respectively; the mean value for AK was lower because more individuals had overlapping areas. Average F-F co-occurring location percentages were 12% for MCP and 15% for AK; F-M percentages 24% (MCP) and 5% (AK). Central (25%) core areas were mainly exclusive, with a mean overall MCP co-occurring location percentage of 5%, and 6% for AK. F-F and F-M location percentages were 4% and 7% for MCP, respectively; and 6% and 6% for AK estimates, respectively.

Table 5a. Percentage of co-occurring locations within 95% minimum convex polygon (MCP) and adaptive kernel (AK) home ranges for male (M) and female (F) bobcats in Breathitt and Perry counties, KY, May 2010 – May 2011.

ID (A-B)	% of Total Co-Occurring Locations for MCP	% of Total Co- Occurring Locations for AK
F1-F2	58	65
F1-F3	36	55
F1-F4	30	39
F2-F3	90	86
F2-F4	26	38
F3-F4	10	32
Mean F-F	42	53
F1-M1	18	22
F2-M1	30	30
F3-M1	13	40
F4-M1	74	52
Mean F-M	34	36
Overall Mean % Co-Occurring Locations	39	46

Table 5b. Percentage of co-occurring locations within 50% minimum convex polygon (MCP) and adaptive kernel (AK) core areas for male (M) and female (F) bobcats in Breathitt and Perry counties, KY, May 2010 – May 2011.

ID (A-B)	% of Total Co-Occurring Locations for MCP	% of Total Co- Occurring Locations for AK
F1-F2	0	5
F1-F3	0	14
F1-F4	0	0
F2-F3	74	70
F2-F4	0	0
F3-F4	0	0
Mean F-F	12	15
F1-M1	0	0
F2-M1	0	0
F3-M1	0	0
F4-M1	97	20
Mean F-M	24	5
Overall Mean % Co-Occurring Locations	17	11

Table 5c. Percentage of co-occurring locations within 25% minimum convex polygon (MCP) and adaptive kernel (AK) core areas for male (M) and female (F) bobcats in Breathitt and Perry counties, KY, May 2010 – May 2011.

ID (A-B)	% of Total Co-Occurring Locations for MCP	% of Total Co- Occurring Locations for AK
F1-F2	0	0
F1-F3	0	0
F1-F4	0	0
F2-F3	27	36
F2-F4	0	0
F3-F4	0	0
Mean F-F	4	6
F1-M1	0	0
F2-M1	0	0
F3-M1	0	0
F4-M1	26	22
Mean F-M	7	6
Overall Mean % Co-Occurring Locations	5	6

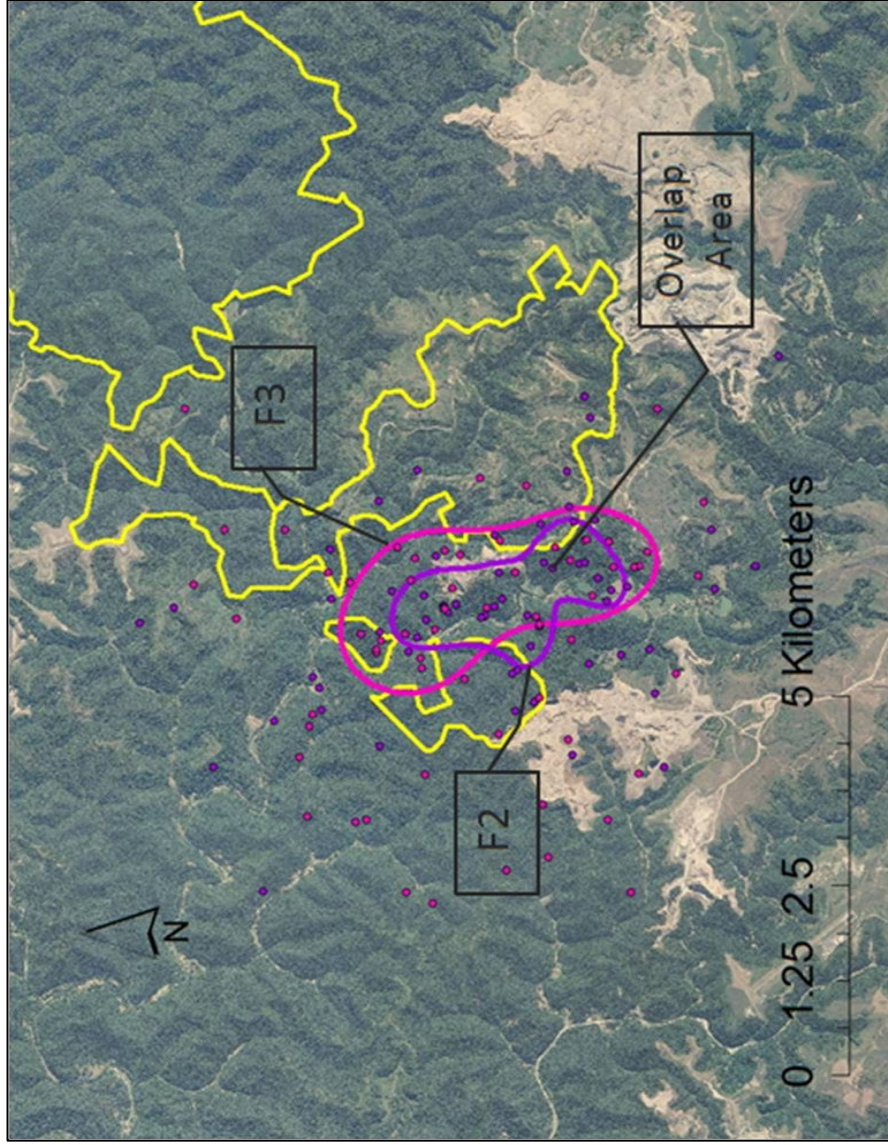


Figure 3. An example of overlapping core areas for female (F) bobcats in eastern Kentucky in 2011. The solid lines are the 50% adaptive kernel home range boundaries; dots represent all locations of bobcats estimated by radio-telemetry; green dots represent a males' locations.

HABITAT SELECTION ANALYSIS

Bobcats (all bobcats pooled) used the forested habitat class in proportion to its availability at the study area spatial scale ($P < 0.001$, Table 6, Figure 4), but they may be selecting for the open habitat class and avoiding the active habitat class. More specialized statistical analyses were not possible without violating minimum sample size requirements (White and Garrott 1990).

Female bobcats 1, 2, and 3 used all habitats in proportion to their availability within the study area (Table 7a, b, c). Female bobcat F4 used open habitat significantly more (Table 7d), but the Bonferroni confidence intervals only than expected, but active and forested habitats were used in proportion to their availability. The male bobcat avoided the forested habitat class, selected for open habitat, and did not show any preference for the active habitat. Female 4 and the male bobcat's selection of the open habitat class likely led to the result seen in the pooled data analysis; given that the other 3 bobcats showed no disproportionate use of available habitat types. No individual bobcat avoided the active habitat class; however, when looking at the number of observed locations, versus expected locations for this habitat class, there is a trend for the observed locations to be less than expected.

Female bobcats 1, 2, and 4, and the male all had similar amounts of each habitat present within their 95% MCP home range. Female bobcat #3 had a greater percentage of forested and lower percentage of open habitat present (Table 8). The pooled data set chi-square analysis indicated that there was a disproportionate use of the three habitat classes. Only the open habitat fell outside the lower confidence interval, indicating a selection for the open habitat class, for the pooled bobcat data set (Table A-3a).

Table 6. Chi-square goodness-of-fit values, with Bonferroni Z calculated confidence intervals, for habitat selection at the study area spatial scale of 3 habitat classes, by all bobcat (n = 5), using the Neu et al. (1974) method.

Habitat Class	Spatial Scale	Area (km ²)	Proportion of available habitat	Observed Locations	Expected Locations	Proportion of observed locations in each area	X ²	df	P	Confidence interval on proportion of habitat use (<i>p</i>)	Avoided/ Selected/ Proportionate/ Ho/Ha
Active	Study area	17	0.1771	41	55	0.131	4.2			0.0852 < <i>p</i> < 0.1768	Avoided
Forested	Study area	70	0.729	218	228	0.703	0.5			0.634 < <i>p</i> < 0.759	Proportionate
Open	Study area	9	0.094	54	29	0.173	23.7			0.121 < <i>p</i> < 0.224	Selected
Total	Study area	96	1.000	313	313	1.0	28.3	2	< 0.001		Ha

Abbreviations: X², Chi-square value; df, degrees of freedom; P, significance level; *, confidence intervals not calculated because the null was not rejected; *p*, represents the theoretical occurrence of locations compared to the proportion of available habitat, in order to determine if the hypothesis of proportional use is accepted or rejected.

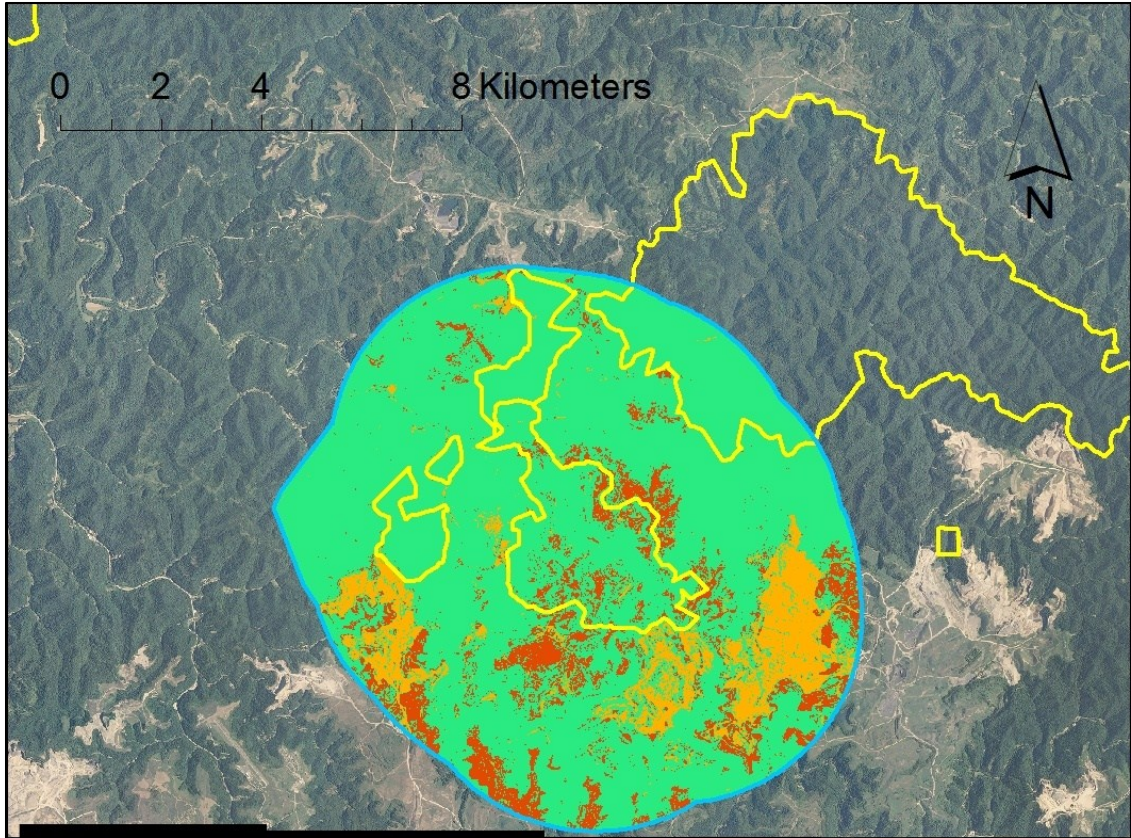


Figure 4. Habitat classes present on the bobcat study area in southeastern Kentucky.

The habitat classes are defined by color: Active habitat (active mining) in light orange, Forested habitat in light green, and Open habitat in orange-red; the border of the Paul Van Booven Wildlife Management Area and Robinson Forest is in yellow, and the border of the project study area is in blue.

Table 7a. Chi-square goodness-of-fit values, with Bonferroni Z calculated confidence intervals, for habitat selection at the study area spatial scale of 3 habitat classes, for female bobcat ‘F1’, using the Neu et al. (1974) method.

Habitat Class	Spatial Scale	Area (km ²)	Proportion of available habitat	Observed Locations	Expected Locations	Proportion of observed locations in each area	X ²	df	P	Confidence interval on proportion of habitat use (<i>p</i>)	Avoided/ Selected/ Proportionate/ Ho/Ha
Active	Study area	17	0.177	5	12	0.077	3.7			*	Proportionate
Forested	Study area	70	0.729	51	47	0.785	0.3			*	Proportionate
Open	Study area	9	0.094	9	6	0.138	1.4			*	Proportionate
Total		96	1.000	65	65	1.000	5.3	2	P > 0.75		Ho

Abbreviations: X², Chi-square value; df, degrees of freedom; P, significance level; *, confidence intervals not calculated because the null was not rejected; *p*, represents the theoretical occurrence of locations compared to the proportion of available habitat, in order to determine if the hypothesis of proportional use is accepted or rejected; Ho, null hypothesis; Ha, alternative hypothesis.

Table 7b. Chi-square goodness-of-fit values, with Bonferroni Z calculated confidence intervals, for habitat selection at the study area spatial scale of 3 habitat classes, for female bobcat ‘F2’, using the Neu et al. (1974) method.

Habitat Class	Spatial Scale	Area (km ²)	Proportion of available habitat	Observed Locations	Expected Locations	Proportion of observed locations in each area	X ²	df	P	Confidence interval on proportion of habitat use (<i>p</i>)	Avoided/ Selected/ Proportionate/ Ho/Ha
Active	Study area	17	0.177	11	11	0.180	0.0			*	Proportionate
Forested	Study area	70	0.729	43	44	0.705	0.0			*	Proportionate
Open	Study area	9	0.094	7	6	0.115	0.3			*	Proportionate
Total		96	1.000	61	61	1.000	0.3	2	P > 0.99		Ho

Abbreviations: X², Chi-square value; df, degrees of freedom; P, significance level; *, confidence intervals not calculated because the null was not rejected; *p*, represents the theoretical occurrence of locations compared to the proportion of available habitat, in order to determine if the hypothesis of proportional use is accepted or rejected; Ho, null hypothesis; Ha, alternative hypothesis.

Table 7c. Chi-square goodness-of-fit values, with Bonferroni Z calculated confidence intervals, for habitat selection at the study area spatial scale of 3 habitat classes, for female bobcat ‘F3’, using the Neu et al. (1974) method.

Habitat Class	Spatial Scale	Area (km ²)	Proportion of available habitat	Observed Locations	Expected Locations	Proportion of observed locations in each area	X ²	df	P	Confidence interval on proportion of habitat use (<i>p</i>)	Avoided/ Selected/ Proportionate/ Ho/Ha
Active	Study area	17	0.177	7	11	0.115	1.3			*	Proportionate
Forested	Study area	70	0.729	53	44	0.869	1.6			*	Proportionate
Open	Study area	9	0.094	1	6	0.016	3.9			*	Proportionate
Total		96	1.000	61	61	1.000	6.9	2	P > 0.75		Ho

Abbreviations: X², Chi-square value; df, degrees of freedom; P, significance level; *, confidence intervals not

calculated because the null was not rejected; *p*, represents the theoretical occurrence of locations compared to the

proportion of available habitat, in order to determine if the hypothesis of proportional use is accepted or rejected; Ho,

null hypothesis; Ha, alternative hypothesis.

Table 7d. Chi-square goodness-of-fit values, with Bonferroni Z calculated confidence intervals, for habitat selection at the study area spatial scale of 3 habitat classes, for female bobcat “F4”, using the Neu et al. (1974) method.

Habitat Class	Spatial Scale	Area (km ²)	Proportion of available habitat	Observed Locations	Expected Locations	Proportion of observed locations in each area	X ²	df	P	Confidence interval on proportion of habitat use (<i>p</i>)	Avoided/ Selected/ Proportionate/ Ho/Ha
Active	Study area	17	0.177	8	11	0.131	0.7			0.027 < <i>p</i> < 0.235	Proportionate
Forested	Study area	70	0.729	39	44	0.639	0.7			0.492 < <i>p</i> < 0.787	Proportionate
Open	Study area	9	0.094	14	6	0.230	11.9			0.100 < <i>p</i> < 0.395	Selected
Total		96	1.000	61	61	1.000	13.3	2	P < 0.001		Ha

Abbreviations: X², Chi-square value; df, degrees of freedom; P, significance level; *, confidence intervals not calculated because the null was not rejected; *p*, represents the theoretical occurrence of locations compared to the proportion of available habitat, in order to determine if the hypothesis of proportional use is accepted or rejected; Ho, null hypothesis; Ha, alternative hypothesis.

Table 7e. Chi-square goodness-of-fit values, with Bonferroni Z calculated confidence intervals, for habitat selection at the study area spatial scale of 3 habitat classes, for male bobcat ‘M1’, using the Neu et al. (1974) method.

Habitat Class	Spatial Scale	Area (km ²)	Proportion of available habitat	Observed Locations	Expected Locations	Proportion of observed locations in each area	X ²	df	P	Confidence interval on proportion of habitat use (<i>p</i>)	Avoided/ Selected/ Proportionate/ Ho/Ha
Active	Study area	17	0.177	10	12	0.154	0.2			0.046 < <i>p</i> < 0.261	Proportionate
Forested	Study area	70	0.729	32	47	0.492	5.0			0.343 < <i>p</i> < 0.641	Avoided
Open	Study area	9	0.094	23	6	0.354	46.7			0.212 < <i>p</i> < 0.496	Selected
Total		96	1.000	65	65	1.000	51.9	2	P < 0.001		Ha

Abbreviations: X², Chi-square value; df, degrees of freedom; P, significance level; *, confidence intervals not calculated because the null was not rejected; *p*, represents the theoretical occurrence of locations compared to the proportion of available habitat, in order to determine if the hypothesis of proportional use is accepted or rejected; Ho, null hypothesis; Ha, alternative hypothesis.

Table 8. Percent of active, forested, and open habitat present within the 95% minimum convex polygon home range of four female (F1, F2, F3, F4) and one male (M1) bobcat in southeastern Kentucky, May 2010 - May 2011.

ID	Habitat Types		
	Active	Forested	Open
F1	15%	74%	11%
F2	16%	76%	8%
F3	16%	80%	4%
F4	20%	63%	17%
M1	16%	59%	25%

Individually, the male (M1), and female bobcats F2 and F3 used each of the three habitat classes in proportion to their availability; even though the overall chi-square test indicated disproportionate use of the 3 habitats combined (Tables A-3b-f). This result is likely due to a sample size issue, similar to that of the Study Area spatial scale analysis, where pooling the data allows for a large enough sample size to show habitat use patterns.

At the 50% MCP core area spatial scale all bobcats had similar percentages of available active habitat, but for the forested habitat class F2 and F3 had a larger percentage than F1, F4, and M1 (Table 9). Conversely, for the open habitat class, F2 and F3 had a much lower percentage available than the other three individuals. The chi-square analysis on the pooled and the individual data sets did not indicate disproportionate use of the available habitat classes (Tables A-4a-f).

At the 25% central core area spatial scale the percent available habitat followed the same trend for the individual bobcats as seen at the 50% MCP spatial scale (Table 10). The chi-square analysis on the pooled data set did not indicate any disproportionate use of the available habitat classes (Tables A-5a-f). The chi-square analysis on the individual cats also did not indicate disproportionate use of the available habitat classes, except for female 2 (F2); F2 avoided the active habitat class and selected the forested habitat class, however this result was likely due to the lack of any locations within the open habitat class (Table A-5c).

Table 9: Percent of active, forested, and open habitat present within the 50% minimum convex polygon home range of four female (F1, F2, F3, F4) and one male (M1) bobcat in southeastern Kentucky, May 2010 - May 2011..

ID	Habitat Type		
	Active	Forested	Open
F1	15.5%	65.5%	19.0%
F2	18.0%	78.5%	3.5%
F3	15.0%	82.0%	3.0%
F4	18.5%	52.0%	29.5%
M1	15.0%	46.0%	39.0%

Table 10: Percent of active, forested, and open habitat present within the 25% minimum convex polygon home range of four female (F1, F2, F3, F4) and one male (M1) bobcat in southeastern Kentucky, May 2010 - May 2011.

ID	Habitat Types		
	Active	Forested	Open
F1	15%	63%	22%
F2	11%	86%	3%
F3	10%	89%	1%
F4	19%	60%	21%
M1	13%	43%	44%

CHAPTER 5

DISCUSSION

SPACE USE BY BOBCATS

Because bobcats are a sexually dimorphic and polygynous species (McCord and Cardoza 1982), males are expected to retain larger home ranges than females to increase the likelihood of reproduction with females during the breeding season (Chamberlain et al. 2003, Cochrane et al. 2006). Numerous past studies have documented this pattern (Fuller et al. 1985, Lovallo and Anderson 1996b, Chamberlain et al. 2003, Cochrane et al. 2006), but in this study females had larger home ranges than the male. Overlap estimates calculated by MCP versus AK analysis differ because kernel volume contours are based on the density distribution of points, creating a more accurate picture as to where actual locations co-occur; whereas MCP overlap may not actually represent where locations occur in space. Perhaps my sample size ($n = 5$) was too small to accurately portray space use by bobcats, or the one male's home range adequately overlapped enough females and contained adequate resources to not necessitate a larger home range size (Feldhamer et al. 2007). Unfortunately, without a greater sample size, particularly of males, it is impossible to make any sex-specific inferences regarding space use by bobcats in the area encompassed by this study.

Effects of Daytime Sampling

AK home ranges generated from only daytime GPS locations resulted in a greatly reduced bobcat home-range size estimate when compared to the home range generated from 24-hr locations. It could be postulated that all home range estimates produced from

daytime VHF aerial locations are likely to be underestimates of actual space use by the bobcats tracked in this study (Kernohan et al. 2001). This underestimation could have a great impact on the interpretation of the overlap and habitat selection results presented herein. However, because of the sample size issue, it is difficult to ascertain what those effects might be.

MCP home ranges for bobcats generated from GPS locations from the entire diel period were similar to daytime-only home range estimates. MCP estimates are generated by creating a polygon from a percentage of the innermost locations, assuming that space use is equal within the polygon (White and Garrott 1990). It is less likely that MCP home range estimates would be affected by daytime bias because the spatial density of locations is not assessed with this estimator as it is with kernel estimates (Kernohan et al. 2001).

Space use changes over time

Whitaker (1988) reported on space use characteristics of bobcats in the vicinity of the PVB WMA by using MCP analysis to calculate home range sizes and areas of overlap. During her study, much of the current-day management area was active mining property or in early-stage reclamation (Whitaker 1988). During this study the PVB area was mostly an area of secondary succession, surrounded by mining property at various stages of the mining process. Bobcat home range, core area, and central core area size estimates within the region encompassed by this study were consistently higher for males than females in the late 1980's (Whitaker 1988); a trend reported elsewhere in the southeastern region of the U.S. (Fuller et al. 1985, Lovallo and Anderson 1996b, Chamberlain et al. 2003, Cochrane et al. 2006). Contrary to the findings of Whitaker

(1988), females in this study had consistently larger home ranges, core areas, and central core areas than the male. The home ranges of female bobcats in this study were much larger (17.4 versus 4.7 km²), and males' home range much smaller (4.9 versus 44.7 km²), than those in Whitaker's (1988) study. My home range overlap data are also contrary to the pattern reported by Whitaker (1988), i.e., her females maintained exclusive home range areas. I also found a much lower female-male home range overlap than Whitaker (1988): 17% in 2011 compared to 100% in 1988. Female-female home range overlap increased, from no overlap in 1988 to 29% in 2011.

In previous studies, female bobcats were found to maintain primarily exclusive home ranges, with little intrasexual overlap (Lawhead 1984, Lovallo and Anderson 1996b, Nielsen and Woolf 2001). Changes in population density have been found to change space use over time, from increased dispersal of juvenile individuals (Kitchings and Story 1984, Kamler et al. 2000, Janecka et al. 2007) to home range contraction (Feldhamer et al. 2007). Female bobcats generally maintain site fidelity, and female juveniles will often disperse to adjacent territories when resources are adequate (Kamler et al. 2000, Janecka et al. 2007). Increased population density in areas with abundant resources could explain the increased levels of home range overlap. Speculatively, if bobcat survival rates have grown since the late 1980's then it is likely that female juveniles are able to share part of their natal home ranges in areas where resources are abundant (Kitchings and Story 1984, Kamler et al. 2000). This also may be why the female-male home range overlap decreased; with an increased number of females in the area, a male bobcat would not need to completely overlap a females' home range in order to secure breeding opportunities (Nielsen and Woolf 2001). However, the superficial (without statistical

assessment) increase in female home range size from the late 1980's to present day suggests that this possibility may not explain the entire story.

When Whitaker (1988) undertook her study there was no legal harvest of bobcats in the state of Kentucky; however, at present there is a season limit of 5 bobcats per trapper. The PVB Management Area is very active during the hunting and trapping seasons. Harvesting may explain the larger home ranges for the females observed in this study, though it does not explain why the male bobcat had such a reduced home range size. Avoidance of human activities could be a more-likely reason females have expanded their home ranges and the male has contracted his. Solely speculating on space use, when I plotted the GPS locations of the male onto the aerial map it appeared that he was avoiding areas most likely used by humans (e.g., roads, active mine areas, and areas with little to no forested habitat), restricting himself to corridors of highly forested tracts (Fig. A-94). Though I was not able to collar the females with GPS units, I believe I may have seen similar trends; occupying bigger areas but restricting themselves to parts of the home ranges away from human activities. Clearly, this is an area that needs further scrutiny in future research projects if management of the area is to be effective for local bobcat population continuation.

HABITAT SELECTION

Habitat Selection Assumptions and Violations

Due to sample size issues, I followed the Neu et al. (1974) use-availability method of habitat selection analysis. This test method has been well established in the literature and can be used with caution on studies with small sample sizes (Byers et al. 1984, White and Garrott 1990, Alldredge and Ratti 1986, Alldredge and Griswold 2006).

There are several assumptions that apply to resource selection studies in general, and most of these assumptions apply to the Neu et al. (1974) habitat selection method (Allredge and Griswold 2006), and I have violated several of them. By pooling the data of all individuals in this study, I have violated the basic assumption of a random sample of individuals and the results generated from these pooled data may have an increased chance of Type I or II errors (Whitlock and Schuller 2009). I had approximately equal location sample sizes for each bobcat, so no one animal should have biased the results. And, by also testing habitat preferences for each bobcat individually I was able to verify when habitat preferences were consistently for all animals.

The selection of resources made by one individual is assumed to be independent of selections made by other individuals, but this assumption can be violated if animals are territorial (Allredge and Griswold 2006). Bobcats are known to be territorial at particular spatial scales, as denoted by areas of exclusive use (e.g., 50% or 25% core area), where habitat that is readily available for one individual may not be available for the others (Hansen 2007). By pooling the data over the study time period, I am assuming areas of exclusive use will not affect the broader pattern of habitat selection made by the population; this assumption is validated by the high degree of space-use overlap I found, where there was both intra- and inter-sexual home range overlap.

I used the same habitat availability estimates for the 12 months covered in this study, so any changes during that time to the three broad cover categories would violate the assumption of constant habitat availability over time (Byers et al. 1984). During this study the area experienced a winter season where deciduous leaf canopy cover was greatly reduced. Additionally, mining activities spread to new areas around the WMA,

re-appropriating a portion of the open and forested habitat types into the active mine habitat type. Because I assumed habitat availability to be static, errors are possible, but are assumed minimal. Furthermore, because a minimum of 30 independent locations is needed for seasonal analysis, seasonal differences in habitat use were not evaluated (Kernohan et al. 2001). I only sought to evaluate overall preferences. Violating these assumptions to a small degree was not expected to impact conclusions under these circumstances.

Other Confounding Factors

Because of small bobcat and location sample sizes, I was not able to assess the influence of edge effects on habitat selection; which may have a large influence on resource selection (Alldredge and Griswold 2006). I estimated a radio-telemetry error of approximately 200 m radius; but to avoid over-complicating the habitat analysis, I did not test the effect this error could have had on habitat selection. This is not a recommended practice; inferences regarding habitat selection in this paper are conservatively made because radio-telemetry error can reduce the power of statistical analyses (Erickson et al. 2001). Additionally, an increase of mining activities changed the proportion of habitats available during the study; but because I did not have access to multiple orthographic GIS files collected over the study period, I could not accurately estimate what proportion of habitat change occurred.

RESULTS INTERPRETATION

The Neu et al. (1974) chi-square goodness-of-fit test can be applied to data pooled across individuals if all animals are selecting for habitat in a similar manner (Alldredge

and Griswold 2006). Looking at the individual habitat use data for this study, each bobcat selected habitats in a similar fashion; so the application of the goodness-of-fit test on my pooled data set seems appropriate (Alldredge and Griswold 2006). Because I used the same number of locations for each individual bobcat, each individual is weighted equally across the pooled data set (Alldredge and Griswold 2006). Data pooling across individuals restricts inferences to this sample of bobcats and cannot be extrapolated to other bobcats in the population (Erickson et al. 2001). When determining habitat selection, the proportional use of habitats must sum to one, since the use of one habitat will preclude the use of the others at any point in time (Alldredge and Griswold 2006).

When assessing the pattern of habitat use at the study area spatial scale for individual bobcats, it was clear that two individuals (F4 and M1) were driving the overall trend present in the pooled results; i.e., both bobcats selected the open habitat class, while the male avoided the forested habitat class. Without a greater sample size, it is difficult to say whether this finding is of biological significance or not.

For the home range spatial scale pooled data set, I found confounding results for the chi-square analysis; only one of the three habitat types (the open habitat class) fell outside of the confidence intervals when the null was rejected. The three individual bobcats, whose null hypothesis was rejected, showed proportionate use of all three habitat types. Intuitively, it would seem that these three individual cats ought to be driving the trend in the pooled data set to a selection of the open habitat class, especially since the other two cats in the study exhibit proportionate use of that habitat class. These results are likely due to low sample size in the individual analysis, but the chi-square results for each habitat class appear to suggest that M1 is indeed using the open habitat class

disproportionate to its availability. Similarly, the chi-square results for females F2 and F3 indicate that they are using the active habitat class disproportionate to its availability. Only increasing sample size would effectively determine if these results are true for the population or if they are an artifact of the small sample size.

Bobcat Habitat Selection Publications

As previously reported in several studies, many of the individuals in my study used habitat types in proportion to their availability, suggesting that habitat variety may be an important component to a bobcat population's long-term viability. In this study, bobcat movement rates were not calculated, so I cannot comment on whether or not the active mine habitat was crossed at a quicker rate than other habitat classes. Further research into bobcat habitat use in areas of industrial activities should be conducted to shed light onto how bobcats use potentially hostile landscapes. Of the two bobcats in this study that did select or avoid habitats, selection was for one habitat class, whereas avoidance was not clearly indicated. This may highlight the fact that individual bobcats may not use all habitats in similar patterns (White and Garrott 1990); e.g., there have been many published reports of differences between sexes and among times of day or seasons (Tigas et al. 2002, Riley et al. 2003, Tucker et al. 2008).

Many bobcat studies have suggested that habitat selection may relate to prey availability or other resource requirements such as breeding opportunities (Lawhead 1984, Rolley and Warde 1985, Rucker et al. 1989, Chamberlain et al. 2003). Similarly, I suggest that the potential preference of open habitat areas within the study area may be due to more abundant prey. Prey such as rabbit and rodent species, upland game birds and waterfowl, white-tailed deer, and elk have been reported as potential food sources for

bobcats (Jones and Smith 1979, Litvaitis et al. 1986, Powers et al. 1989, Hansen 2007), and hunters or trappers may discard unwanted animal parts while field dressing (R.B. Frederick, 2010, Chair and Professor, Department of Biological Sciences, Eastern Kentucky University, personal communication). All of these species are present on the PVB WMA, but their abundance and density within particular habitat types has not been assessed. However, the lack of habitat preference and high percentage of overlap between individual bobcat home ranges suggests that prey resources may be abundant enough to support a higher density of bobcats than was present in the late 1980's. However, interspecific competition with other mesopredators as well as habitat fragmentation caused by mining activities can limit population growth over time and because these important factors were not assessed in this study any conclusions based on the results present herein are highly speculative. Therefore, bobcat research should continue on the PVB WMA and in neighboring areas. Mining activities in the study area are expected to increase over time, so evaluation of land-use changes and changes in bobcat habitat use, along with population estimates, would help establish bobcat population parameters and management practices that will provide for the species long-term viability in the region.

CHAPTER 6

MANAGEMENT IMPLICATIONS

There are several limitations to carrying out a long-term bobcat monitoring plan, the biggest of which is the expense. It is imperative to balance rigorous scientific study and application with budgetary considerations. Currently, GPS tracking technology remains costly; but with this equipment in rapid development, costs should decrease substantially in the near future. Additionally, GPS radio-collars are currently bulky which limits the size of animals capable of carrying these collars; thereby restricting data collection to larger animals. Aerial tracking of conventional VHF collars remains costly, but is much more cost effective than tracking the animals on foot. The use of interns or graduate students for ground tracking may be seen as a cost-effective means for using VHF technology, but in the rolling to mountainous terrain of eastern Kentucky, ground tracking is imprecise, at best. Only reliable data can establish that statewide bobcat populations are on the rise and genetically viable, potentially allowing harvest limits to be raised and potentially bringing in additional revenues from hunter and trapper license purchases to support continued monitoring.

FUTURE STUDY SUGGESTIONS

Greater sample size

The small number of bobcats that I captured in this study limited the analyses and inferences that could be made from the data collected. While I did work with one private trapper during the trapping season, a potentially more productive method would have

been to use multiple, experienced, local trappers providing trapped cats for a nominal reward. This may have yielded a greater sample size for the effort and expense, and effectively expanded the study area into surrounding properties where additional, larger animals, capable of carrying the larger GPS units may have been present. This trapping method was used effectively by Whitaker (1988) in the same vicinity, resulting in a larger sample size when, presumably, the population of bobcats in the region was lower.

Non-Invasive Genetic Sampling

Non-invasive genetic analyses are commonplace in wildlife research at present, have been used with a wide variety of species, and allow for the genetic sampling of species that are difficult to study (Long et al. 2008). Non-invasive sampling does not require direct handling or observation of individuals (Long et al. 2008). Non-invasive genetic methods offer the ability to sample representatively, over large areas, for cryptic species such as bobcats (Long et al. 2008). Additionally, sampling equipment is inexpensive and resulting molecular data can be identified down to the individual (Long et al. 2008). Hair snares and fecal surveys have been used effectively on felid species, utilizing barbed rub pads and trained scat dogs (McDaniel et al. 2000, Harrison 2006, Long et al. 2008, Tom 2012). Genetic analyses from hair snare and fecal surveys can garner information about species occurrence, relative abundance, genetic variation and relatedness of a population, connectivity between populations, effective population size, or harvest rates (Long et al. 2008). While genetic testing also presents a large expense, the costs may be limited by conducting all testing in-house or with an affiliate university. The costs would potentially be significantly less than a more-intensive telemetry study involving GPS

collars, and the population data produced might be more-directly applied to management decisions.

Harvest Reports

Anthropogenic sources of mortality can affect felid populations greatly if populations are below carrying capacity or are genetically limited by inbreeding (e.g., cheetahs; MacDonald and Loveridge 2010). Two of the most common sources of anthropogenic increases in mortality are hunting and trapping (MacDonald and Loveridge 2010, Litvaitis et al. 1987). In 2011, the season limit for bobcats in Kentucky was only 5 per person, 3 of which could have been taken with a firearm (<http://fw.ky.gov/pdf/deerguide10smallgame.pdf>, accessed 2011). Felid populations have been observed to rebound from population reductions when immigration from adjacent populations is possible (Stahl et al. 2001). Statewide harvest data may be useful in identifying population trends over time (Loveridge et al. 2010). However, these data may not be practical for population analyses beyond generalized trends, though they offer a cost-effective way of identifying trends when independent, less-biased data are not available. Harvest data may also be categorized by region to identify potential over-harvesting trends by county (MacDonald and Loveridge 2010). Like any other sampling design, using harvest data is not without issues; pelt prices, harvest methods, and social and environmental conditions affect harvest rates, while hunters and trappers not keeping records or submitting reports affects data efficacy (Gese 2001). Requiring additional trapper information such as GPS coordinates of trap sites or whether or not a female has reared litters previously, as well as requiring submission from hunters and trappers of bobcat samples such as teeth, hair, feces, skin, and intact guts or whole carcasses, is

likely to enhance ecology and demographic information on current bobcat populations over a broad area.

CHAPTER 7

GPS COLLARED MALE BOBCAT

METHODS

From the full data set for the GPS collared male, I calculated annual and seasonal home range, 50% and 25% core areas, and proportional habitat selection by using the same methods described in Chapter 3, but without statistical tests due to sample size limitations (D. Mundfrom, 2011, personal communication). Additionally, I calculated annual, seasonal, and temporal perceived movement rates by taking the mean straight-line distance between two consecutive locations and dividing by the time interval (Chamberlain et al. 2003).

Seasonal home range analysis was divided into two groups: 1) 4-season analysis, which consisted of the 4 calendar year seasons (Whitaker 1988), and 2) 3-season analysis, which was broken into breeding, rearing, and winter seasons (Chamberlain et al. 2003). The 4-seasons were defined as: spring (1 March 2010 - 31 May 2010), summer (1 June 2010 – 31 August 2010), fall (1 September 2010 – 30 November 2010), and winter (1 December 2010 – 28 February 2011) (Cochrane et al. 2006, Whitaker 1988); the 3-seasons were defined as: breeding (1 February 2010 - 31 May 2010), rearing (1 June 2010 – 30 September 2010), and winter (1 October 2010 – 31 January 2011) (Chamberlain et al. 2003). Temporal analysis was defined as diurnal (a period from 0901 hr to 1659 hr; fixes at 1200 hr, noon), nocturnal (a period from 2101 hr to 0459 hr; fixes at 0000 hr, midnight), and crepuscular (periods from 0500 hr to 0900 hr, and from 1700 hr to 2100 hr; fixes at 0600 and 1800 hrs, respectively), based on average sunrise and

sunset times during the sampling period for the Hazard, KY area (<http://www.sunrisesunset.com>, accessed 2011).

RESULTS

Space Use Estimates

Annual home range size was generated from 1065 GPS locations collected 4 times per day (0000, 0600, 1200, and 1800 hr) from 10 March 2010 to 25 December 2010, and was estimated to be 10 km² by the 95% MCP method, and 5.5 km² by the 95% AK method. The core area was estimated to be 3.3 km² (50% MCP) and 0.04 km² (50% AK), while the central core area estimates were 1.2 km² and 0.003 km² for 25% MCP and AK, respectively (Tables 11a-b, Figure 5).

For the 4-season analysis, the male's home range size declined by approximately half during the fall and winter seasons for the 95% MCP estimate, but only by 38% for the fall 95% AK estimate, and by 65% for the winter (Tables 11a-b, Figure 6). Three season 95% MCP home range sizes for the male were approximately the same as the annual home range size for both the breeding and rearing seasons, but were 43% smaller for the winter season. A similar seasonal pattern held for both the 50% and 25% MCP home range sizes (Table 11a). For the 95% AK sizes, the rearing season estimate was 16% larger than the annual estimate, while the breeding season was 38% smaller and the winter season was 76% smaller than the annual home range size (Table 11b).

Table 11a. The male's space use estimates based on 95%, 50%, and 25% use contours, for the minimum convex polygon (MCP) estimation method.

Season	Estimator	Size (km ²)	No. Locations
Annual	95%	10.0	1065
	50%	3.3	
	25%	1.2	
Breeding 1 Feb 2010 - 31 May 2010	95%	9.8	375
	50%	3.4	
	25%	1.4	
Rearing 1 Jun 2010 – 30 Sep 2010	95%	9.0	448
	50%	3.4	
	25%	1.5	
Winter 1 Oct 2010 – 31 Jan 2011	95%	5.7	313
	50%	1.7	
	25%	0.8	
Spring 1 Mar 2010 - 31 May 2010	95%	9.8	305
	50%	3.4	
	25%	1.4	
Summer 1 Jun 2010 – 31 Aug 2010	95%	9.0	333
	50%	4.3	
	25%	2.5	
Fall 1 Sep 2010 – 30 Nov 2010	95%	5.7	352
	50%	2.7	
	25%	1.3	
Winter 1 Dec 2010 – 28 Feb 2011	95%	4.5	79
	50%	0.5	
	25%	0.2	

Table 11b. The male's space use estimates based on 95%, 50%, and 25% use contours, for the adaptive kernel (AK) estimation method.

Season	Estimator	Size (km ²)	No. Locations
Annual	95%	5.5	1065
	50%	0.04	
	25%	0.003	
Breeding 1 Feb 2010 - 31 May 2010	95%	3.4	375
	50%	0.04	
	25%	0.004	
Rearing 1 Jun 2010 – 30 Sep 2010	95%	6.4	448
	50%	0.07	
	25%	0.006	
Winter 1 Oct 2010 – 31 Jan 2011	95%	1.3	313
	50%	0.008	
	25%	0.001	
Spring 1 Mar 2010 - 31 May 2010	95%	3.4	305
	50%	0.04	
	25%	0.004	
Summer 1 Jun 2010 – 31 Aug 2010	95%	5.0	333
	50%	0.06	
	25%	0.005	
Fall 1 Sep 2010 – 30 Nov 2010	95%	3.4	352
	50%	0.03	
	25%	0.006	
Winter 1 Dec 2010 – 28 Feb 2011	95%	1.9	79
	50%	0.07	
	25%	0.005	

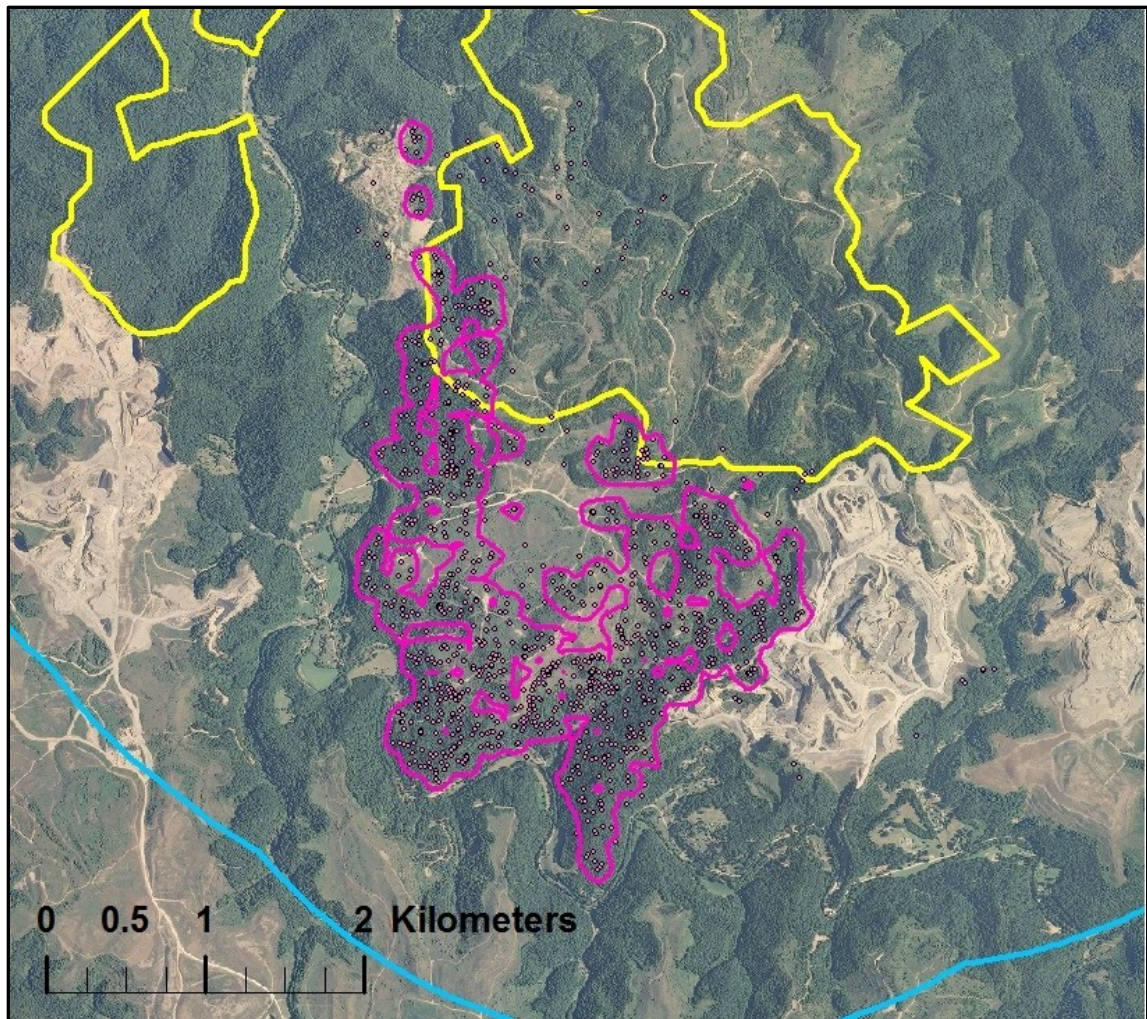


Figure 5. The male's 95% adaptive kernel home range mapped in relation to the borders of the Paul Van Booven wildlife management area, Kentucky, March - December, collected at a rate of 4 locations per day for 266 days, 1065 locations.

The PVB border is represented by the yellow colored line, the 95% home range is represented by the dark pink polygon, and the locations for M1 are in pink; the blue line is part of the southern boundary of the study area.

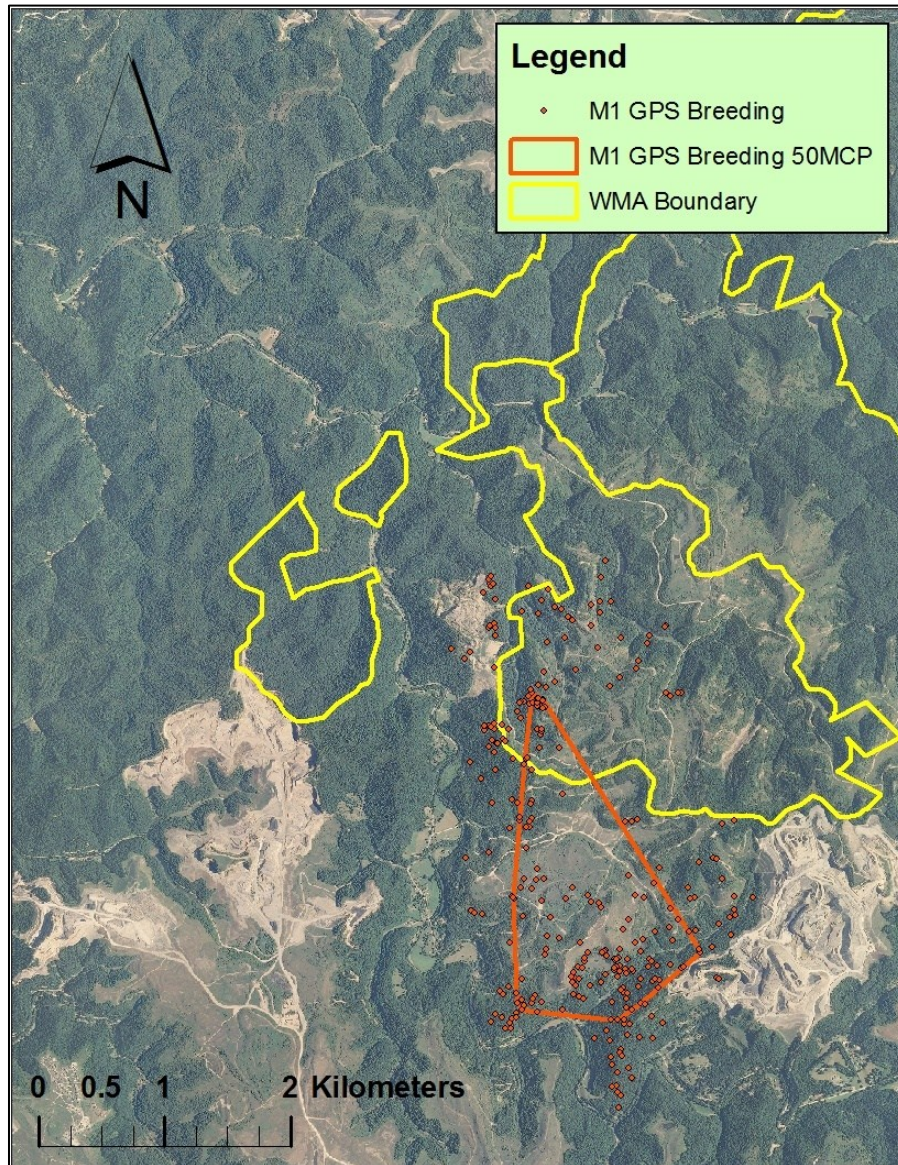


Figure 6. The male's 50% core area as estimated by minimum convex polygon (MCP) during the breeding season of 2011, in southeastern Kentucky.

The PVB border is represented by the yellow colored line, the 50% core area is represented by the reddish polygon, and the male's locations are in the same reddish color.

Habitat Selection Analysis

Annually, the GPS collared male bobcat selected for forested habitats over open or active mine habitat classes at all spatial scales (Tables 12 and A-2, Figure 7). This trend followed through both 3- and 4-season partitioning, at all spatial scales. There was proportionally more forest habitat available at all spatial scales than open or active mine habitat classes, but the male's locations were in forested habitat at a greater proportion than available at all scales of analysis. The greater availability of forested habitat within the entire study area (73%; Table 7a) versus within the 95%, 50%, and 25% annual and seasonal polygons (60-67%; Table 12), however, could partly explain the higher use of this habitat (Erickson et al. 2001). The proportion of bobcat locations in forest was higher than the study area availability, however, in all except the winter season (Table 12)

Table 12. The male's annual and seasonal percent available habitat within the 95% minimum convex polygon spatial scale, with the number of locations in parentheses next to the percentage.

Season	Habitat Class		
	Active	Forested	Open
Annual	16 (n=123)	67 (n=778)	17 (n=110)
Breeding	15 (n=29)	67 (n=241)	18 (n=18)
Rearing	23 (n=53)	60 (n=326)	17 (n=46)
Winter	15.5 (n= 37)	64.5 (n=214)	20 (n=45)

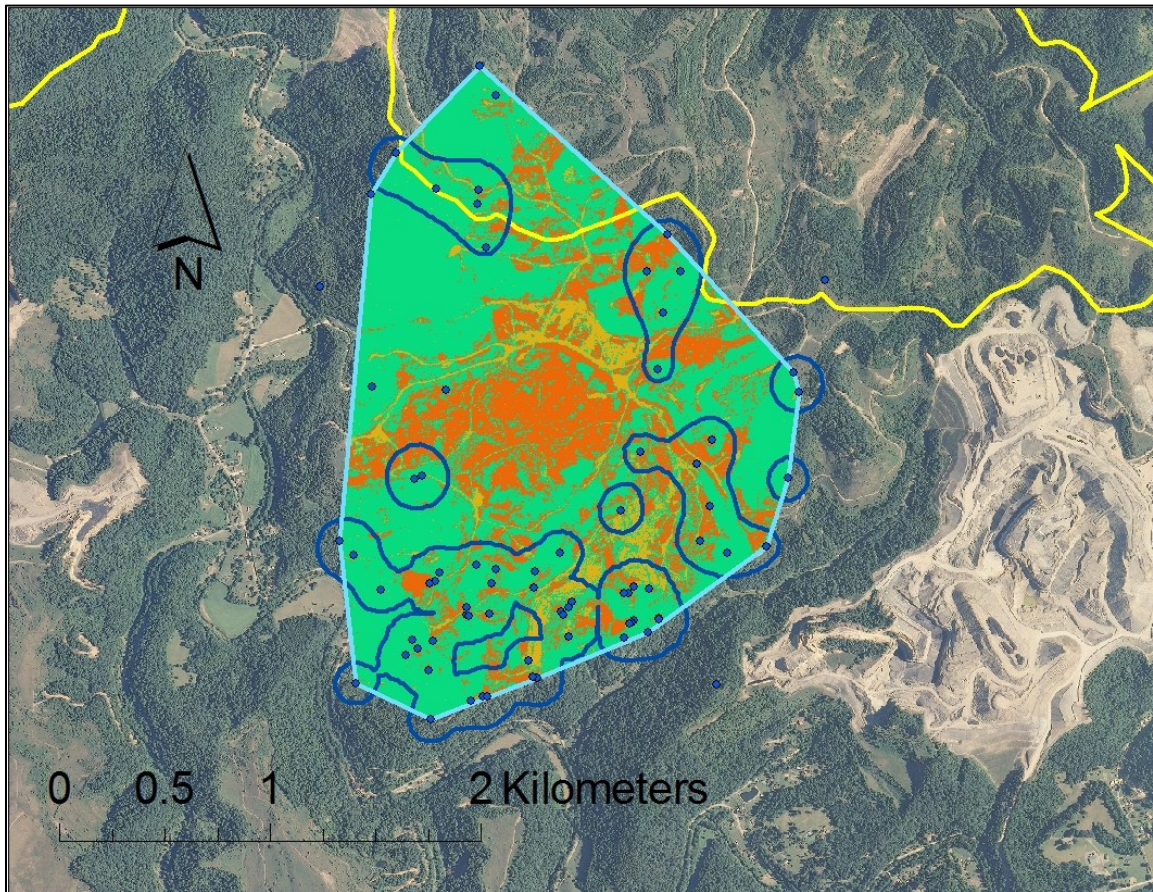


Figure 7. An example of the seasonal habitat reclassification, in this instance the winter season, overlaid with the male's 95% adaptive kernel (AK) home range contour with the associated locations.

The PVB management area border is represented by the yellow line, while the 95% AK polygon is in dark blue, and the associated locations are also in dark blue. The reclassified habitat types are: Forested in aqua, Open in orange, and Active in gold.

Movement Rates

The annual movement rate for the GPS collared male was 0.12 km/h, while seasonal movement rates varied from 0.11 km/h in the breeding season to 0.13 km/h in the winter (Table 13). Movement rates varied more with time of day than among seasons, with the lowest rate being during the diurnal period and higher, approximately equal rates during crepuscular and nocturnal periods (Table 14).

Table 13. Seasonal movement rates for the male, in kilometers per hour, with the number of associated locations.

Season	Movement Rate (km/h)	No. Locations
Breeding	0.110	278
Rearing	0.114	412
Winter	0.127	294
Annual	0.117	984

Seasons: breeding season from 1 February 2010 to 31 May 2010, rearing sesason from 1 June 2010 to 30 September 2010, and winter season from 1 October 2010 to 31 January 2011. The perceived movement rates were calculated by taking the mean straight-line distance between two locations and dividing by the time interval.

Table 14. Time of day movement rates for the male, with the associated number of locations.

Time of Day	Time of Day Class	Movement Rate (km/h)	No. Locations
(12:00)	Diurnal	0.088	228
(06:00, 18:00)	Crepuscular	0.125	490
(0:00)	Nocturnal	0.123	261

The perceived movement rates were calculated by taking the mean straight-line distance between two consecutive locations and dividing by the time interval of 6 hours.

Temporal periods were defined as diurnal (a period from 0901 hr to 1659 hr; fixes at 1200 hr, noon), nocturnal (a period from 2101 hr to 0459 hr; fixes at 0000 hr, midnight), and crepuscular (periods from 0500 hr to 0900 hr, and from 1700 hr to 2100 hr; fixes at 0600 and 1800 hrs, respectively), based on average sunrise and sunset times during the sampling period for the Hazard, KY area.

DISCUSSION

It has been suggested previously that human disturbance may affect bobcat movements and space use (Bailey 1979, Chamberlain et al. 2003). Seasonal movement rates, particularly during the winter, may be influenced by the disturbance caused by various hunting and trapping seasons causing bobcats to move quickly to areas of refuge (Chamberlain et al. 1999). Additionally, a declining prey base during the winter months forces individuals to move over larger areas to meet resource demands (Odum 1955, Chamberlain et al. 2003). In my study, winter home range, core area, and central core area sizes declined greatly, which may suggest that prey densities were not greatly affected by the winter environmental changes, that hunting produced more opportunities for feeding on wounded animals or animal parts, or that prey are more detectable in the winter season. It may also suggest that there may be another competing factor causing the male to move at greater speeds over a smaller area, such as overlapping space with another male.

Rucker et al. (1989) used locations at 15 min intervals to calculate movement rates on a study site in Arkansas, and found a mean annual movement rate of 2.2 km/h. Overall, bobcats moved the least in the fall and the most during the spring seasons, but male's moved the greatest in the winter and the spring, and moved less in the summer than females.

Another previous study found that bobcats generally moved faster at nocturnal times, and in the winter season (Chamberlain et al. 2003). The authors also found a mean male annual movement rate of 0.38 km/h, with a breeding movement rate of 0.38 km/h, rearing movement rate of 0.34 km/h, and a winter movement rate of 0.41 km/h (Chamberlain et

al. 2003). Looking at movement rates for times of day, male's had a crepuscular rate of 0.37 km/h, diurnal rate of 0.34 km/h, and a nocturnal movement rate of 0.43 km/h (Chamberlain et al. 2003).

Overall, the male's movement rates were lower than previously reported for bobcats in the southeast. However, because I used GPS technology my movement rates are not affected by the high error associated with VHF telemetry which can inflate movement rates. Conversely, the GPS unit was programmed to take locations once every 6 hours whereas past studies using VHF transmitters based estimates on locations once every 15 minutes in a sequential method, perhaps showing movement at a finer temporal scale that would have been missed in my study. Because the movement rates are based on one individual, they are considered to be very rough gauges of rates for the population using PVB WMA.

While GPS technology currently has its limitations, the precision of data collected by this method allowed me to view trends in space use such as movement within corridors of forested habitat and avoidance of particular land formations (e.g., steep slopes leading to open habitats). This data precision may be why I see a confounding of results: preference for forested habitat when assessing the entire GPS data set versus the apparent preference for open habitat when looking at the VHF data or the sub-sampled male data set. Continuing to mark bobcats with this technology may provide greater insights into contradictions in the current data such as these, and research using GPS radio-telemetry will be important in adequately determining bobcat population parameters and long-term viability.

BRIEF CONCLUSION SUMMARY

Without a greater sample size or finer scale location data (e.g., Kie et al. 2010), it is difficult to draw definitive conclusions on habitat and space use by bobcats in southeastern Kentucky. However, the data do seem to suggest that forested habitats play an important role in monitoring bobcat populations in the vicinity of extractive industrial activities. Smaller and lighter GPS/GSM collars that allow the marking of smaller female bobcats would provide the precise data needed, if care is taken to make certain there is adequate cellular provider coverage in the study area so data can be collected from animals in a timely manner. Though I programmed the GPS collars to transmit collected locations daily, more often than not these locations were transmitted at various time periods when the bobcat passed through an area with cellular coverage, sometimes not sending data for well over a week.

Therefore, I suggest that Kentucky wildlife managers' future research aim to increase sample size, use data from locations collected at smaller time intervals, conduct simultaneous non-invasive genetic surveys, and continue to research bobcat ecology in areas that have a representative landscape of the broader region such as areas impacted by logging and surface mining.

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APPENDIX A

SUPPLEMENTAL PHOTOGRAPHS, TABLES, AND FIGURES

Table A-1. Percent available habitat for each individual at the 95%, 50%, and 25% minimum convex polygon (MCP) spatial scales, including the number of actual VHF locations (or sub-sampled GPS locations for M1) that fall within each habitat class.

Bobcat ID	Spatial Scale	Habitat Class	Area (km ²)	No. Locations	% of Available Habitat
NA	Study area	Active	17	NA	18
NA	Study area	Forested	70	NA	73
NA	Study area	Open	9	NA	9
F1	95MCP	Active	1.4	4	15
F1	95MCP	Forested	6.8	49	74
F1	95MCP	Open	1	7	11
F1	50MCP	Active	0.4	2	15
F1	50MCP	Forested	1.7	24	63
F1	50MCP	Open	0.5	7	19
F1	25MCP	Active	0.12	1	15
F1	25MCP	Forested	0.5	12	63
F1	25MCP	Open	0.17	2	21
F2	95MCP	Active	3.6	11	16
F2	95MCP	Forested	17	42	76
F2	95MCP	Open	1.8	5	8
F2	50MCP	Active	0.7	7	18
F2	50MCP	Forested	3	23	79
F2	50MCP	Open	0.13	0	3
F2	25MCP	Active	0.13	5	12
F2	25MCP	Forested	0.98	10	89
F2	25MCP	Open	0.03	0	3
F3	95MCP	Active	4	6	16
F3	95MCP	Forested	20.3	52	80
F3	95MCP	Open	1	1	4
F3	50MCP	Active	0.7	4	16
F3	50MCP	Forested	3.7	27	82
F3	50MCP	Open	0.14	0	3
F3	25MCP	Active	0.12	0	10
F3	25MCP	Forested	1.09	15	91
F3	25MCP	Open	0.02	0	2

Table A-1 (continued)

Bobcat ID	Spatial Scale	Habitat Class	Area (km ²)	No. Locations	% of Available Habitat
F4	95MCP	Active	2.5	8	20
F4	95MCP	Forested	7.9	37	63
F4	95MCP	Open	2.1	13	17
F4	50MCP	Active	0.5	3	19
F4	50MCP	Open	0.8	6	31
F4	25MCP	Active	0.09	1	18
F4	25MCP	Forested	0.28	11	56
F4	25MCP	Open	0.1	3	20
M1	95MCP	Active	0.8	10	16
M1	95MCP	Forested	2.9	28	59
M1	95MCP	Open	1.2	23	24
M1	50MCP	Active	0.2	3	15
M1	50MCP	Forested	0.6	14	46
M1	50MCP	Open	0.5	14	38
M1	25MCP	Active	0.08	1	13
M1	25MCP	Forested	0.27	7	45
M1	25MCP	Open	0.28	7	47

Table A-2. Percent available habitat for M1 at the 95%, 50%, and 25% minimum convex polygon (MCP) spatial scales, including the number of actual GPS locations that fall within each habitat class, for annual and both 3- and 4-seasonal seasons.

Time Frame	Spatial Scale	Habitat Class	Area (km2)	No Locs	Percent of Available Habitat
	Study area	Active	17	NA	18
	Study area	Forested	70	NA	73
	Study area	Open	9	NA	9
Annual	95MCP	Active	1.59	123	16
Annual	95MCP	Forested	6.69	778	67
Annual	95MCP	Open	1.75	110	18
Annual	50MCP	Active	0.55	66	17
Annual	50MCP	Forested	1.86	389	56
Annual	50MCP	Open	0.87	76	26
Annual	25MCP	Active	0.22	33	18
Annual	25MCP	Forested	0.74	195	62
Annual	25MCP	Open	0.23	38	7
Fall	95MCP	Active	0.84	35	15
Fall	95MCP	Forested	3.61	250	63
Fall	95MCP	Open	1.21	47	21
Fall	50MCP	Active	0.53	25	20
Fall	50MCP	Forested	1.25	115	46
Fall	50MCP	Open	0.89	35	33
Fall	25MCP	Active	0.2	8	15
Fall	25MCP	Forested	0.6	65	46
Fall	25MCP	Open	0.51	10	39
Winter	95MCP	Active	0.76	12	17
Winter	95MCP	Forested	2.63	48	58
Winter	95MCP	Open	1.16	13	26
Winter	50MCP	Active	0.1	7	20
Winter	50MCP	Forested	0.3	23	60
Winter	50MCP	Open	0.11	8	22
Winter	25MCP	Active	0.05	2	25
Winter	25MCP	Forested	0.13	12	65
Winter	25MCP	Open	0.06	5	30

Table A-2 (continued)

Time Frame	Spatial Scale	Habitat Class	Area (km ²)	No Locs	Percent of Available Habitat
Spring	95MCP	Active	1.5	29	15
Spring	95MCP	Forested	6.61	241	67
Spring	95MCP	Open	1.71	18	17
Spring	50MCP	Active	0.61	15	18
Spring	50MCP	Forested	1.92	119	56
Spring	50MCP	Open	0.91	16	27
Spring	25MCP	Active	0.24	6	17
Spring	25MCP	Forested	0.74	63	53
Spring	25MCP	Open	0.42	7	0.3
Summer	95MCP	Active	2.08	39	23
Summer	95MCP	Forested	5.44	242	60
Summer	95MCP	Open	1.52	33	17
Summer	50MCP	Active	0.7	24	16
Summer	50MCP	Forested	2.52	118	59
Summer	50MCP	Open	1.05	23	24
Summer	25MCP	Active	0.45	15	18
Summer	25MCP	Forested	1.38	54	55
Summer	25MCP	Open	0.64	13	26
Breeding	95MCP	Active	1.5	29	15
Breeding	95MCP	Forested	6.61	241	67
Breeding	95MCP	Open	1.71	18	17
Breeding	50MCP	Active	0.61	16	18
Breeding	50MCP	Forested	1.92	118	56
Breeding	50MCP	Open	0.91	16	27
Breeding	25MCP	Active	0.24	5	17
Breeding	25MCP	Forested	0.74	61	53
Breeding	25MCP	Open	0.42	8	30

Table A-2 (continued)

Time Frame	Spatial Scale	Habitat Class	Area (km ²)	No Locs	Percent of Available Habitat
Rearing	95MCP	Active	2.06	53	23
Rearing	95MCP	Forested	5.41	326	60
Rearing	95MCP	Open	1.52	46	17
Rearing	50MCP	Active	0.57	28	17
Rearing	50MCP	Forested	1.98	165	58
Rearing	50MCP	Open	0.88	32	26
Rearing	25MCP	Active	0.27	14	18
Rearing	25MCP	Forested	0.84	80	56
Rearing	25MCP	Open	0.41	17	27
3S Winter	95MCP	Active	0.87	37	15
3S Winter	95MCP	Forested	3.65	214	64
3S Winter	95MCP	Open	1.15	45	20
3S Winter	50MCP	Active	0.27	21	16
3S Winter	50MCP	Forested	0.99	108	58
3S Winter	50MCP	Open	0.46	26	27
3S Winter	25MCP	Active	0.12	6	15
3S Winter	25MCP	Forested	0.45	56	56
3S Winter	25MCP	Open	0.24	15	30

Table A-3a. Chi-square goodness-of-fit test, with Bonferroni Z calculated confidence intervals, for habitat

selection at the 95% MCP home range spatial scale of 3 habitat classes, by pooled bobcat individuals, using the

Neu et al. (1974) method.

Habitat Class	Spatial Scale	Area (km ²)	Proportion of available habitat	Observed Locations	Expected Locations	Proportion of observed locations in each area	X ²	df	P	Confidence interval on proportion of habitat use (<i>p</i>)	Avoided/ Selected/ Proportionate/ Ho/Ha
Active	95% MCP	12.3	0.166	39	49	0.132	2.0			0.086 < <i>p</i> < 0.178	Proportionate
Forested	95% MCP	54.9	0.739	208	219	0.703	0.5			0.641 < <i>p</i> < 0.765	Proportionate
Open	95% MCP	7.1	0.096	49	28	0.166	15.2			0.115 < <i>p</i> < 0.216	Selected
Total		74.3	1.000	296	296	1.000	17.7	2	P < 0.001		Ha

Abbreviations: X², Chi-square value; df, degrees of freedom; P, significance level; *, confidence intervals not calculated because the null was not rejected; *p*, represents the theoretical occurrence of locations compared to the proportion of available habitat, in order to determine if the hypothesis of proportional use is accepted or rejected.

Table A-3b. Chi-square goodness-of-fit test, with Bonferroni Z calculated confidence intervals, for habitat selection at the 95% MCP home range spatial scale of 3 habitat classes, for bobcat F1, using the Neu et al. (1974) method.

Habitat Class	Spatial Scale	Area (km ²)	Proportion of available habitat	Observed Locations	Expected Locations	Proportion of observed locations in each area	X ²	df	P	Confidence interval on proportion of habitat use (<i>p</i>)	Avoided/ Selected/ Proportionate/ Ho/Ha
Active	95% MCP	1.4	0.152	4	9	0.07	2.9			*	Proportionate
Forested	95% MCP	6.8	0.739	49	44	0.82	0.5			*	Proportionate
Open	95% MCP	1	0.109	7	7	0.12	0.0			*	Proportionate
Total		9.2	1.000	60	60	1.00	3.4	2	P > 0.10		Ho

Abbreviations: X², Chi-square value; df, degrees of freedom; P, significance level; *, confidence intervals not calculated because the null was not rejected; *p*, represents the theoretical occurrence of locations compared to the proportion of available habitat, in order to determine if the hypothesis of proportional use is accepted or rejected.

Table A-3c. Chi-square goodness-of-fit test, with Bonferroni Z calculated confidence intervals, for habitat selection at the 95% MCP home range spatial scale of 3 habitat classes, for bobcat F2, using the Neu et al. (1974) method.

Habitat Class	Spatial Scale	Area (km ²)	Proportion of available habitat	Observed Locations	Expected Locations	Proportion of observed locations in each area	X ²	df	P	Confidence interval on proportion of habitat use (<i>p</i>)	Avoided/ Selected/ Proportionate/ Ho/Ha
Active	95% MCP	3.6	0.161	11	9	0.19	0.302			0.069 < <i>p</i> < 0.310	Proportionate
Forested	95% MCP	17	0.759	42	44	0.72	0.093			0.587 < <i>p</i> < 0.861	Proportionate
Open	95% MCP	1.8	0.080	5	5	0.09	0.025			0.000 < <i>p</i> < 0.172	Proportionate
Total		22.4	1.000	58	58	1.00	0.419	2	P < 0.001		Ha

Abbreviations: X², Chi-square value; df, degrees of freedom; P, significance level; *, confidence intervals not calculated because the null was not rejected; *p*, represents the theoretical occurrence of locations compared to the proportion of available habitat, in order to determine if the hypothesis of proportional use is accepted or rejected.

Table A-3d. Chi-square goodness-of-fit test, with Bonferroni Z calculated confidence intervals, for habitat selection at the 95% MCP home range spatial scale of 3 habitat classes, for bobcat F3, using the Neu et al. (1974) method.

Habitat Class	Spatial Scale	Area (km ²)	Proportion of available habitat	Observed Locations	Expected Locations	Proportion of observed locations in each area	X ²	df	P	Confidence interval on proportion of habitat use (<i>p</i>)	Avoided/ Selected/ Proportionate/ Ho/Ha
Active	95% MCP	4	0.158	6	9	0.10	1.2			$0.009 < p < 0.195$	Proportionate
Forested	95% MCP	20.3	0.802	52	47	0.88	0.5			$0.782 < p < 0.981$	Proportionate
Open	95% MCP	1	0.040	1	2	0.017	0.8			$0.000 < p < 0.057$	Proportionate
Total		25.3	1.000	59	59	1.00	2.4	2	$P < 0.001$		Ha

Abbreviations: X², Chi-square value; df, degrees of freedom; P, significance level; *, confidence intervals not calculated because the null was not rejected; *p*, represents the theoretical occurrence of locations compared to the proportion of available habitat, in order to determine if the hypothesis of proportional use is accepted or rejected.

Table A-3e. Chi-square goodness-of-fit test, with Bonferroni Z calculated confidence intervals, for habitat selection at the 95% MCP home range spatial scale of 3 habitat classes, for bobcat F4, using the Neu et al. (1974) method.

Habitat Class	Spatial Scale	Area (km ²)	Proportion of available habitat	Observed Locations	Expected Locations	Proportion of observed locations in each area	X ²	df	P	Confidence interval on proportion of habitat use (<i>p</i>)	Avoided/ Selected/ Proportionate/ Ho/Ha
Active	95% MCP	2.5	0.200	8	12	0.14	1.1			*	Proportionate
Forested	95% MCP	7.9	0.632	37	37	0.64	0.0			*	Proportionate
Open	95% MCP	2.1	0.168	13	10	0.22	1.1			*	Proportionate
Total		12.5	1.000	58	58	1.00	2.2	2	P > 0.25		Ho

Abbreviations: X², Chi-square value; df, degrees of freedom; P, significance level; *, confidence intervals not calculated because the null was not rejected; *p*, represents the theoretical occurrence of locations compared to the proportion of available habitat, in order to determine if the hypothesis of proportional use is accepted or rejected.

Table A-3f. Chi-square goodness-of-fit test, with Bonferroni Z calculated confidence intervals, for habitat selection at the 95% MCP home range spatial scale of 3 habitat classes, for bobcat M1, using the Neu et al. (1974) method.

Habitat Class	Spatial Scale	Area (km ²)	Proportion of available habitat	Observed Locations	Expected Locations	Proportion of observed locations in each area	X ²	df	P	Confidence interval on proportion of habitat use (<i>p</i>)	Avoided/ Selected/ Proportionate/ Ho/Ha
Active	95% MCP	0.8	0.163	10	10	0.164	0.0			$0.054 < p < 0.274$	Proportionate
Forested	95% MCP	2.9	0.592	28	36	0.459	1.8			$0.311 < p < 0.607$	Proportionate
Open	95% MCP	1.2	0.245	23	15	0.377	4.3			$0.233 < p < 0.521$	Proportionate
Total		4.9	1.000	61	61	1.00	6.2	2	$P < 0.05$		Ha

Abbreviations: X², Chi-square value; df, degrees of freedom; P, significance level; *, confidence intervals not calculated because the null was not rejected; *p*, represents the theoretical occurrence of locations compared to the proportion of available habitat, in order to determine if the hypothesis of proportional use is accepted or rejected.

Table A-4a. Chi-square goodness-of-fit test, with Bonferroni Z calculated confidence intervals, for habitat selection at the 50% MCP core area spatial scale of 3 habitat classes, by pooled bobcat individuals, using the Neu et al. (1974) method.

Habitat Class	Spatial Scale	Area (km ²)	Proportion of available habitat	Observed Locations	Expected Locations	Proportion of observed locations in each area	X ²	df	P	Confidence interval on proportion of habitat use (<i>p</i>)	Avoided/ Selected/ Proportionate/ Ho/Ha
Active	50% MCP	2.5	0.167	19	26	0.12	1.8			*	Proportionate
Forested	50% MCP	10.4	0.695	109	108	0.70	0.0			*	Proportionate
Open	50% MCP	2.07	0.138	27	21	0.17	1.4			*	Proportionate
Total		14.97	1.000	155	155	1.00	3.3	2	P > 0.10		Ho

Abbreviations: X², Chi-square value; df, degrees of freedom; P, significance level; *, confidence intervals not calculated because the null was not rejected; *p*, represents the theoretical occurrence of locations compared to the proportion of available habitat, in order to determine if the hypothesis of proportional use is accepted or rejected.

Table A-4b. Chi-square goodness-of-fit test, with Bonferroni Z calculated confidence intervals, for habitat selection at the 50% MCP core area spatial scale of 3 habitat classes, for bobcat F1, using the Neu et al. (1974) method.

Habitat Class	Spatial Scale	Area (km ²)	Proportion of available habitat	Observed Locations	Expected Locations	Proportion of observed locations in each area	X ²	df	P	Confidence interval on proportion of habitat use (<i>p</i>)	Avoided/ Selected/ Proportionate/ Ho/Ha
Active	50% MCP	0.4	0.154	2	5	0.06	1.9			*	Proportionate
Forested	50% MCP	1.7	0.654	24	22	0.73	0.3			*	Proportionate
Open	50% MCP	0.5	0.192	7	6	0.21	0.1			*	Proportionate
Total		2.6	1.000	33	33	1.00	2.2	2	P > 0.25		Ho

Abbreviations: X², Chi-square value; df, degrees of freedom; P, significance level; *, confidence intervals not calculated because the null was not rejected; *p*, represents the theoretical occurrence of locations compared to the proportion of available habitat, in order to determine if the hypothesis of proportional use is accepted or rejected.

Table A-4c. Chi-square goodness-of-fit test, with Bonferroni Z calculated confidence intervals, for habitat selection at the 50% MCP core area spatial scale of 3 habitat classes, for bobcat F2, using the Neu et al. (1974) method.

Habitat Class	Spatial Scale	Area (km ²)	Proportion of available habitat	Observed Locations	Expected Locations	Proportion of observed locations in each area	X ²	df	P	Confidence interval on proportion of habitat use (<i>p</i>)	Avoided/ Selected/ Proportionate/ Ho/Ha
Active	50% MCP	0.7	0.183	7	5	0.23	0.4			*	Proportionate
Forested	50% MCP	3.0	0.783	23	23	0.77	0.0			*	Proportionate
Open	50% MCP	0.13	0.034	0	1	0.00	1.0			*	Proportionate
Total		3.83	1.000	30	30	1.00	1.4	2	P > 0.25		Ho

Abbreviations: X², Chi-square value; df, degrees of freedom; P, significance level; *, confidence intervals not calculated because the null was not rejected; *p*, represents the theoretical occurrence of locations compared to the proportion of available habitat, in order to determine if the hypothesis of proportional use is accepted or rejected.

Table A-4d. Chi-square goodness-of-fit test, with Bonferroni Z calculated confidence intervals, for habitat selection at the 50% MCP core area spatial scale of 3 habitat classes, for bobcat F3, using the Neu et al. (1974) method.

Habitat Class	Spatial Scale	Area (km ²)	Proportion of available habitat	Observed Locations	Expected Locations	Proportion of observed locations in each area	X ²	df	P	Confidence interval on proportion of habitat use (<i>p</i>)	Avoided/ Selected/ Proportionate/ Ho/Ha
Active	50% MCP	0.7	0.154	4	5	0.13	0.1			*	Proportionate
Forested	50% MCP	3.7	0.815	27	25	0.87	0.1			*	Proportionate
Open	50% MCP	0.14	0.031	0	1	0.00	1.0			*	Proportionate
Total		4.54	1.000	31	31	1.00	1.2	2	P > 0.50		Ho

Abbreviations: X², Chi-square value; df, degrees of freedom; P, significance level; *, confidence intervals not calculated because the null was not rejected; *p*, represents the theoretical occurrence of locations compared to the proportion of available habitat, in order to determine if the hypothesis of proportional use is accepted or rejected.

Table A-4e. Chi-square goodness-of-fit test, with Bonferroni Z calculated confidence intervals, for habitat selection at the 50% MCP core area spatial scale of 3 habitat classes, for bobcat F4, using the Neu et al. (1974) method.

Habitat Class	Spatial Scale	Area (km ²)	Proportion of available habitat	Observed Locations	Expected Locations	Proportion of observed locations in each area	X ²	df	P	Confidence interval on proportion of habitat use (<i>p</i>)	Avoided/ Selected/ Proportionate/ Ho/Ha
Active	50% MCP	0.5	0.185	3	6	0.10	1.2			*	Proportionate
Forested	50% MCP	1.4	0.519	21	16	0.70	1.9			*	Proportionate
Open	50% MCP	0.8	0.296	6	9	0.20	0.9			*	Proportionate
Total		2.7	1.000	30	30	1.00	4.0	2	P > 0.10		Ho

Abbreviations: X², Chi-square value; df, degrees of freedom; P, significance level; *, confidence intervals not calculated because the null was not rejected; *p*, represents the theoretical occurrence of locations compared to the proportion of available habitat, in order to determine if the hypothesis of proportional use is accepted or rejected.

Table A-4f. Chi-square goodness-of-fit test, with Bonferroni Z calculated confidence intervals, for habitat selection at the 50% MCP core area spatial scale of 3 habitat classes, for bobcat M1, using the Neu et al. (1974) method.

Habitat Class	Spatial Scale	Area (km ²)	Proportion of available habitat	Observed Locations	Expected Locations	Proportion of observed locations in each area	X ²	df	P	Confidence interval on proportion of habitat use (<i>p</i>)	Avoided/ Selected/ Proportionate/ Ho/Ha
Active	50% MCP	0.2	0.154	3	5	0.10	0.7			*	Proportionate
Forested	50% MCP	0.6	0.462	14	14	0.45	0.0			*	Proportionate
Open	50% MCP	0.5	0.385	14	12	0.45	0.4			*	Proportionate
Total		1.3	1.000	31	31	1.00	1.0	2	P > 0.50		Ho

Abbreviations: X², Chi-square value; df, degrees of freedom; P, significance level; *, confidence intervals not calculated because the null was not rejected; *p*, represents the theoretical occurrence of locations compared to the proportion of available habitat, in order to determine if the hypothesis of proportional use is accepted or rejected.

Table A-5a. Chi-square goodness-of-fit test, with Bonferroni Z calculated confidence intervals, for habitat selection at the 25% MCP core area spatial scale of 3 habitat classes, by pooled bobcat individuals, using the Neu et al. (1974) method.

Habitat Class	Spatial Scale	Area (km ²)	Proportion of available habitat	Observed Locations	Expected Locations	Proportion of observed locations in each area	X ²	df	P	Confidence interval on proportion of habitat use (<i>p</i>)	Avoided/ Selected/ Proportionate/ Ho/Ha
Active	25% MCP	0.54	0.127	8	10	0.11	0.2			*	Proportionate
Forested	25% MCP	3.12	0.732	55	55	0.73	0.0			*	Proportionate
Open	25% MCP	0.6	0.141	12	11	0.16	0.2			*	Proportionate
Total		4.26	1.000	75	75	1.00	0.4	2	P > 0.75		Ho

Abbreviations: X², Chi-square value; df, degrees of freedom; P, significance level; *, confidence intervals not calculated because the null was not rejected; *p*, represents the theoretical occurrence of locations compared to the proportion of available habitat, in order to determine if the hypothesis of proportional use is accepted or rejected.

Table A-5b. Chi-square goodness-of-fit test, with Bonferroni Z calculated confidence intervals, for habitat selection at the 25% MCP core area spatial scale of 3 habitat classes, for bobcat FI, using the Neu et al. (1974) method.

Habitat Class	Spatial Scale	Area (km ²)	Proportion of available habitat	Observed Locations	Expected Locations	Proportion of observed locations in each area	X ²	df	P	Confidence interval on proportion of habitat use (<i>p</i>)	Avoided/ Selected/ Proportionate/ Ho/Ha
Active	25% MCP	0.12	0.152	1	2	0.07	0.7			*	Proportionate
Forested	25% MCP	0.5	0.633	12	9	0.80	0.7			*	Proportionate
Open	25% MCP	0.17	0.215	2	3	0.13	0.5			*	Proportionate
Total		0.79	1.000	15	15	1.00	1.8	2	P > 0.25		Ho

Abbreviations: X², Chi-square value; df, degrees of freedom; P, significance level; *, confidence intervals not calculated because the null was not rejected; *p*, represents the theoretical occurrence of locations compared to the proportion of available habitat, in order to determine if the hypothesis of proportional use is accepted or rejected.

Table A-5c. Chi-square goodness-of-fit test, with Bonferroni Z calculated confidence intervals, for habitat selection at the 25% MCP core area spatial scale of 3 habitat classes, for bobcat F2, using the Neu et al. (1974) method.

Habitat Class	Spatial Scale	Area (km ²)	Proportion of available habitat	Observed Locations	Expected Locations	Proportion of observed locations in each area	X ²	df	P	Confidence interval on proportion of habitat use (<i>p</i>)	Avoided/ Selected/ Proportionate/ Ho/Ha
Active	25% MCP	0.13	0.114	5	2	0.33333	6.3			0.188 < <i>p</i> < 0.478	Avoided
Forested	25% MCP	0.98	0.860	10	13	0.67	0.6			0.522 < <i>p</i> < 0.812	Selected
Open	25% MCP	0.03	0.026	0	0	0.00	0.4			0.000 < <i>p</i> < 0.000	NA
Total		1.14	1.000	15	15	1.00	7.4	2	P < 0.025		Ha

Abbreviations: X², Chi-square value; df, degrees of freedom; P, significance level; *, confidence intervals not calculated because the null was not rejected; *p*, represents the theoretical occurrence of locations compared to the proportion of available habitat, in order to determine if the hypothesis of proportional use is accepted or rejected.

Table A-5d. Chi-square goodness-of-fit test, with Bonferroni Z calculated confidence intervals, for habitat selection at the 25% MCP core area spatial scale of 3 habitat classes, for bobcat F3, using the Neu et al. (1974) method.

Habitat Class	Spatial Scale	Area (km ²)	Proportion of available habitat	Observed Locations	Expected Locations	Proportion of observed locations in each area	X ²	df	P	Confidence interval on proportion of habitat use (<i>p</i>)	Avoided/ Selected/ Proportionate/ Ho/Ha
Active	25% MCP	0.12	0.098	0	1	0.00	1.5			*	NA
Forested	25% MCP	1.09	0.886	15	13	1.00	0.2			*	Proportionate
Open	25% MCP	0.02	0.016	0	0	0.00	0.2			*	NA
Total		1.23	1.000	15	15	1.00	1.9	2	P > 0.25		Ho

Abbreviations: X², Chi-square value; df, degrees of freedom; P, significance level; *, confidence intervals not calculated because the null was not rejected; *p*, represents the theoretical occurrence of locations compared to the proportion of available habitat, in order to determine if the hypothesis of proportional use is accepted or rejected.

Table A-5e. Chi-square goodness-of-fit test, with Bonferroni Z calculated confidence intervals, for habitat selection at the 25% MCP core area spatial scale of 3 habitat classes, for bobcat F4, using the Neu et al. (1974) method.

Habitat Class	Spatial Scale	Area (km ²)	Proportion of available habitat	Observed Locations	Expected Locations	Proportion of observed locations in each area	X ²	df	P	Confidence interval on proportion of habitat use (<i>p</i>)	Avoided/ Selected/ Proportionate/ Ho/Ha
Active	25% MCP	0.09	0.191	1	3	0.07	1.2			*	Proportionate
Forested	25% MCP	0.28	0.596	11	9	0.73	0.5			*	Proportionate
Open	25% MCP	0.1	0.213	3	3	0.20	0.0			*	Proportionate
Total		0.47	1.000	15	15	1.00	1.7	2	P > 0.25		Ho

Abbreviations: X², Chi-square value; df, degrees of freedom; P, significance level; *, confidence intervals not calculated because the null was not rejected; *p*, represents the theoretical occurrence of locations compared to the proportion of available habitat, in order to determine if the hypothesis of proportional use is accepted or rejected.

Table A-5f. Chi-square goodness-of-fit test, with Bonferroni Z calculated confidence intervals, for habitat selection at the 25% MCP core area spatial scale of 3 habitat classes, for bobcat M1, using the Neu et al. (1974) method.

Habitat Class	Spatial Scale	Area (km ²)	Proportion of available habitat	Observed Locations	Expected Locations	Proportion of observed locations in each area	X ²	df	P	Confidence interval on proportion of habitat use (<i>p</i>)	Avoided/ Selected/ Proportionate/ Ho/Ha
Active	25% MCP	0.08	0.127	1	2	0.07	0.4			*	Proportionate
Forested	25% MCP	0.27	0.429	7	6	0.47	0.1			*	Proportionate
Open	25% MCP	0.28	0.444	7	7	0.47	0.0			*	Proportionate
Total		0.63	1.000	15	15	1.00	0.5	2	P > 0.75		Ho

Abbreviations: X², Chi-square value; df, degrees of freedom; P, significance level; *, confidence intervals not calculated because the null was not rejected; *p*, represents the theoretical occurrence of locations compared to the proportion of available habitat, in order to determine if the hypothesis of proportional use is accepted or rejected.



Figure A-1. Active surface mine adjacent to Paul Van Booven wildlife management area, Breathitt, Perry, and Knott counties, Kentucky (Photo: Andrea J. Shipley, April 2011).



Figure A-2. Bobcat captured as part of the study, restrained by using a catch-pole (Photo: Bryan M. Tom, March 2010).

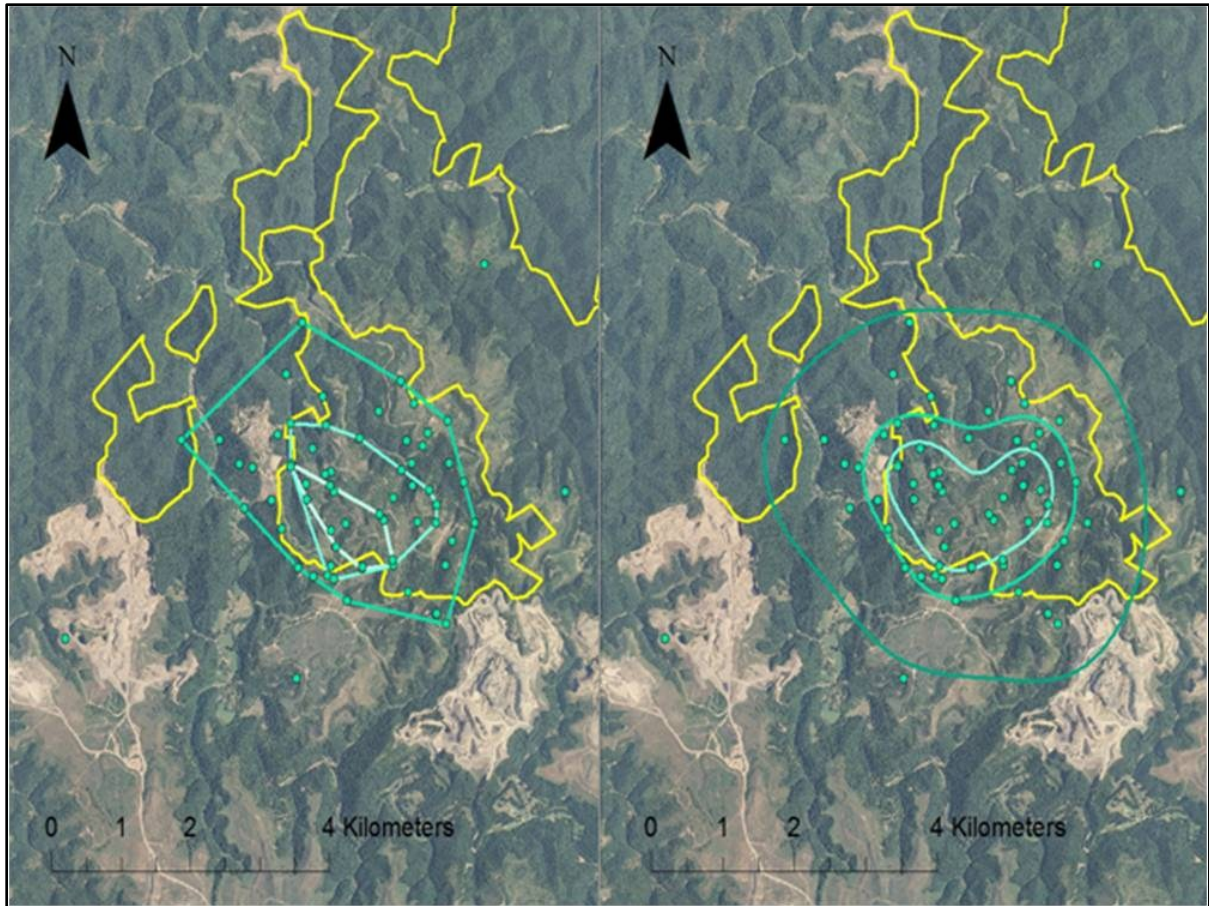


Figure A-3. The 95%, 50%, and 25% minimum convex polygons (at left) and adaptive kernels (at right) for F1.

Each scale is in a progressively lighter shade of blue, with the associated locations, the wildlife management area and Robinson Forest boundaries are represented in yellow.

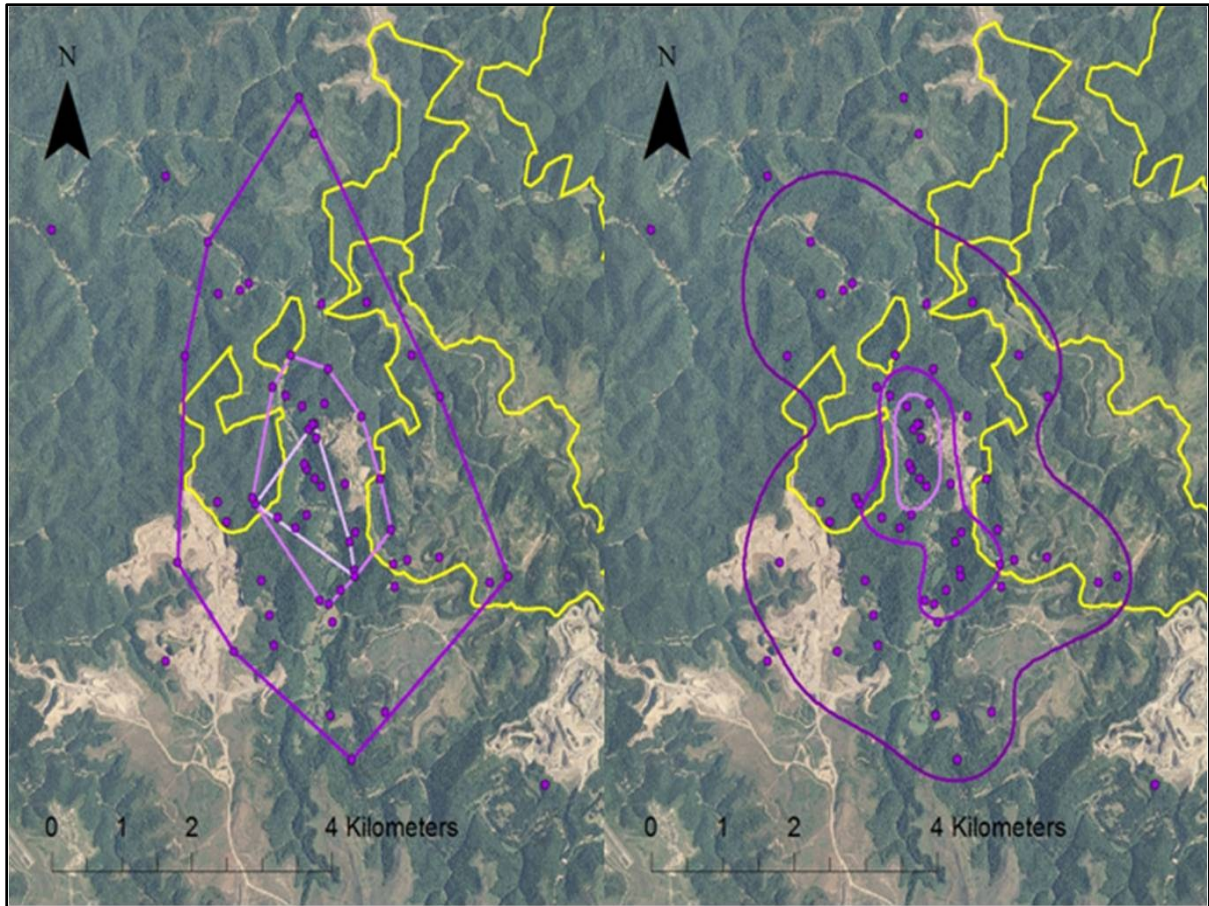


Figure A-4. The 95%, 50%, and 25% minimum convex polygons (at left) and adaptive kernels (at right) for F2.

Each scale is in a progressively lighter shade of purple, with the associated locations, the wildlife management area and Robinson Forest boundaries are represented in yellow.

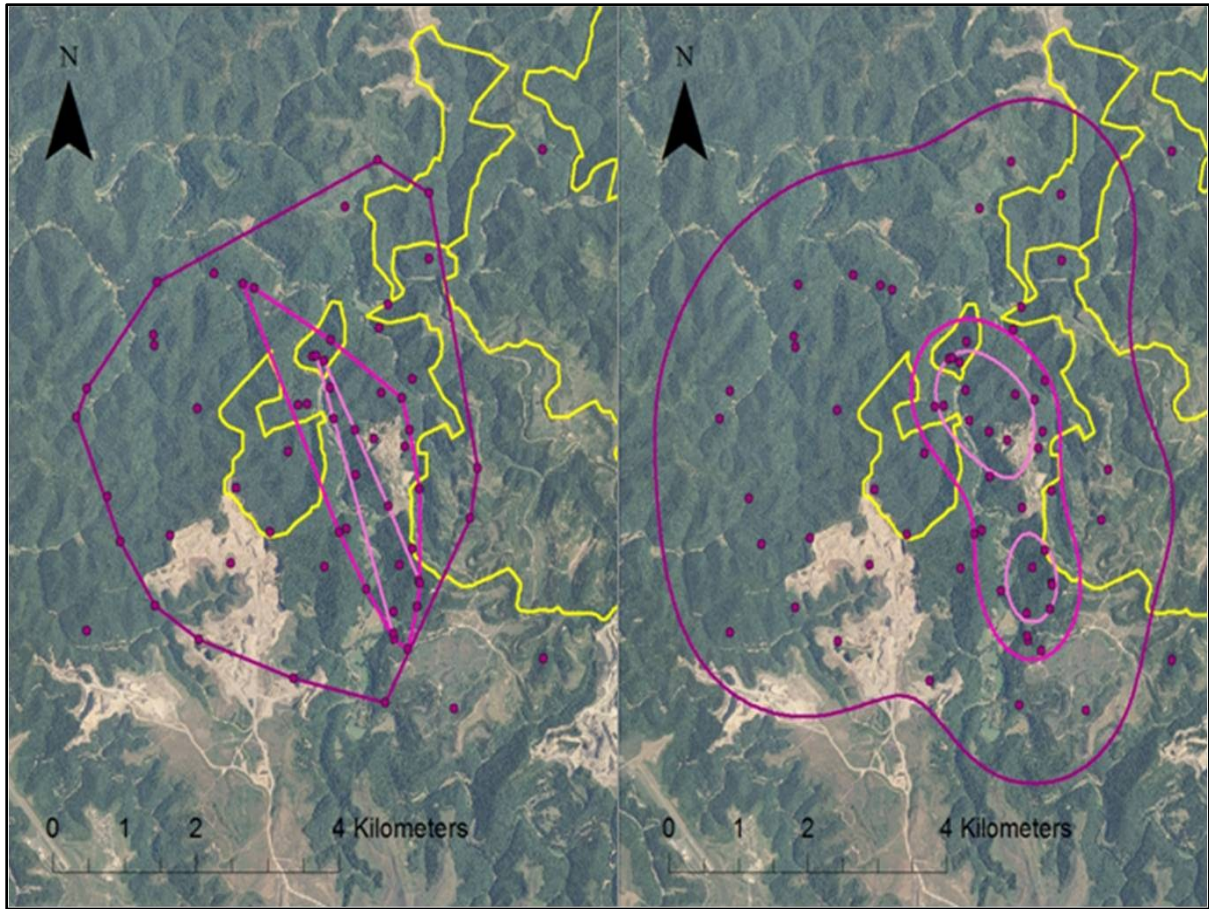


Figure A-5. The 95%, 50%, and 25% minimum convex polygons (at left) and adaptive kernels (at right) for F3.

Each scale is in a progressively lighter shade of pink, with the associated locations, the wildlife management area and Robinson Forest boundaries are represented in yellow.

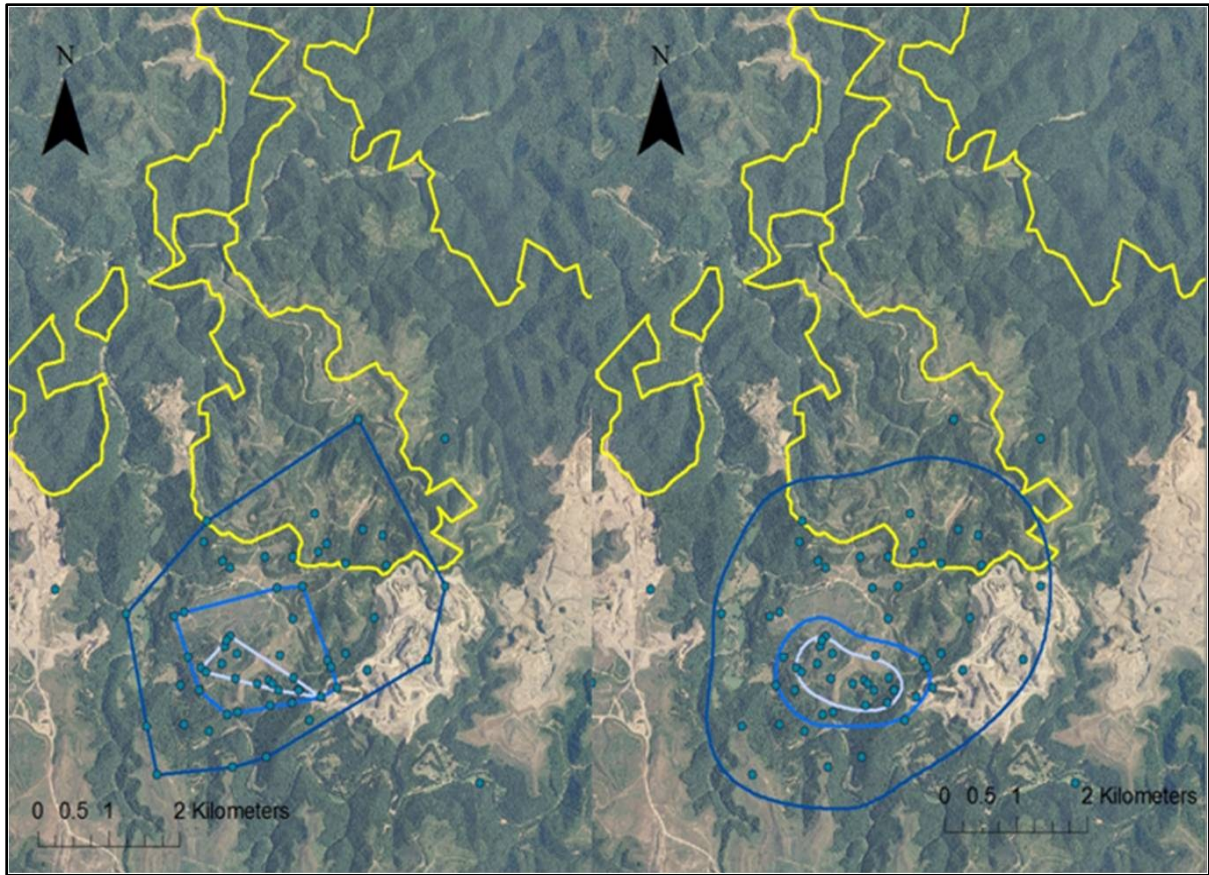


Figure A-6. The 95%, 50%, and 25% minimum convex polygons (at left) and adaptive kernels (at right) for F4.

Each scale is in a progressively lighter shade of dark blue, with the associated locations, the wildlife management area and Robinson Forest boundaries are represented in yellow.

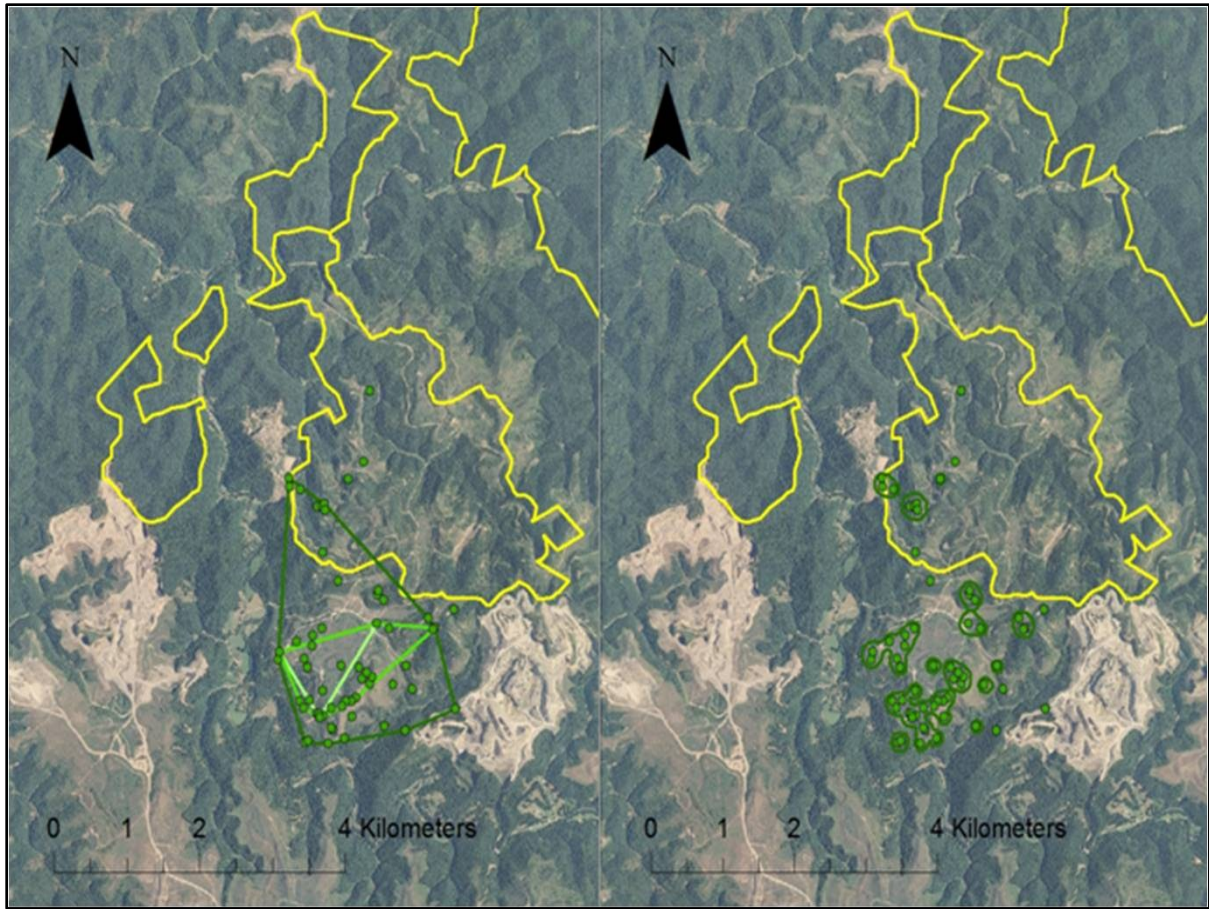


Figure A-7. The 95%, 50%, and 25% minimum convex polygons (at left) and adaptive kernels (at right) for M1.

Each scale is in a progressively lighter shade of green, with the associated locations, the wildlife management area and Robinson Forest boundaries are represented in yellow.

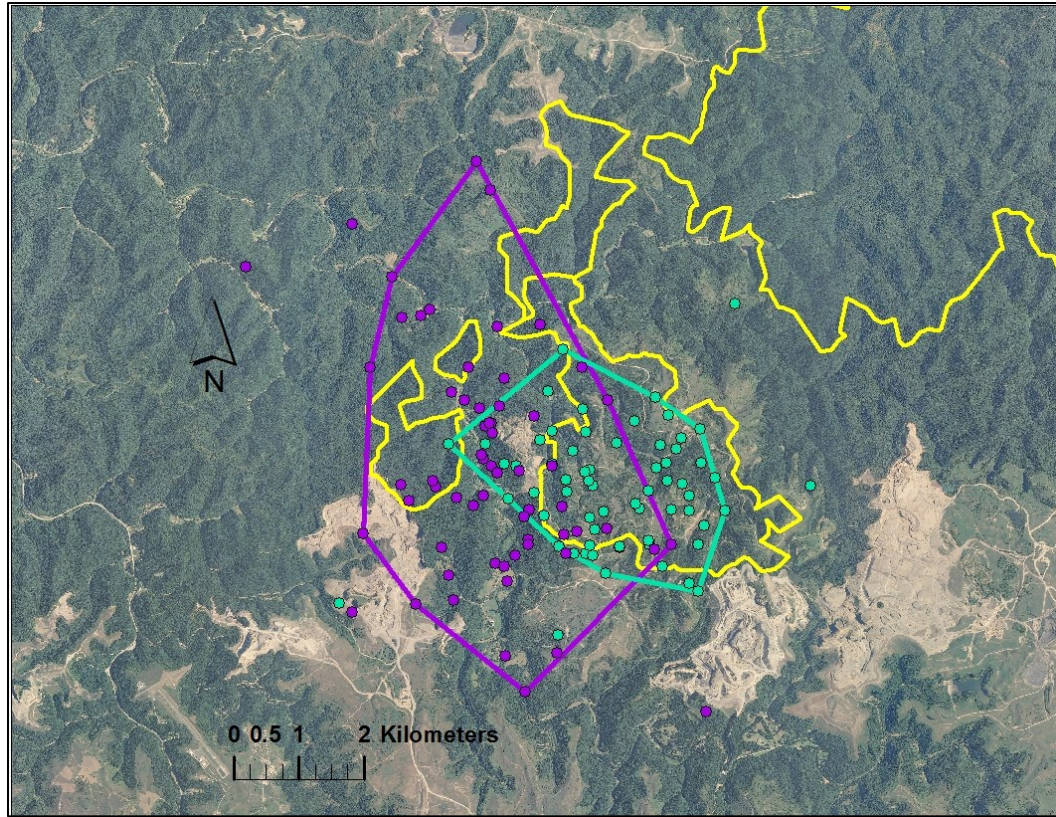


Figure A-8. The 95% minimum convex polygon home range overlap for F1 and F2 with the associated locations.

F1 in teal and F2 in purple; the wildlife management area and Robinson Forest boundaries are represented in yellow.

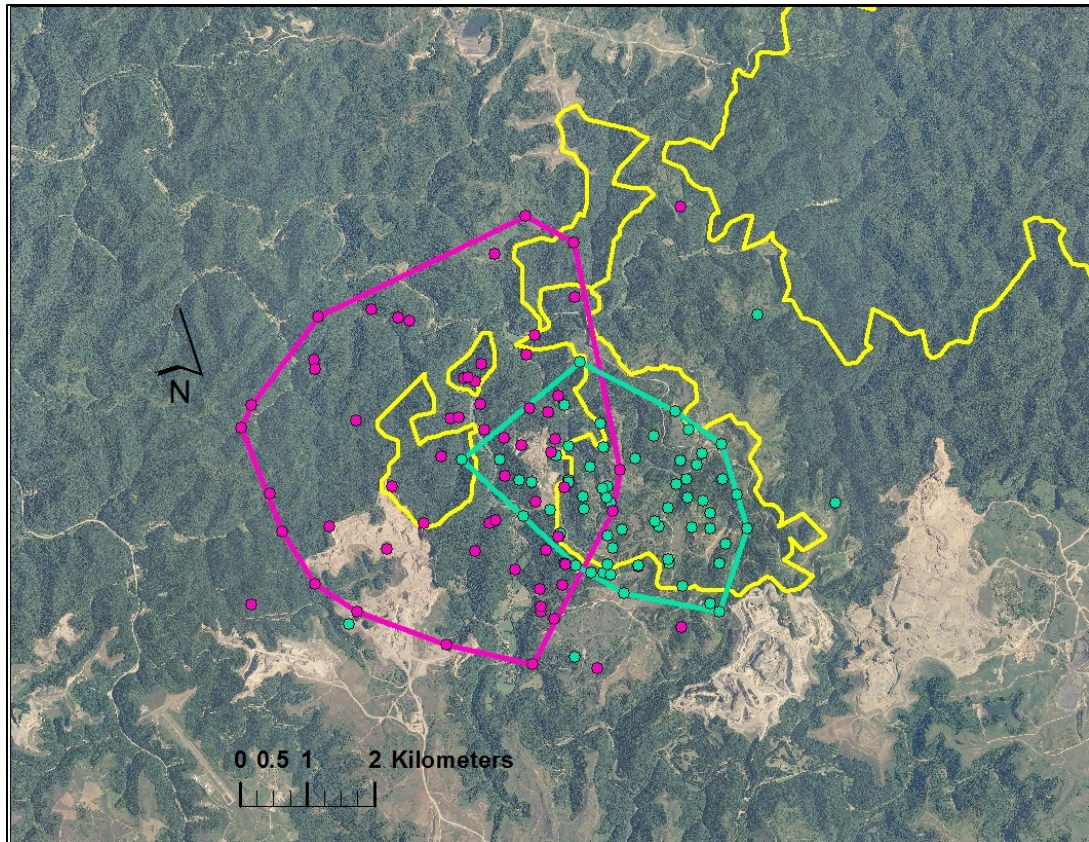


Figure A-9. The 95% minimum convex polygon home range overlap for F1 and F3 with the associated locations.

F1 in teal and F3 in dark pink; the wildlife management area and Robinson Forest boundaries are represented in yellow.

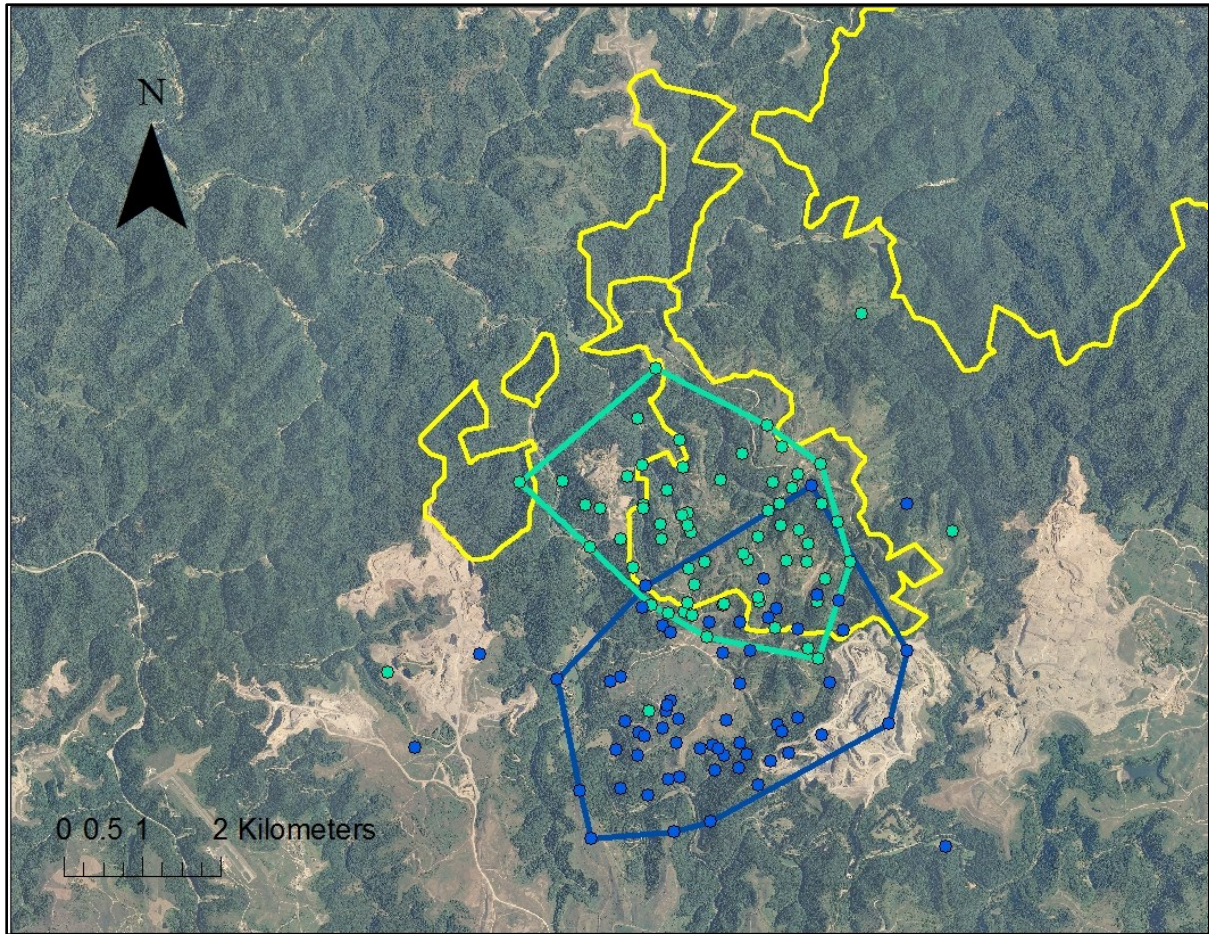


Figure A-10. The 95% minimum convex polygon home range overlap for F1 and F4 with the associated locations.

F1 in teal and F4 in dark blue; the wildlife management area and Robinson Forest boundaries are represented in yellow.

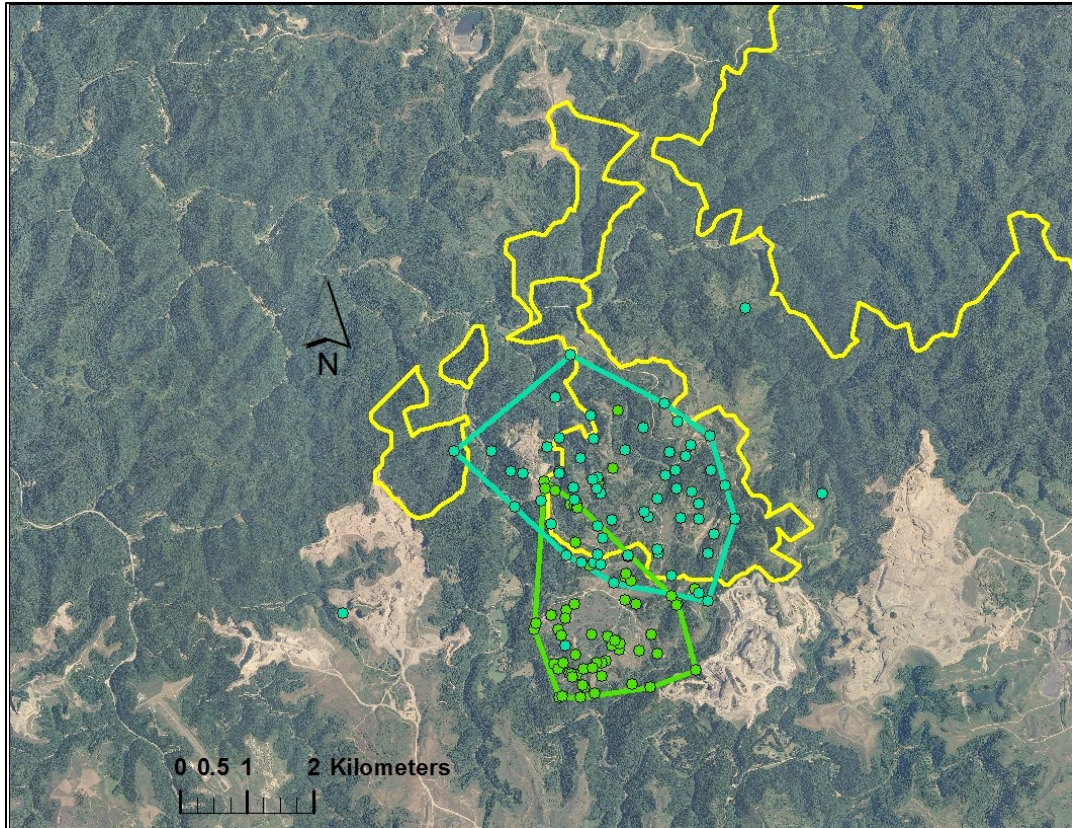


Figure A-11. The 95% minimum convex polygon home range overlap for F1 and M1 with the associated locations.

F1 in teal and M1 in bright green; the wildlife management area and Robinson Forest boundaries are represented in yellow.

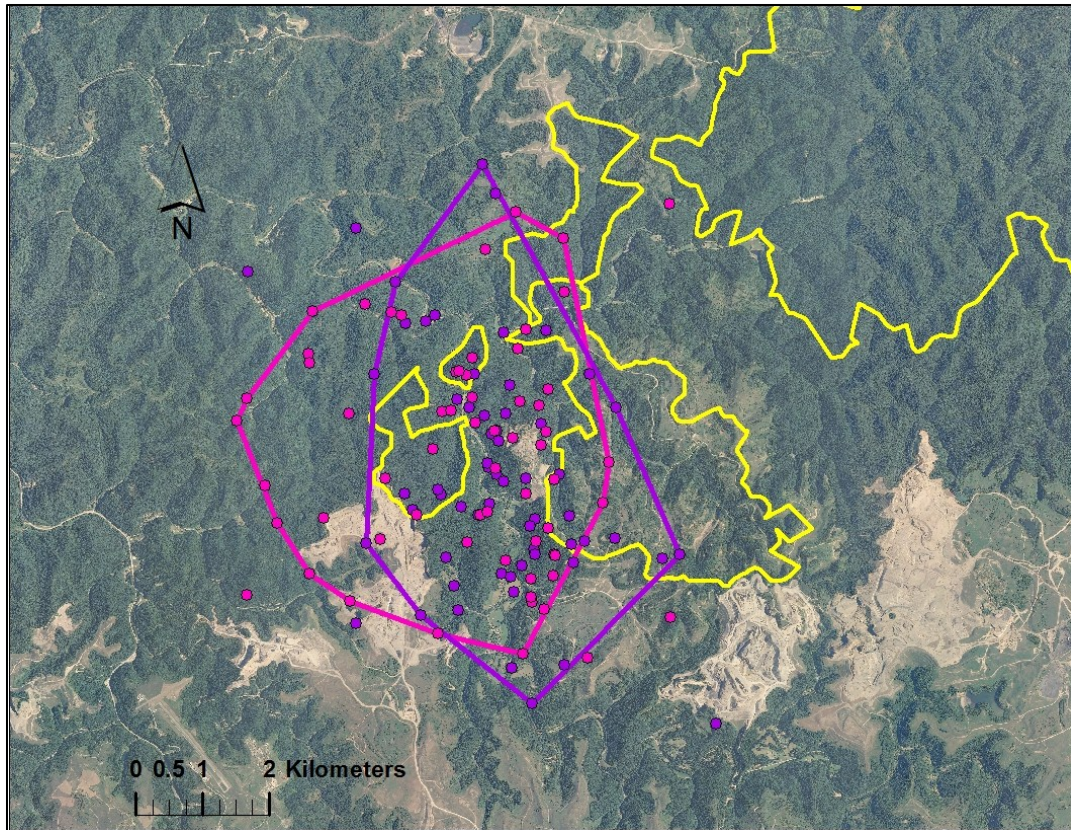


Figure A-12. The 95% minimum convex polygon home range overlap for F2 and F3 with the associated locations.

F2 in purple and F3 in dark pink; the wildlife management area and Robinson Forest boundaries are represented in yellow.

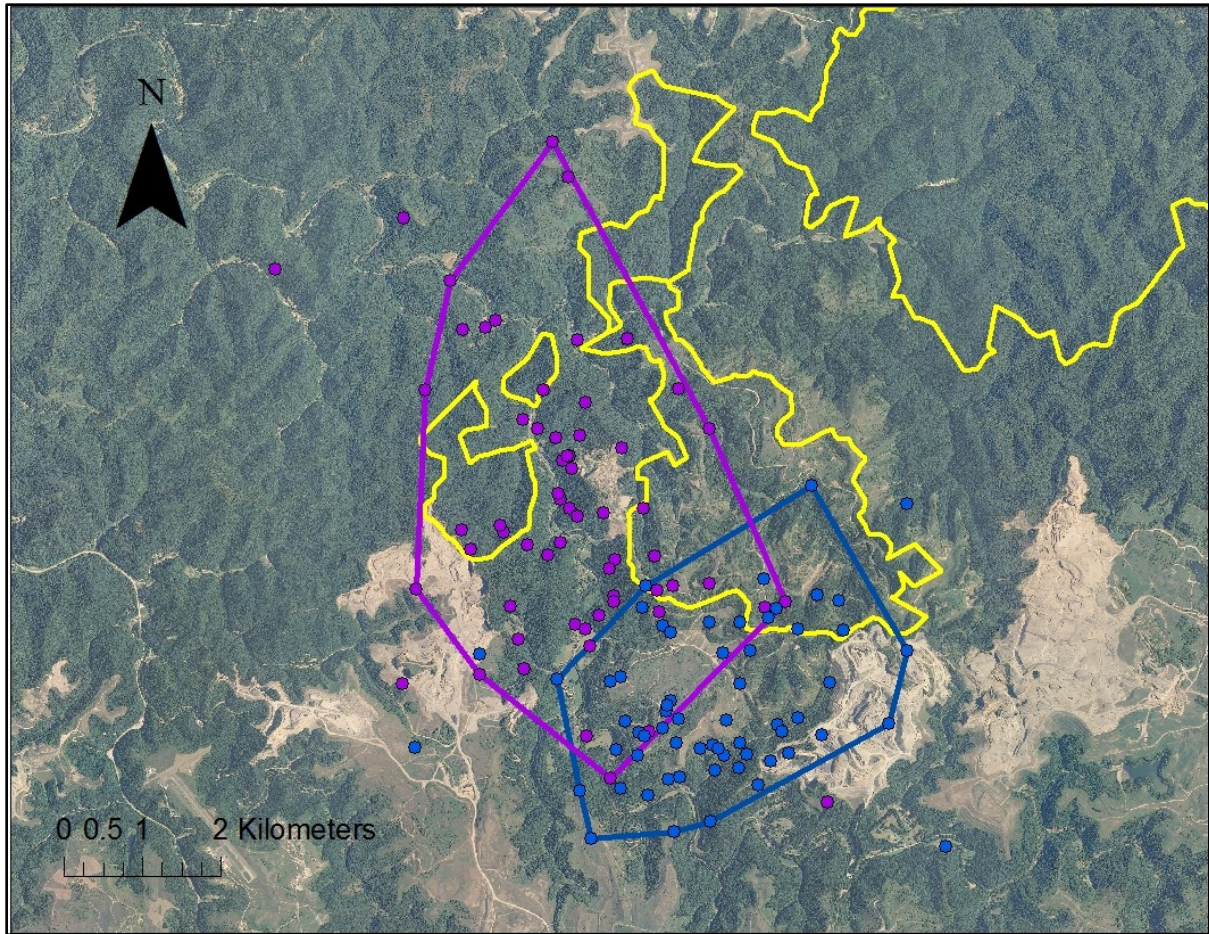


Figure A-13. The 95% minimum convex polygon home range overlap for F2 and F4 with the associated locations.

F2 in purple and F3 in dark blue; the wildlife management area and Robinson Forest boundaries are represented in yellow.

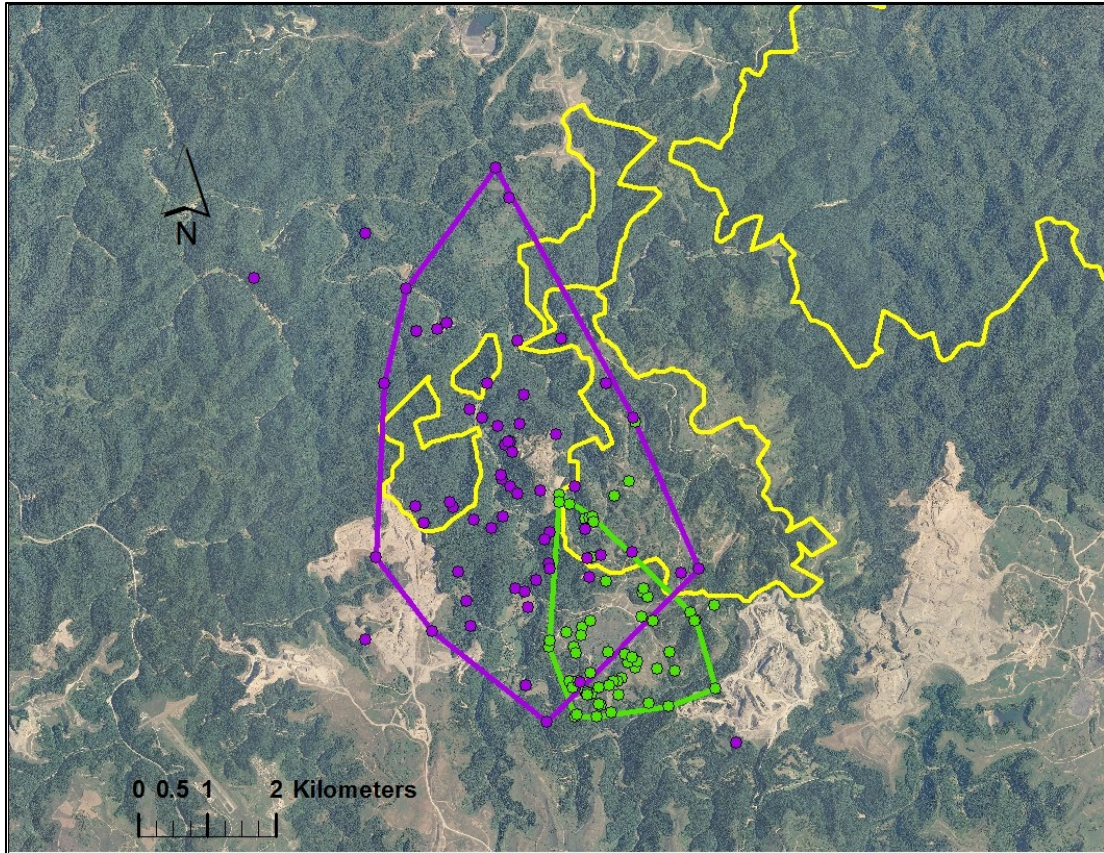


Figure A-14. The 95% minimum convex polygon home range overlap for F2 and M1 with the associated locations.

F2 in purple and M1 in bright green; the wildlife management area and Robinson Forest boundaries are represented in yellow.

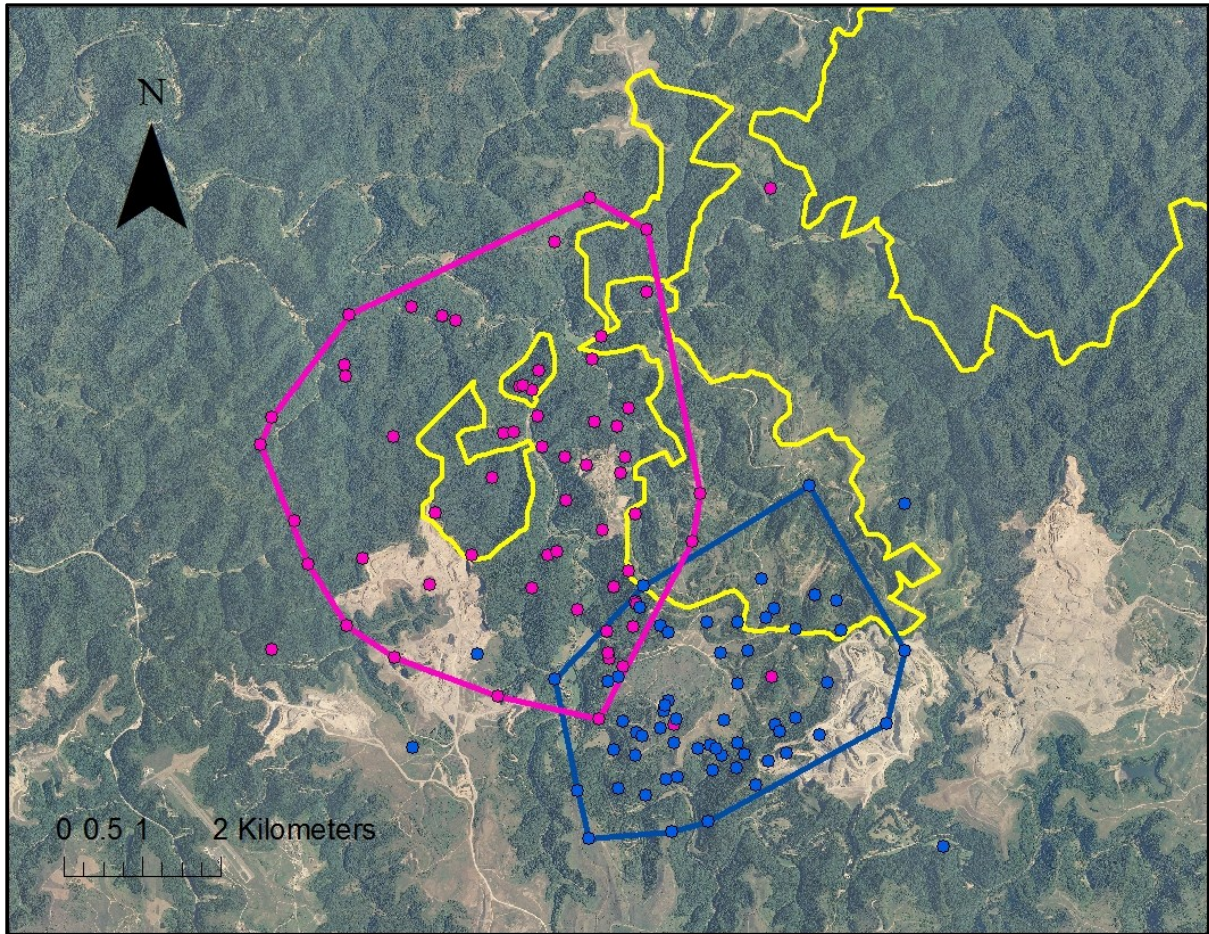


Figure A-15. The 95% minimum convex polygon home range overlap for F3 and F4 with the associated locations.

F3 in dark pink and F4 in dark blue; the wildlife management area and Robinson Forest boundaries are represented in yellow.

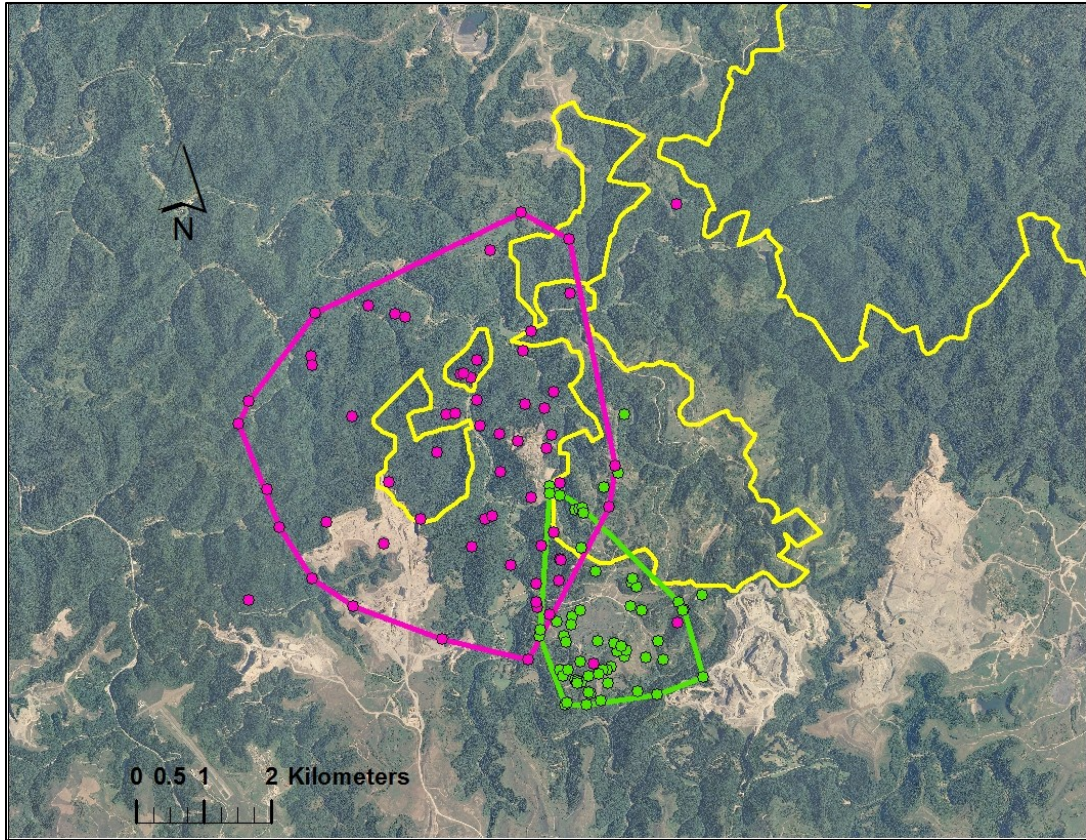


Figure A-16. The 95% minimum convex polygon home range overlap for F3 and M1 with the associated locations.

F3 in dark pink and M1 in bright green; the wildlife management area and Robinson Forest boundaries are represented in yellow.

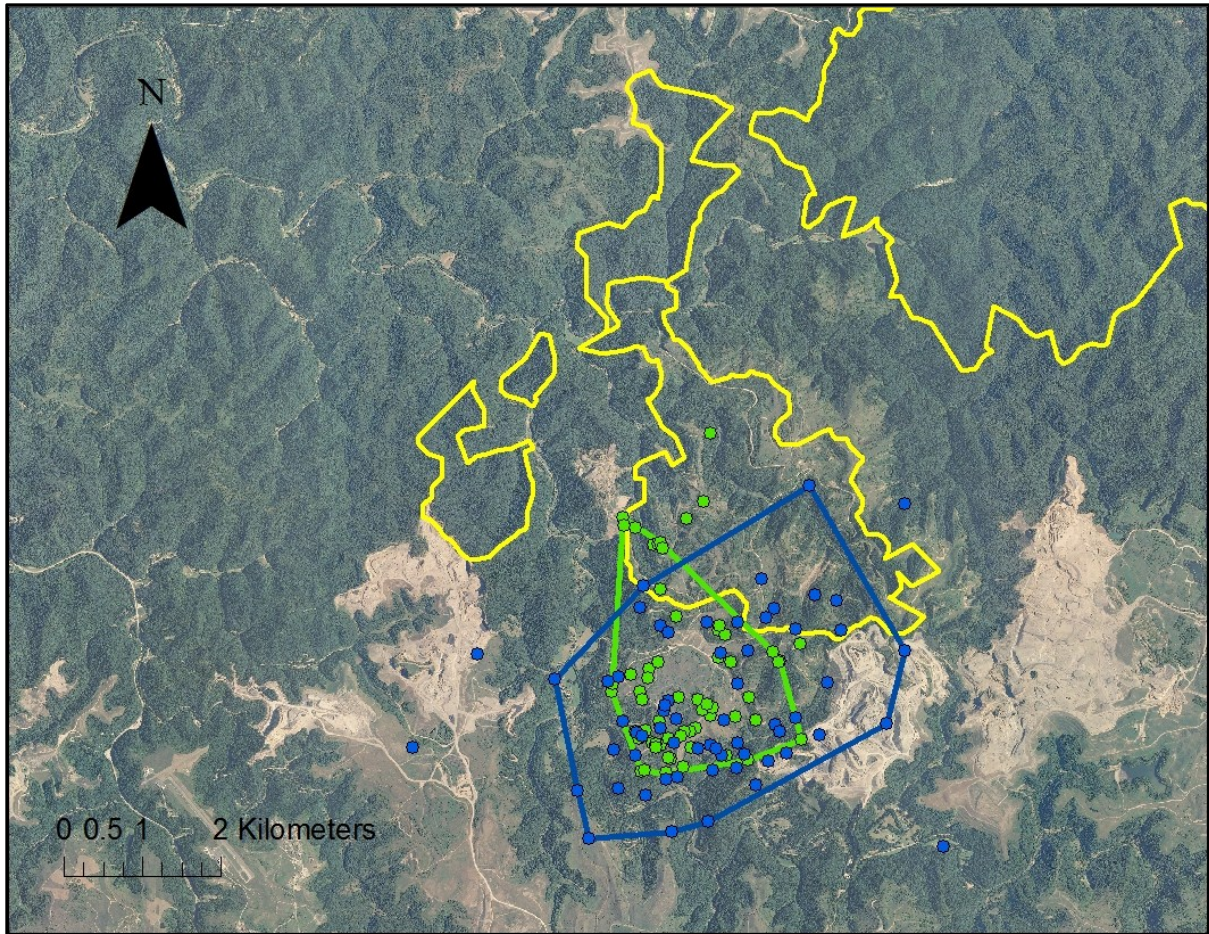


Figure A-17. The 95% minimum convex polygon home range overlap for F4 and M1 with the associated locations.

F4 in dark blue and M1 in bright green; the wildlife management area and Robinson Forest boundaries are represented in yellow.

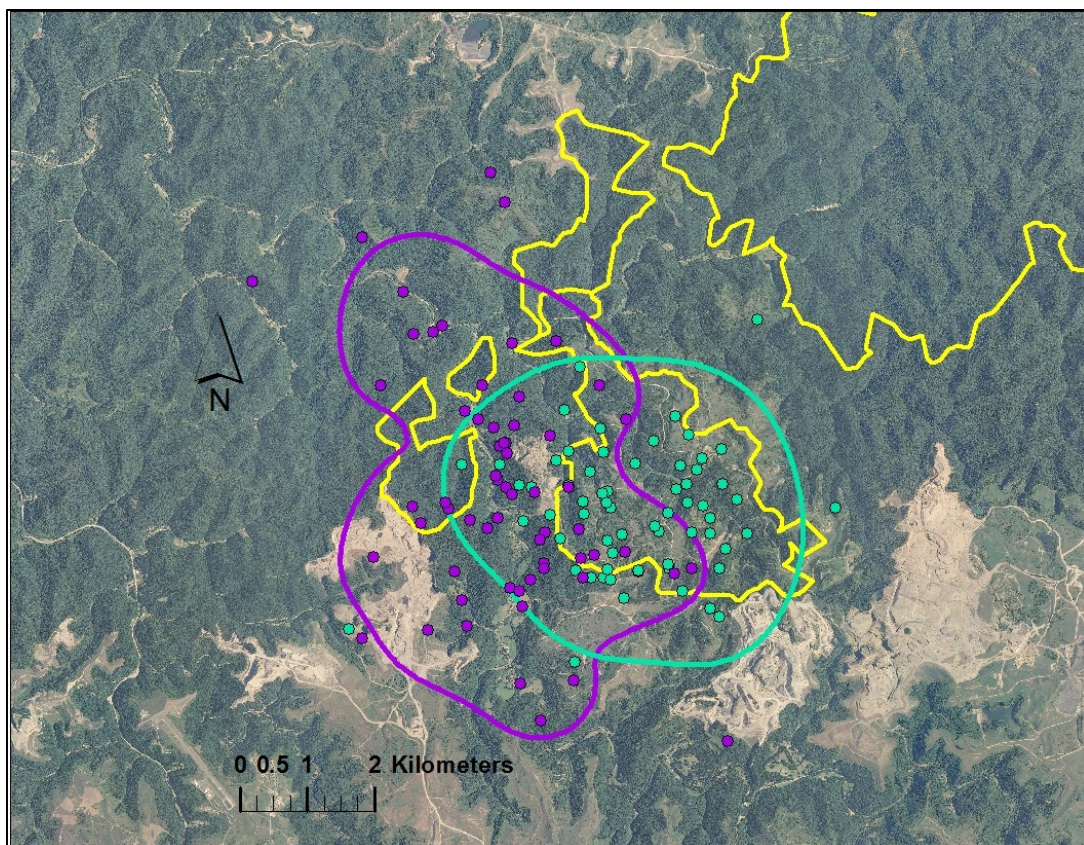


Figure A-18. The 95% adaptive kernel home range overlap for F1 and F2 with the associated locations.

F1 in teal and F2 in purple; the wildlife management area and Robinson Forest boundaries are represented in yellow.

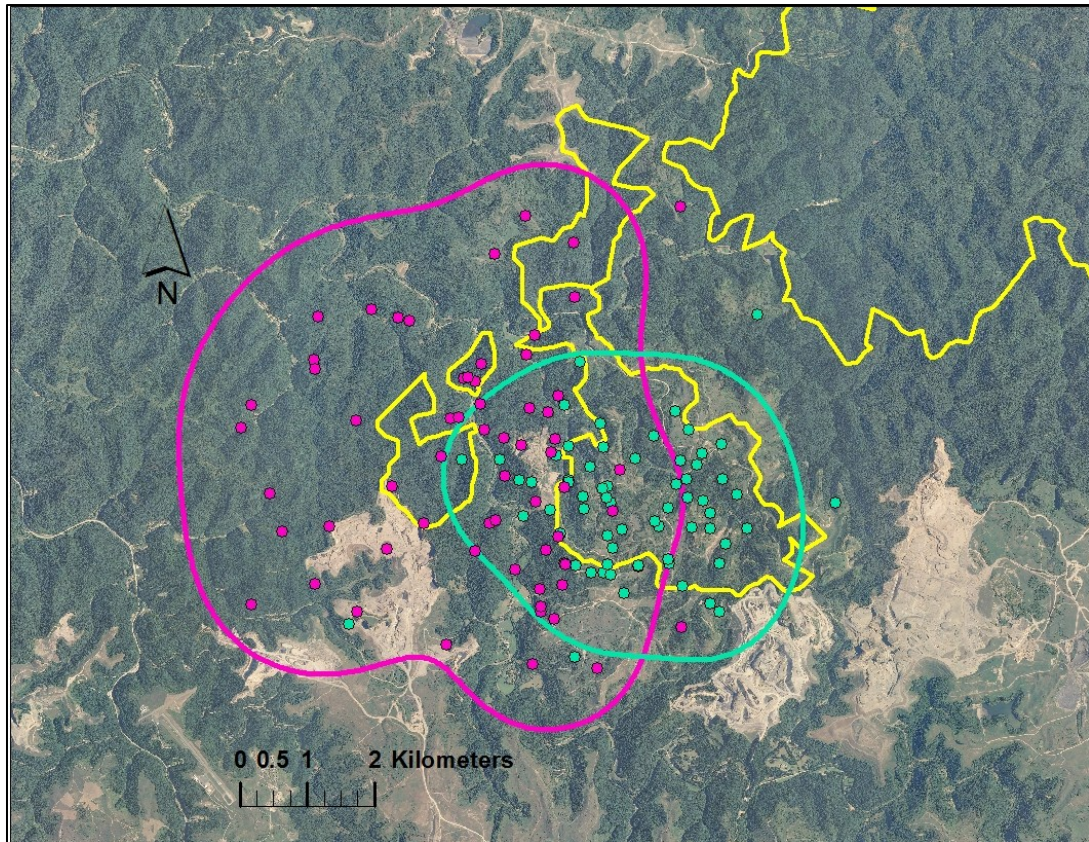


Figure A-19. The 95% adaptive kernel home range overlap for F1 and F3 with the associated locations.

F1 in teal and F3 in dark pink; the wildlife management area and Robinson Forest boundaries are represented in yellow.

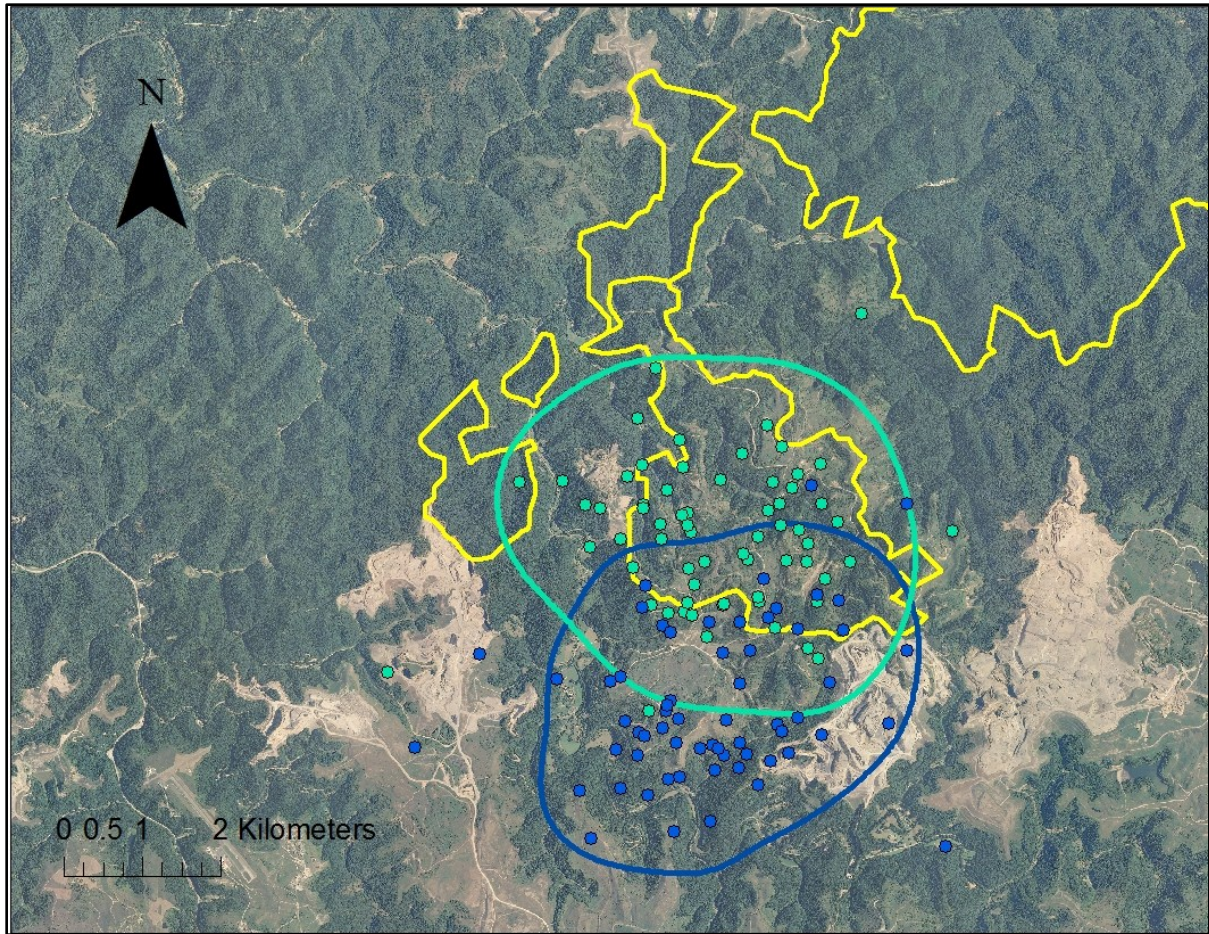


Figure A-20. The 95% adaptive kernel home range overlap for F1 and F4 with the associated locations.

F1 in teal and F4 in dark blue; the wildlife management area and Robinson Forest boundaries are represented in yellow.

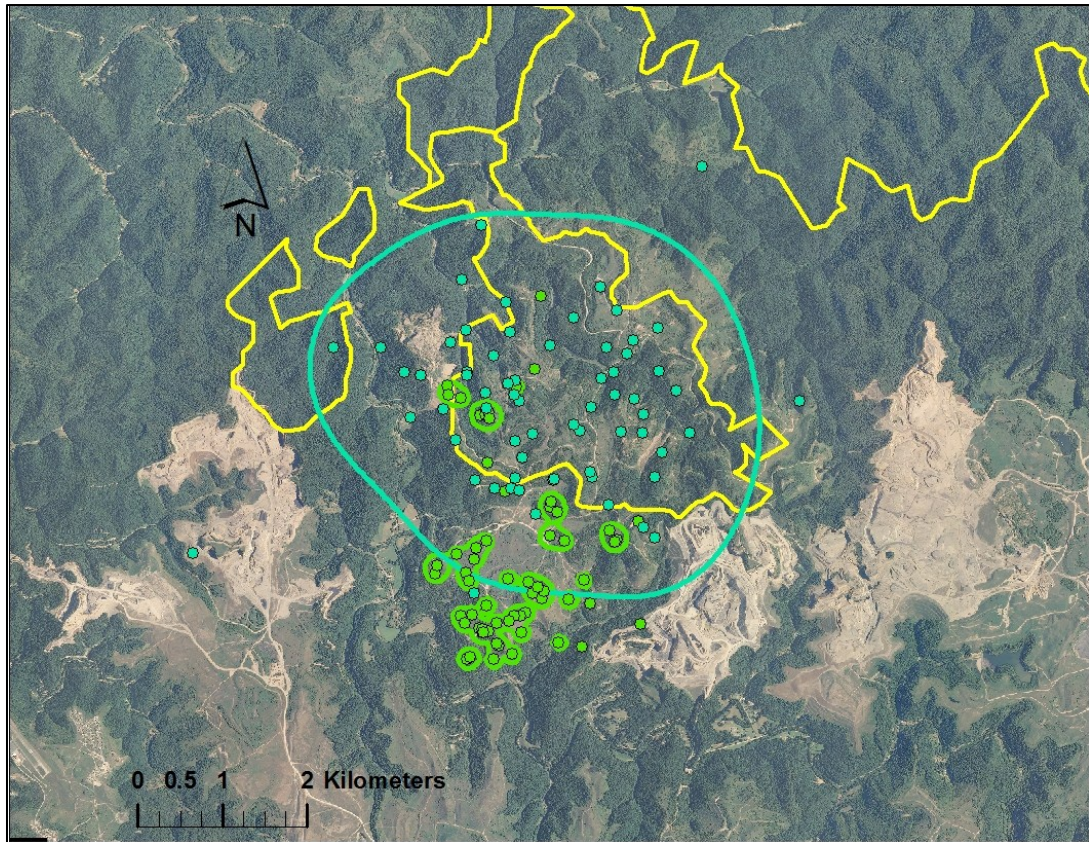


Figure A-21. The 95% adaptive kernel home range overlap for F1 and M1 with the associated locations.

F1 in teal and M1 in bright green; the wildlife management area and Robinson Forest boundaries are represented in yellow.

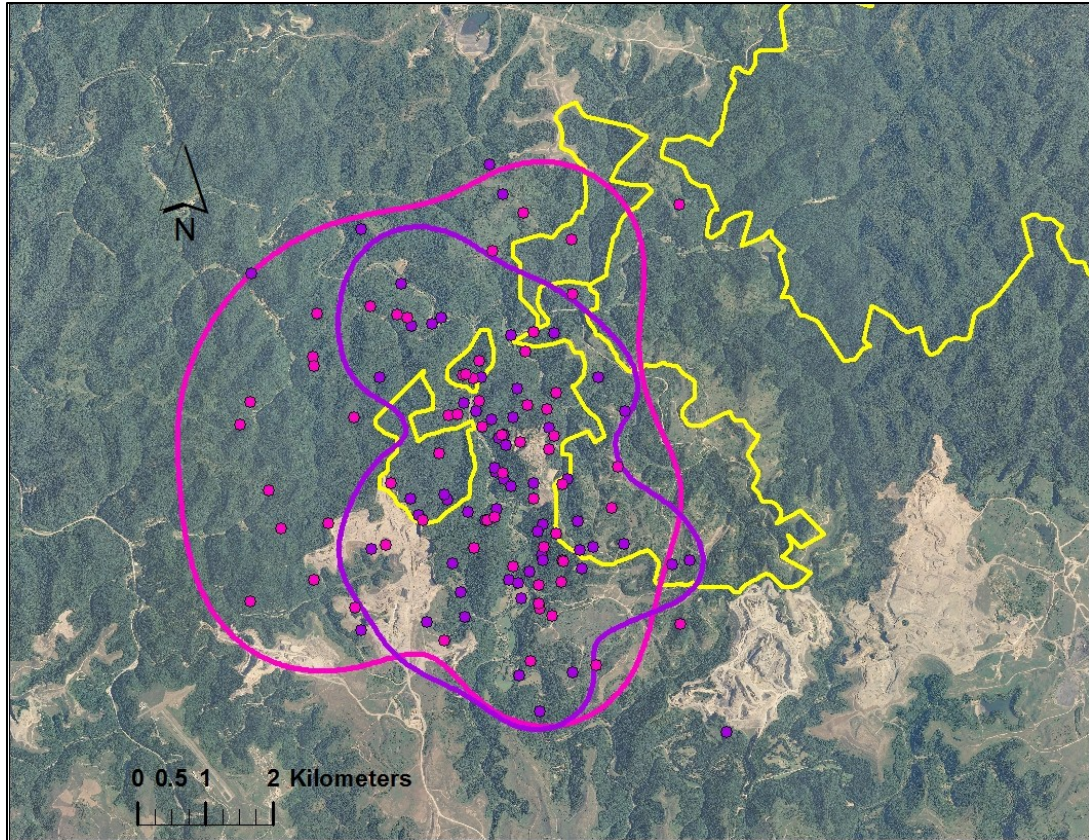


Figure A-22. The 95% adaptive kernel home range overlap for F2 and F3 with the associated locations.

F2 in purple and F3 in dark pink; the wildlife management area and Robinson Forest boundaries are represented in yellow.

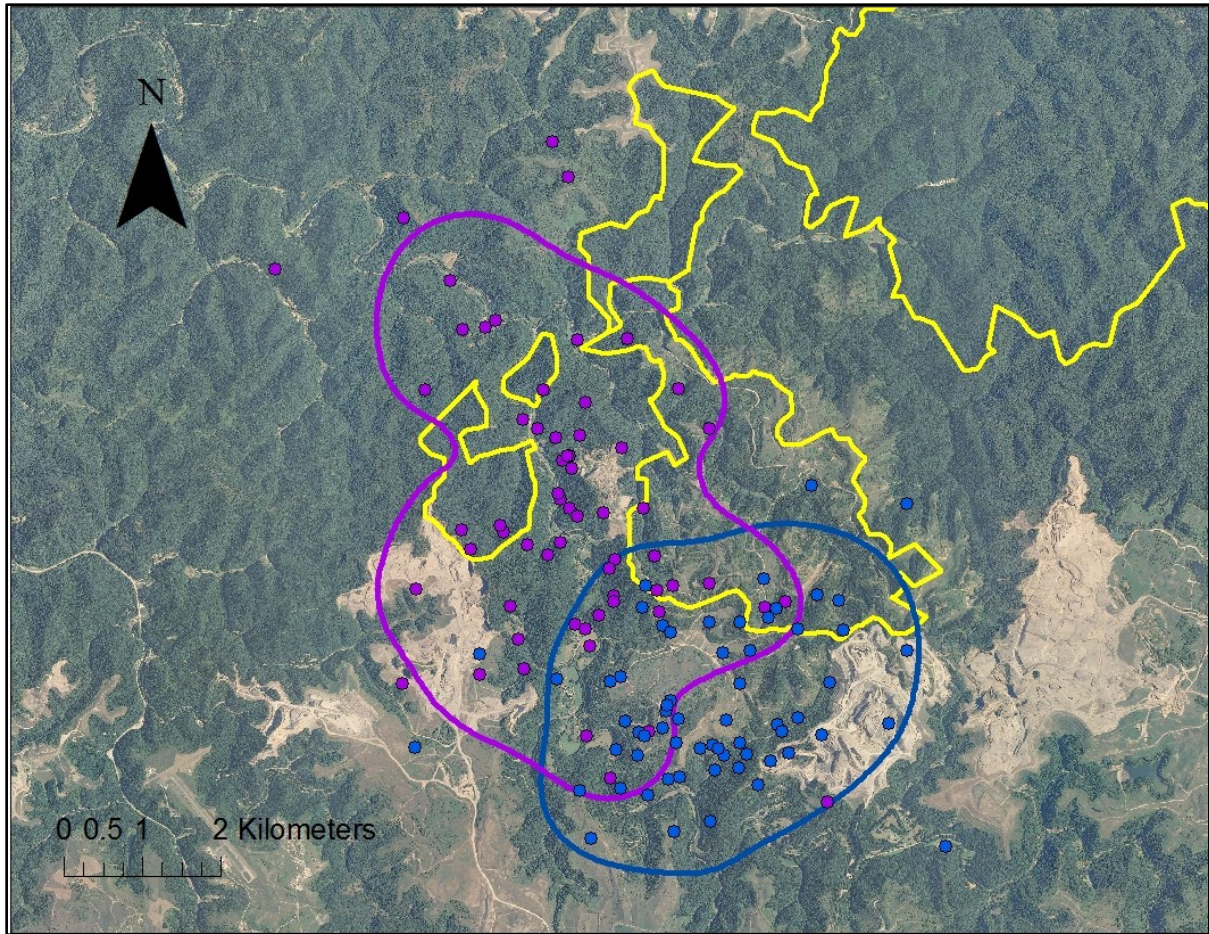


Figure A-23. The 95% adaptive kernel home range overlap for F2 and F4 with the associated locations.

F2 in purple and F4 in dark blue; the wildlife management area and Robinson Forest boundaries are represented in yellow.

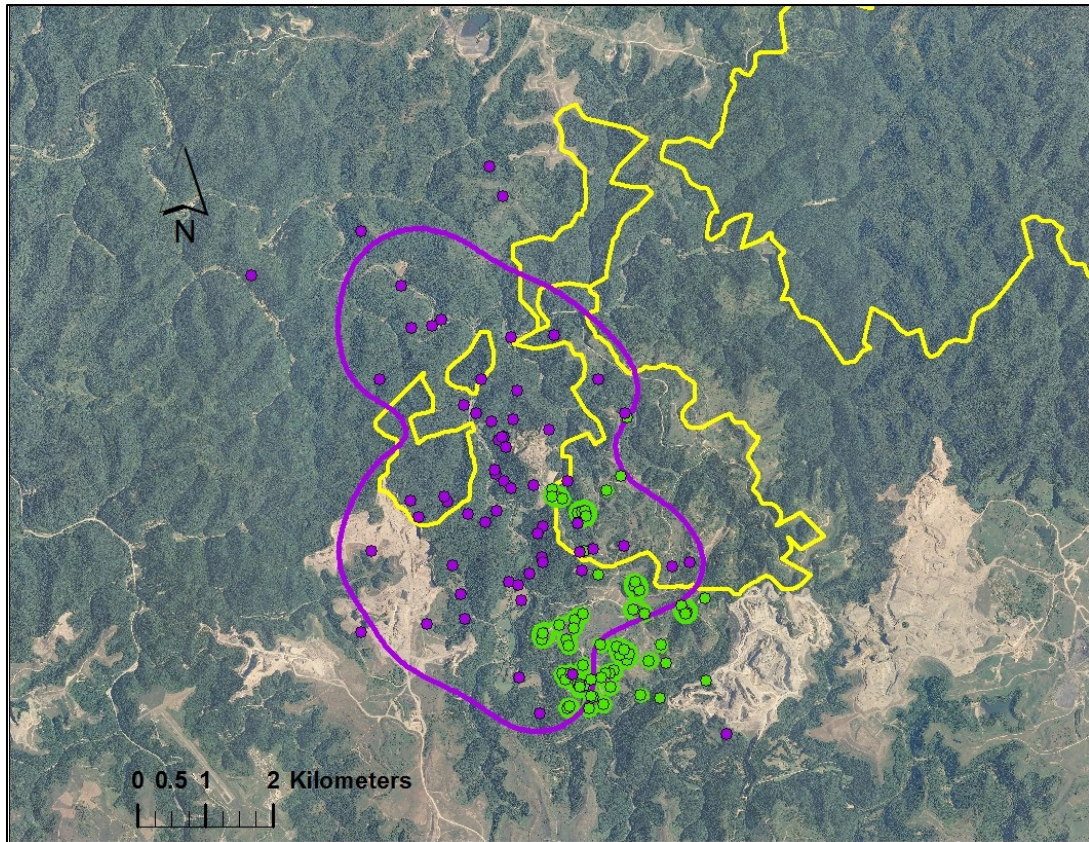


Figure A-24. The 95% adaptive kernel home range overlap for F2 and M1 with the associated locations.

F2 in purple and M1 in bright green; the wildlife management area and Robinson Forest boundaries are represented in yellow.

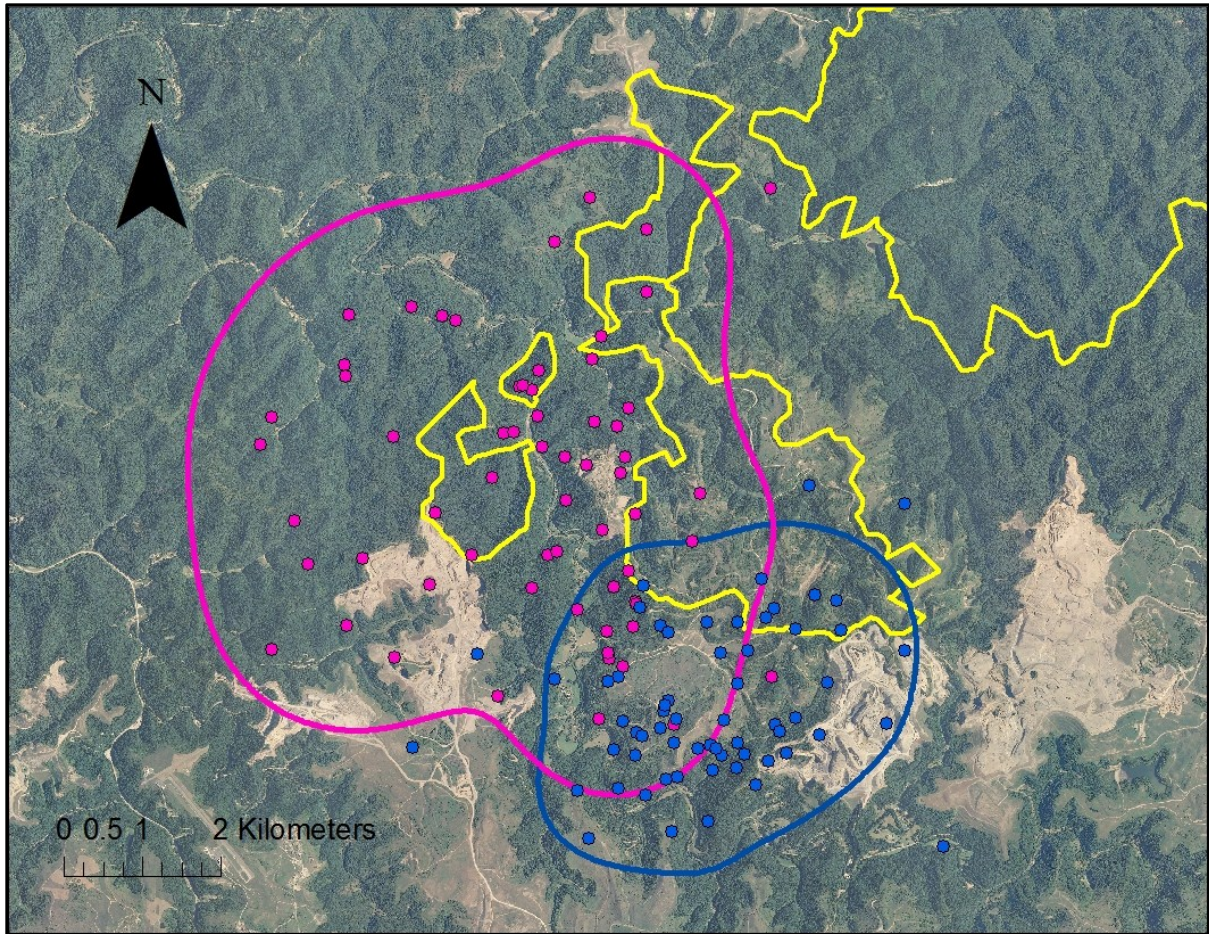


Figure A-25. The 95% adaptive kernel home range overlap for F3 and F4 with the associated locations.

F3 in dark pink and F4 in dark blue; the wildlife management area and Robinson Forest boundaries are represented in yellow.

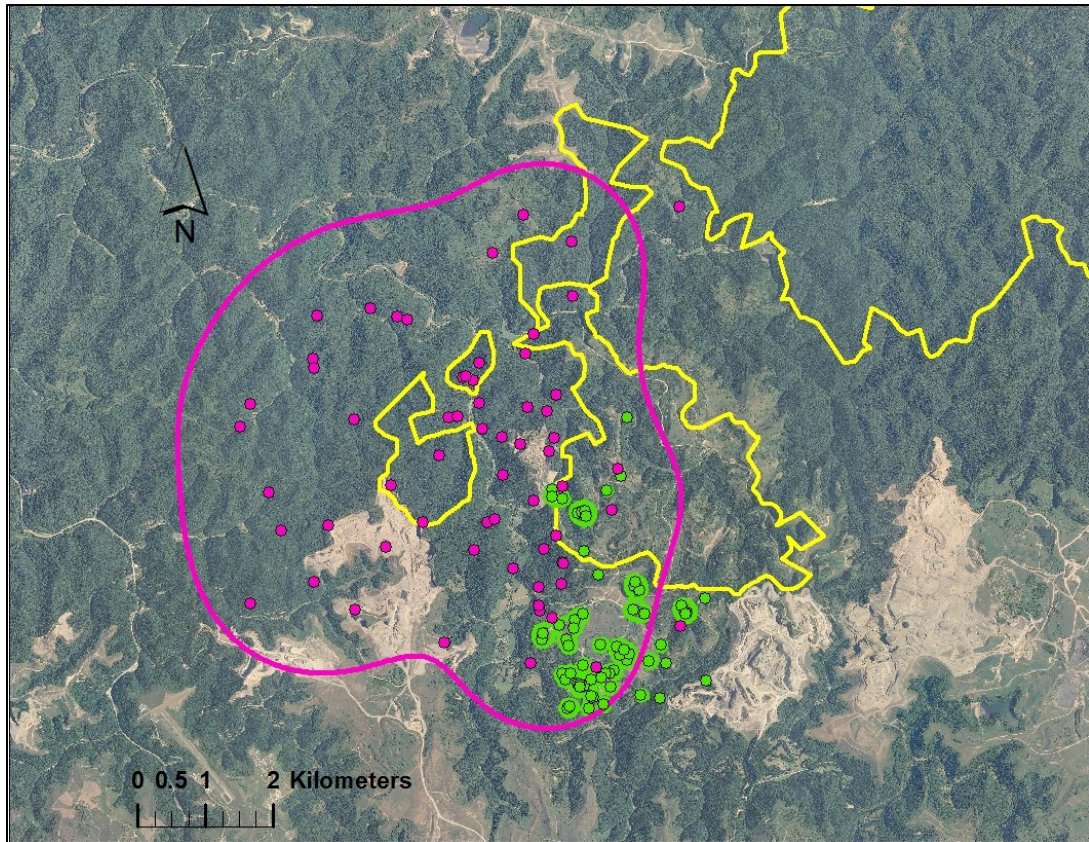


Figure A-26. The 95% adaptive kernel home range overlap for F3 and M1 with the associated locations.

F3 in dark pink and M1 in bright green; the wildlife management area and Robinson Forest boundaries are represented in yellow.

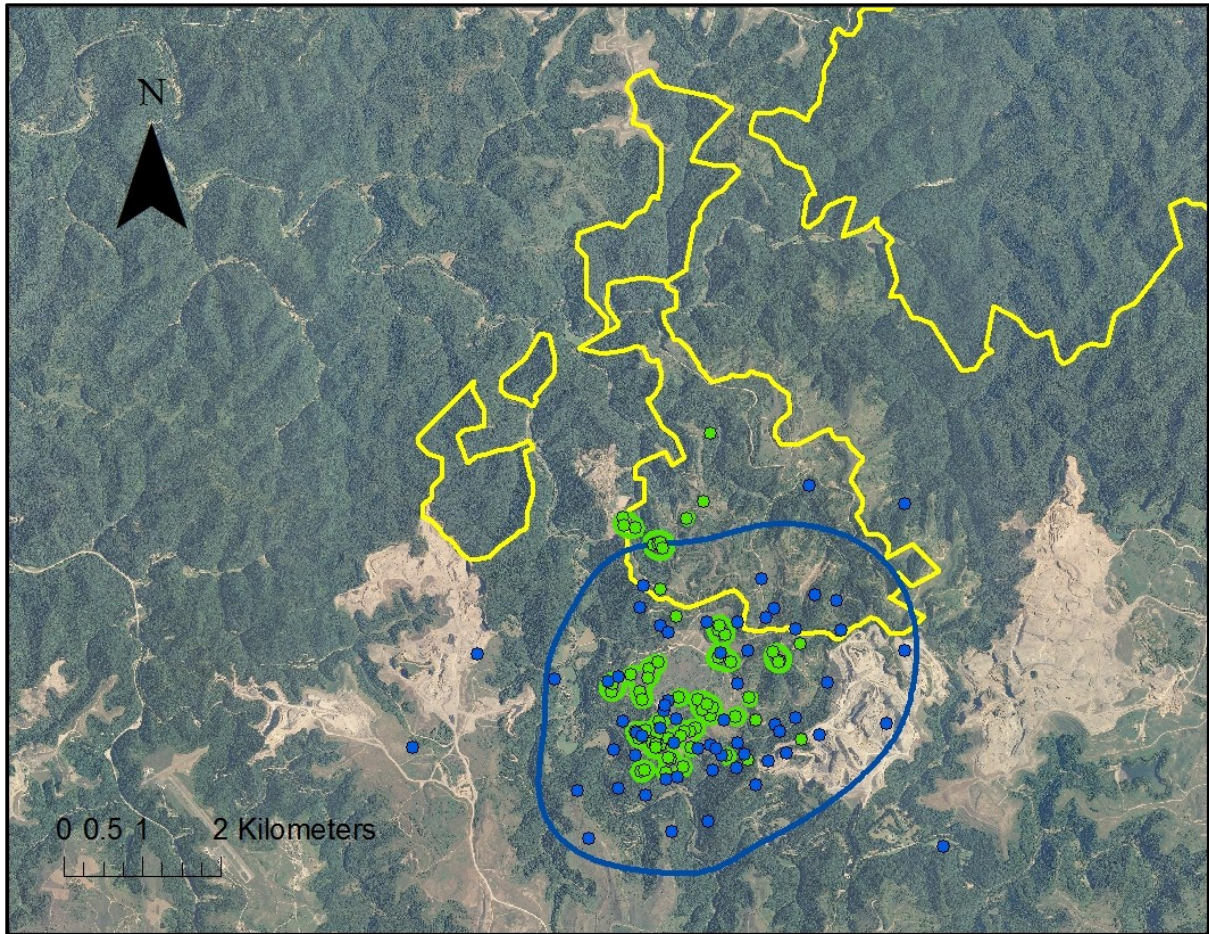


Figure A-27. The 95% adaptive kernel home range overlap for F4 and M1 with the associated locations.

F4 in dark blue and M1 in bright green; the wildlife management area and Robinson Forest boundaries are represented in yellow.

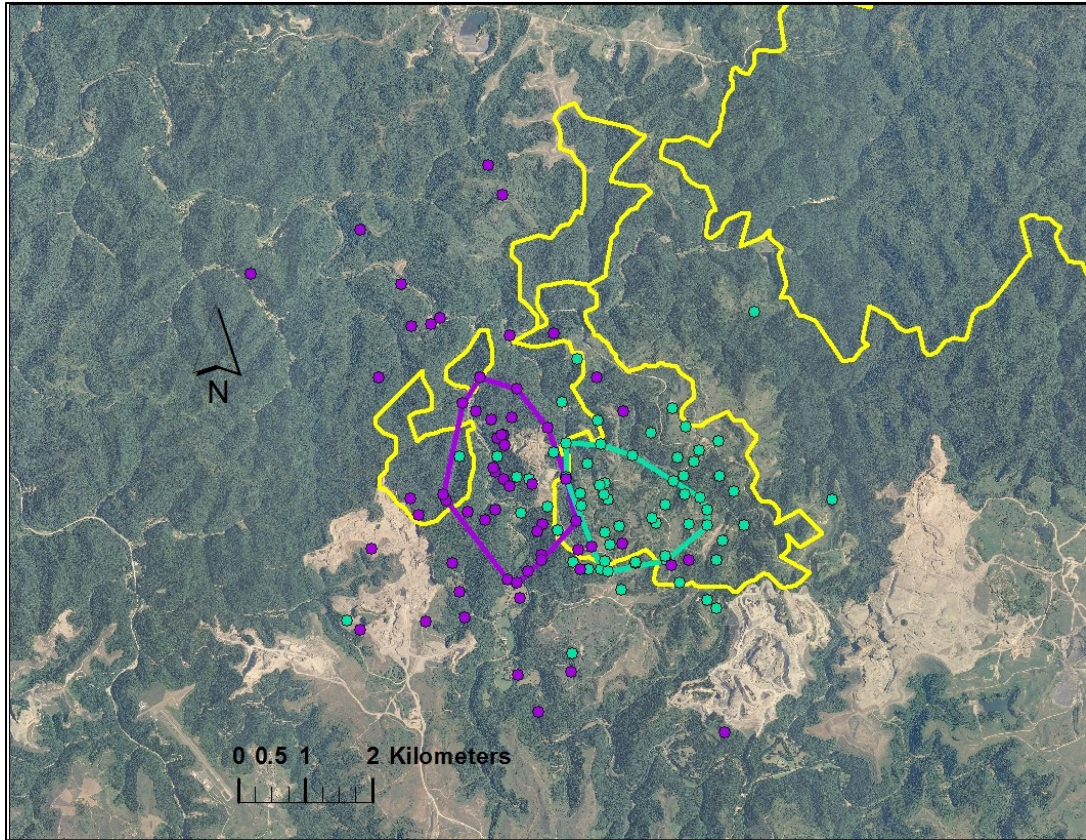


Figure A-28. The 50% minimum convex polygon core area overlap for F1 and F2 with the associated locations.

F1 in teal and F2 in purple; the wildlife management area and Robinson Forest boundaries are represented in yellow.

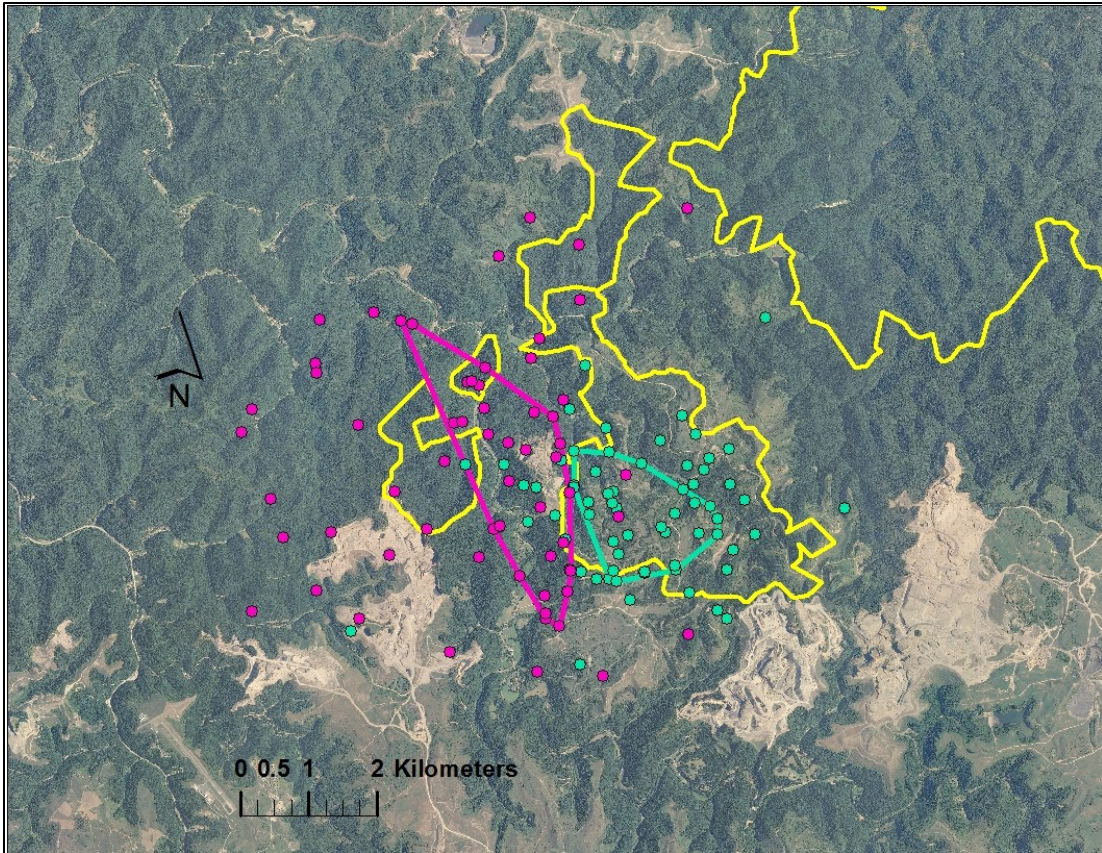


Figure A-29. The 50% minimum convex polygon core area overlap for F1 and F3 with the associated locations.

F1 in teal and F3 in dark pink; the wildlife management area and Robinson Forest boundaries are represented in yellow.

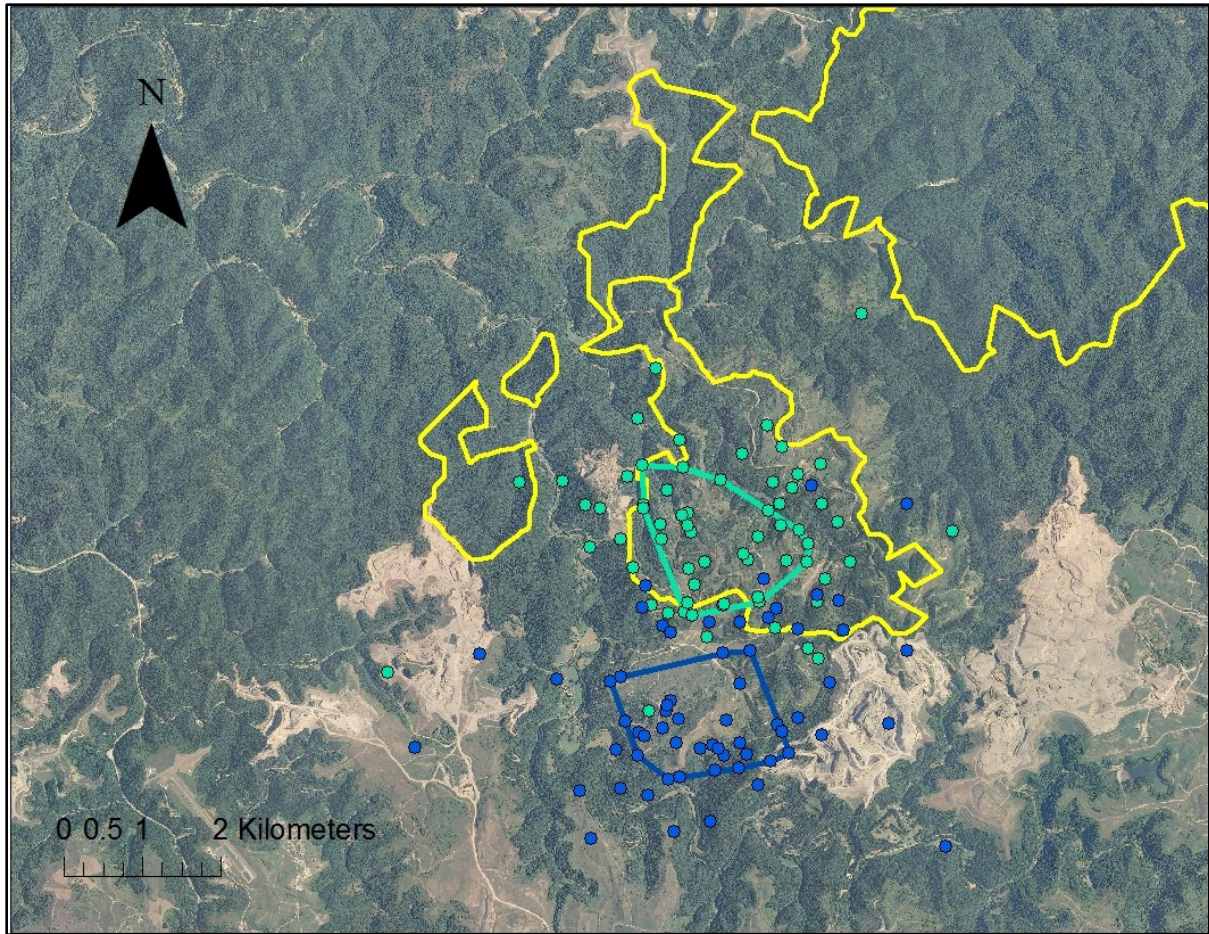


Figure A-30. The 50% minimum convex polygon core area overlap for F1 and F4 with the associated locations.

F1 in teal and F4 in dark blue; the wildlife management area and Robinson Forest boundaries are represented in yellow.

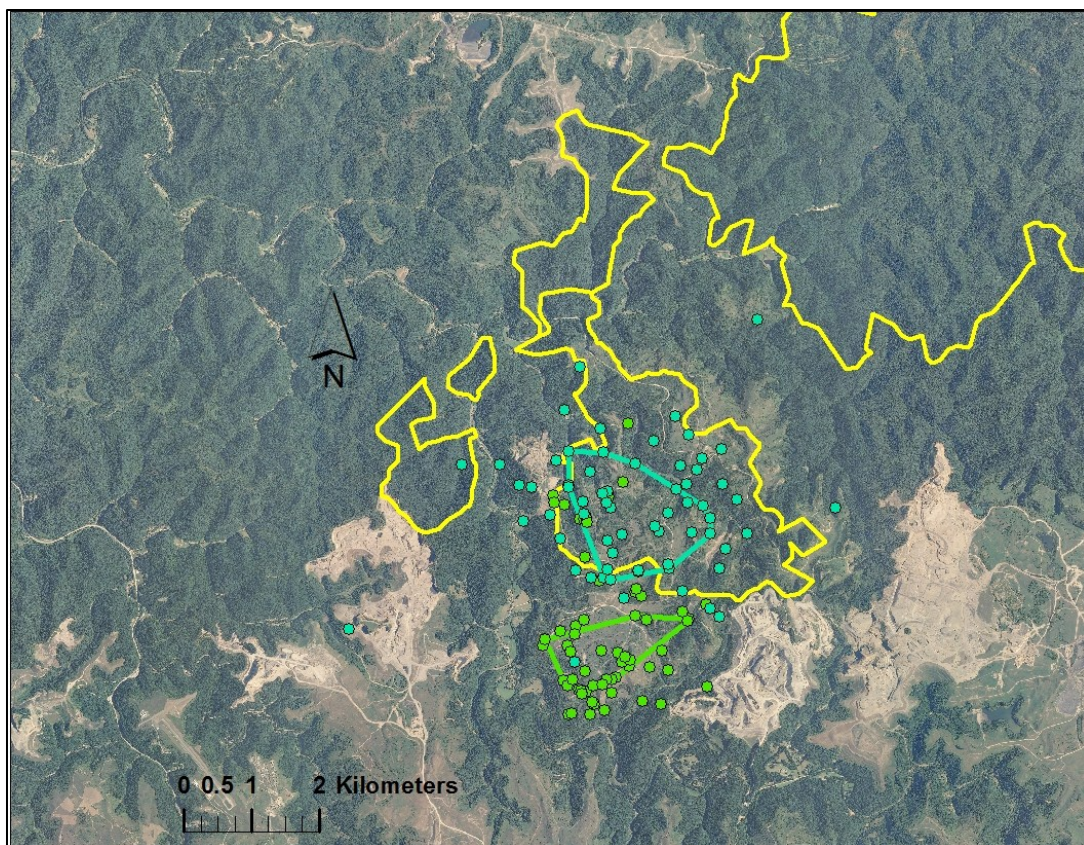


Figure A-31. The 50% minimum convex polygon core area overlap for F1 and M1 with the associated locations.

F1 in teal and M1 in bright green; the wildlife management area and Robinson Forest boundaries are represented in yellow.

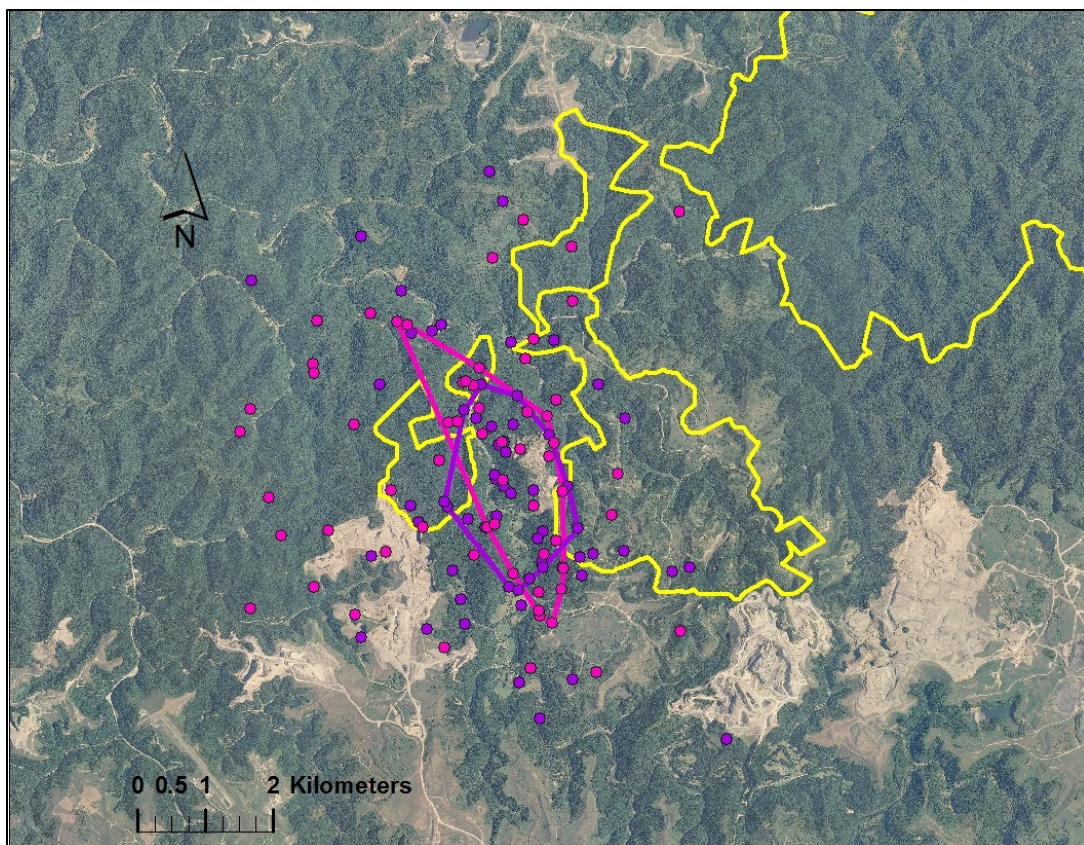


Figure A-32. The 50% minimum convex polygon core area overlap for F2 and F3 with the associated locations.

F2 in purple and F3 in dark pink; the wildlife management area and Robinson Forest boundaries are represented in yellow.

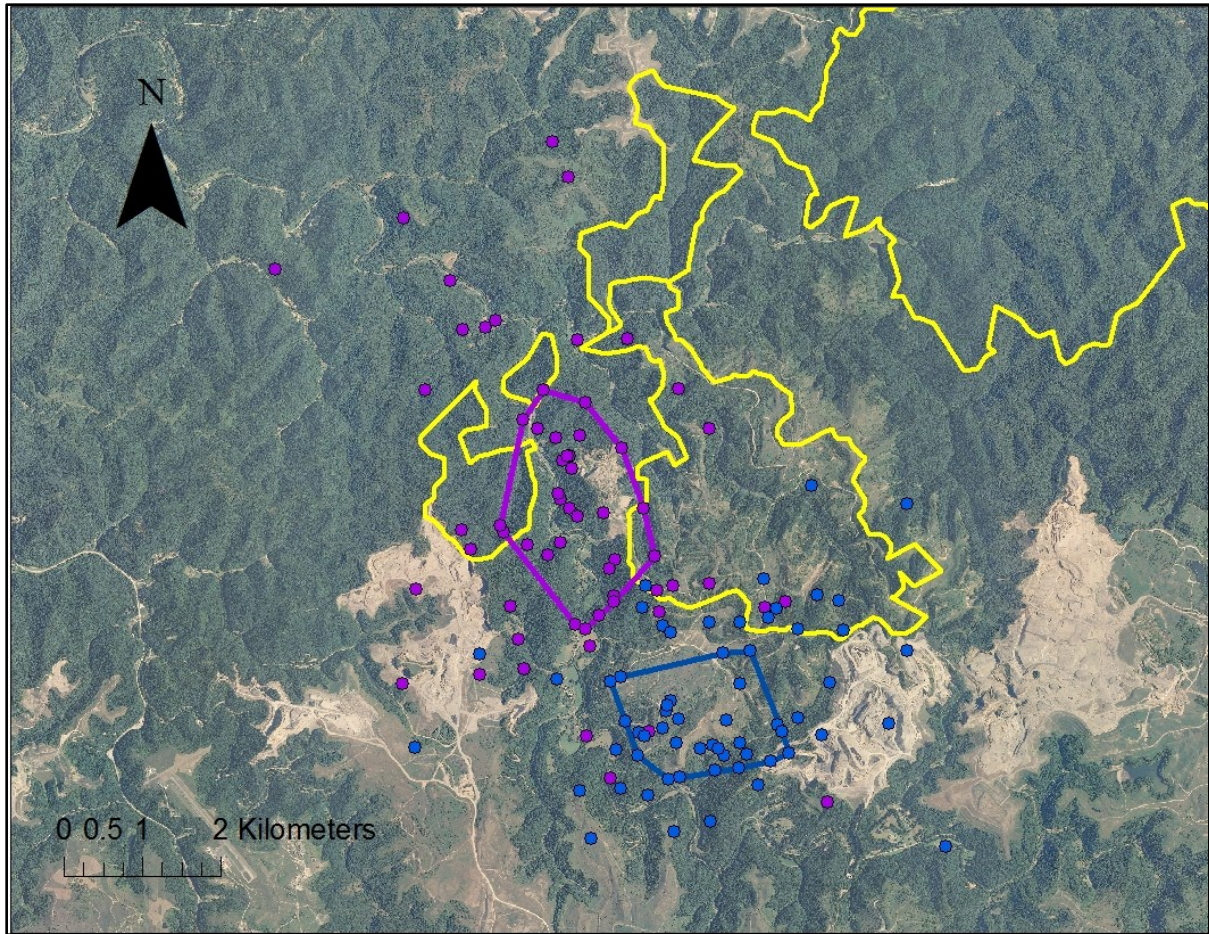


Figure A-33. The 50% minimum convex polygon core area overlap for F2 and F4 with the associated locations.

F2 in purple and F4 in dark blue; the wildlife management area and Robinson Forest boundaries are represented in yellow.

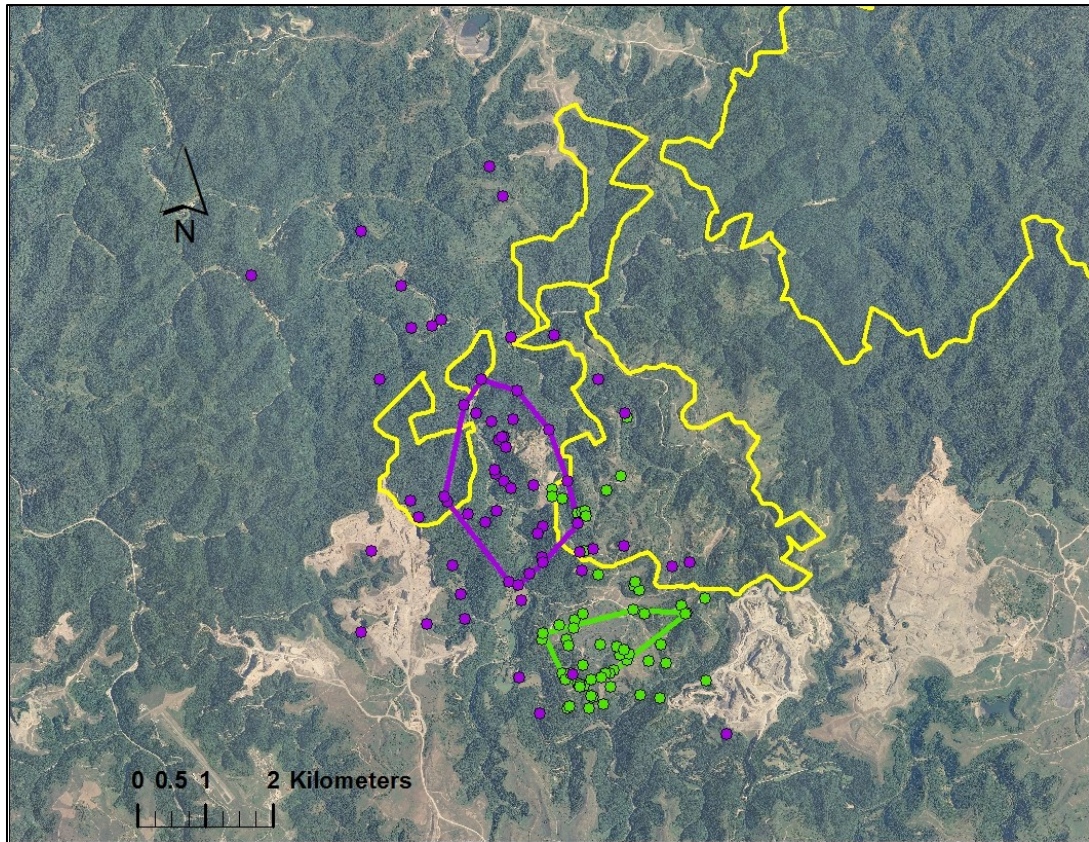


Figure A-34. The 50% minimum convex polygon core area overlap for F2 and M1 with the associated locations.

F2 in purple and M1 in bright green; the wildlife management area and Robinson Forest boundaries are represented in yellow.

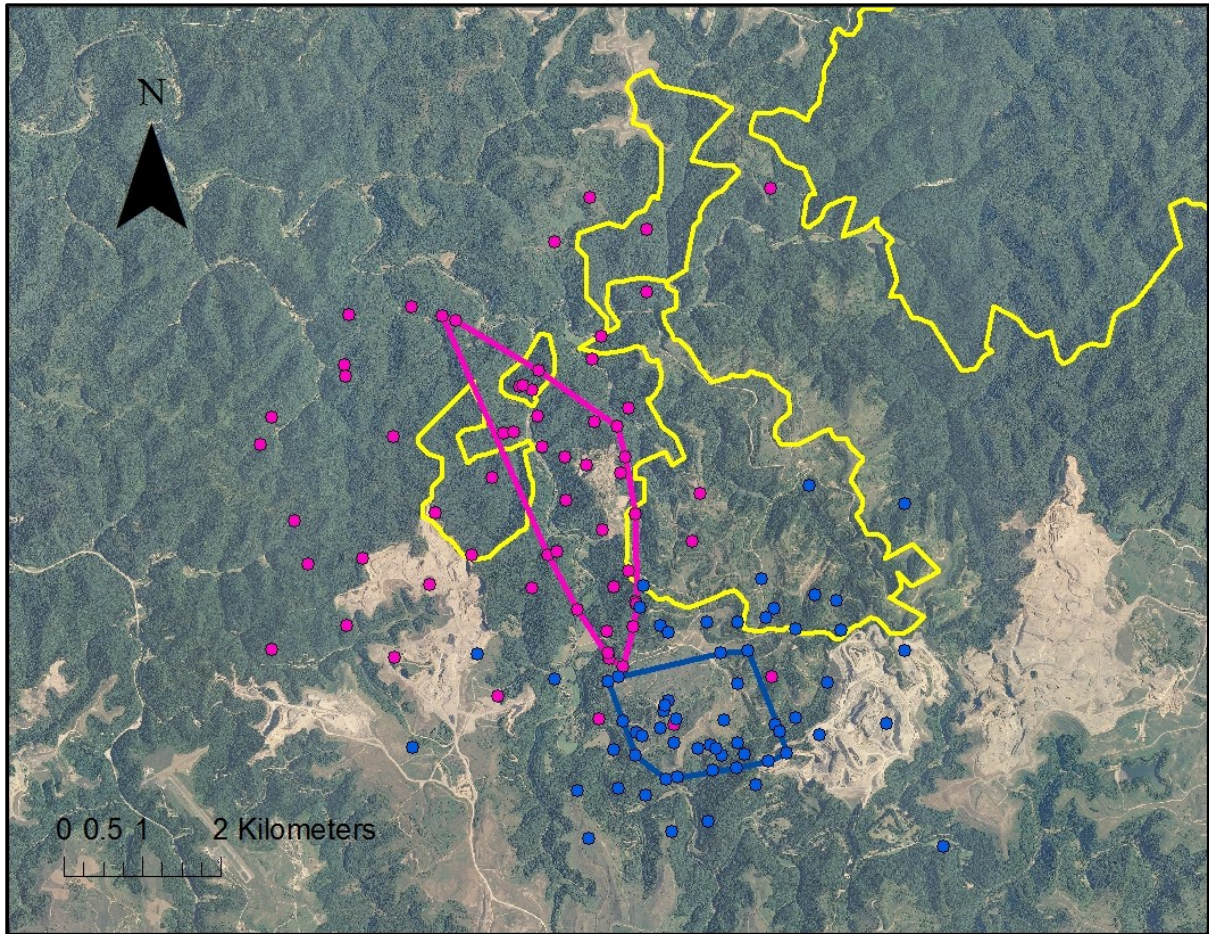


Figure A-35. The 50% minimum convex polygon core area overlap for F3 and F4 with the associated locations.

F3 in dark pink and F4 in dark blue; the wildlife management area and Robinson Forest boundaries are represented in yellow.

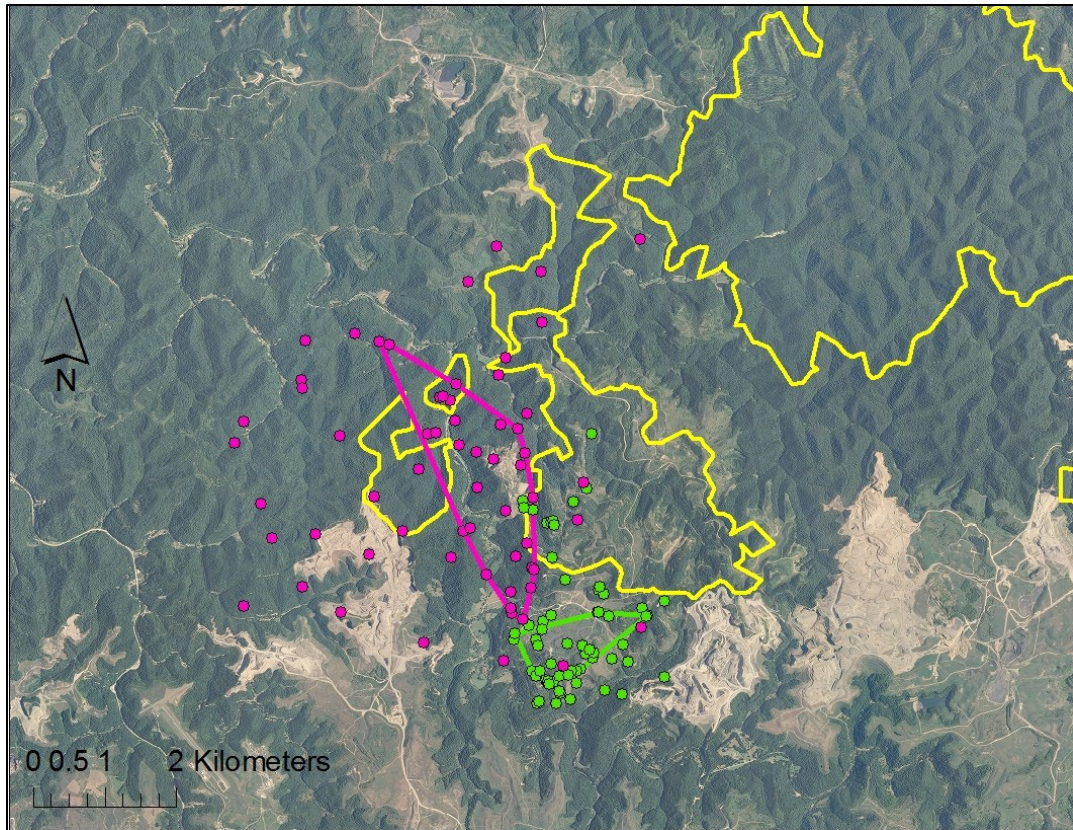


Figure A-36. The 50% minimum convex polygon core area overlap for F3 and M1 with the associated locations.

F3 in dark pink and M1 in bright green; the wildlife management area and Robinson Forest boundaries are represented in yellow.

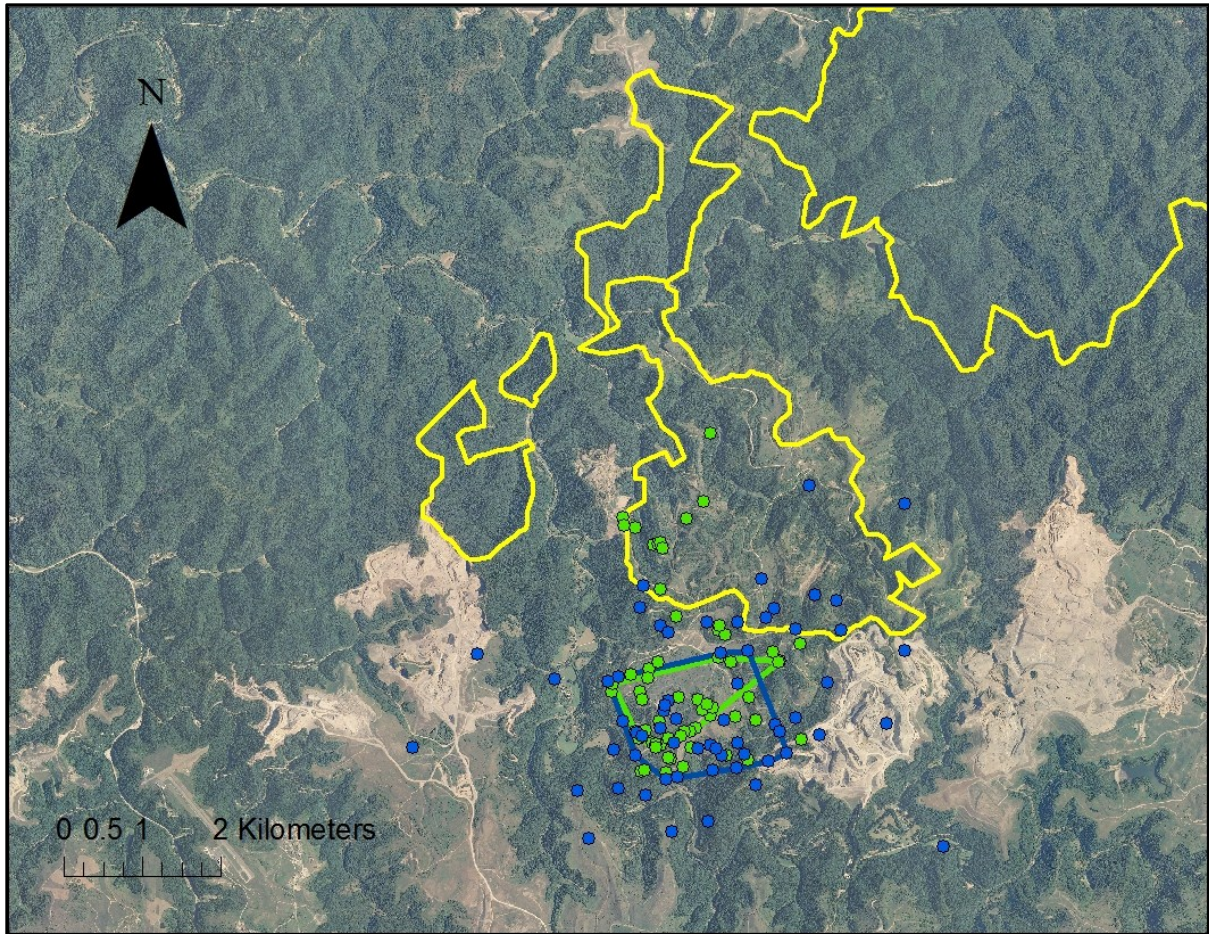


Figure A-37. The 50% minimum convex polygon core area overlap for F4 and M1 with the associated locations.

F4 in dark blue and M1 in bright green; the wildlife management area and Robinson Forest boundaries are represented in yellow.

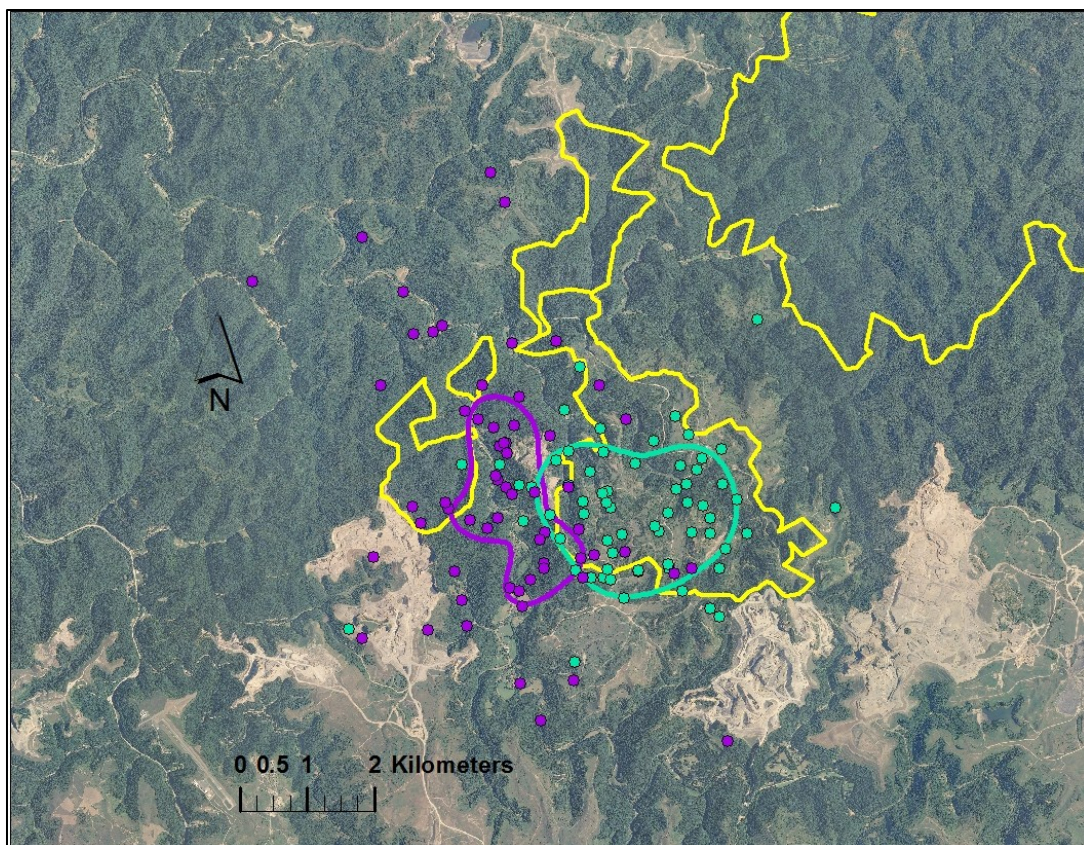


Figure A-38. The 50% adaptive kernel core area overlap for F1 and F2 with the associated locations.

F1 in teal and F2 in purple; the wildlife management area and Robinson Forest boundaries are represented in yellow.

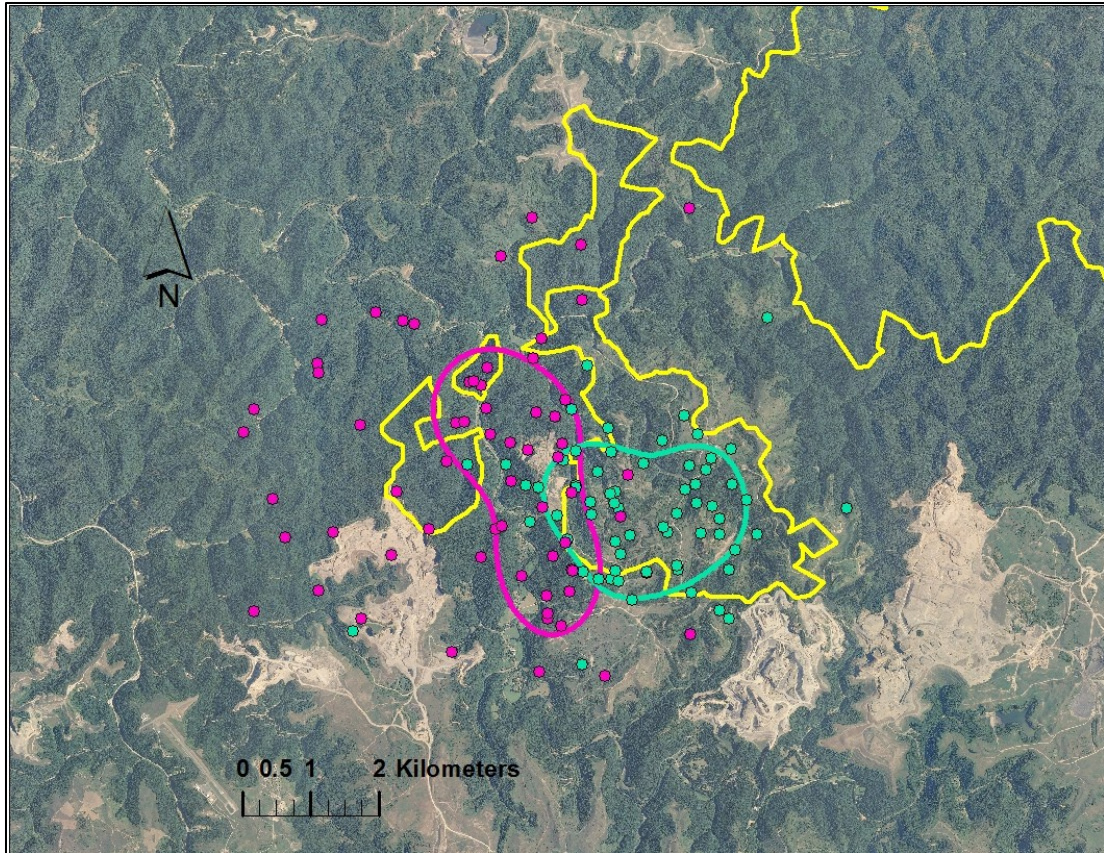


Figure A-39. The 50% adaptive kernel core area overlap for F1 and F3 with the associated locations.

F1 in teal and F3 in dark pink; the wildlife management area and Robinson Forest boundaries are represented in yellow.

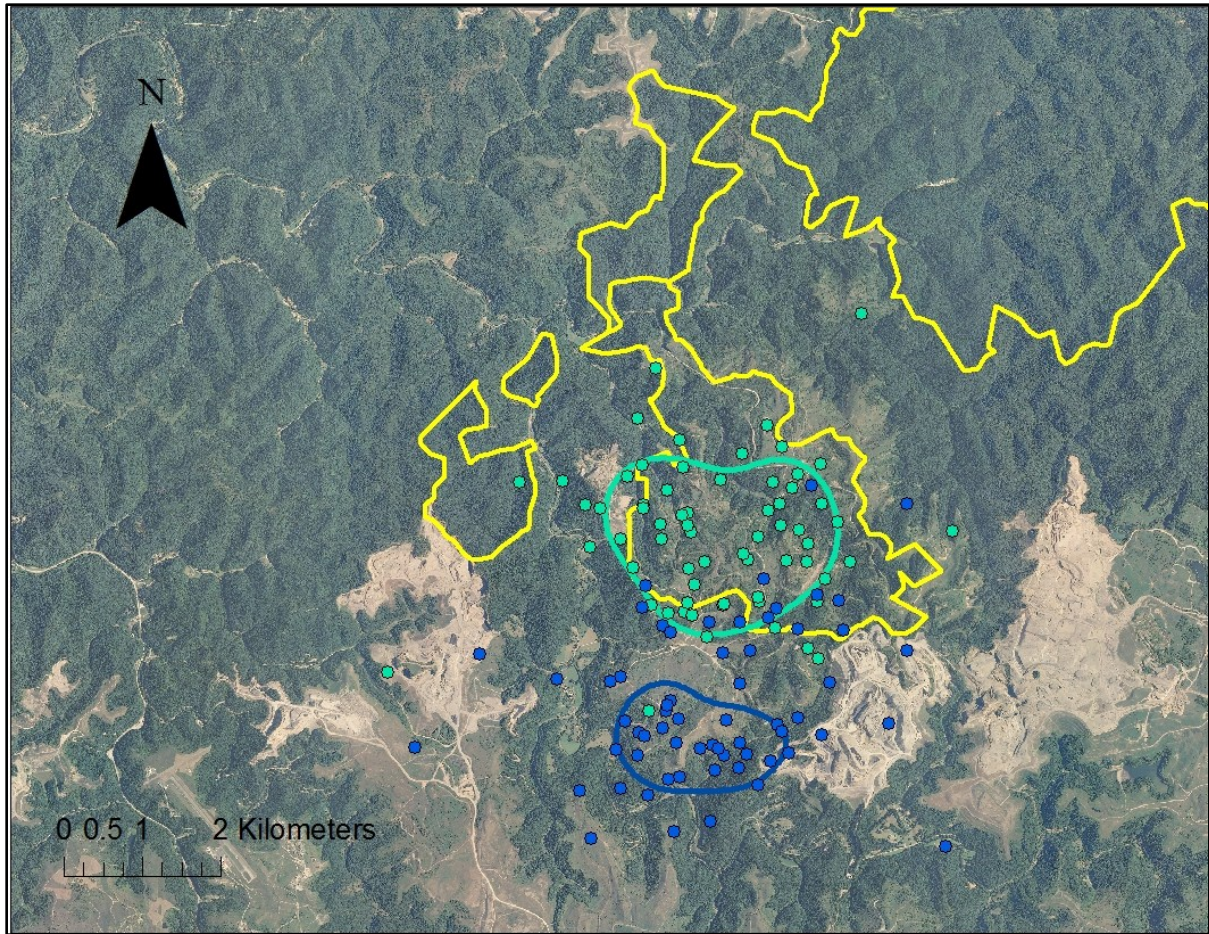


Figure A-40. The 50% adaptive kernel core area overlap for F1 and F4 with the associated locations.

F1 in teal and F4 in dark blue; the wildlife management area and Robinson Forest boundaries are represented in yellow.

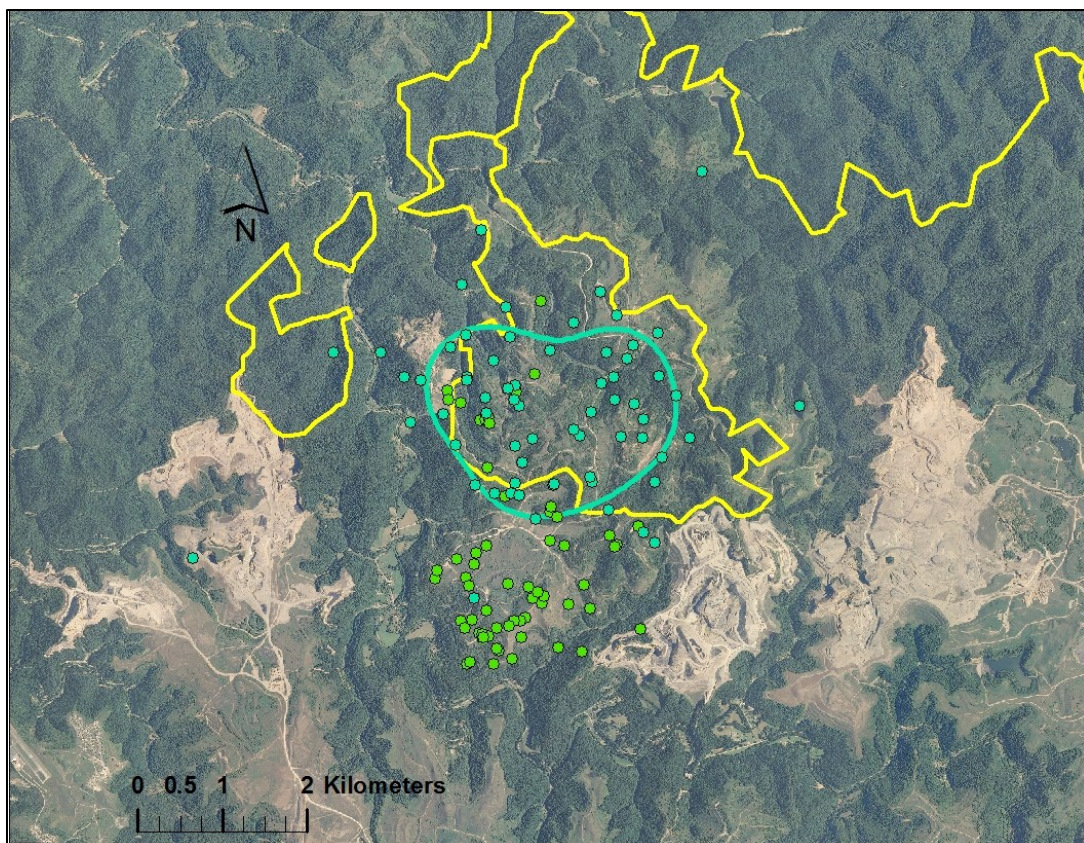


Figure A-41. The 50% adaptive kernel core area overlap for F1 and M1 with the associated locations.

F1 in teal and M1 in bright green; the wildlife management area and Robinson Forest boundaries are represented in yellow.

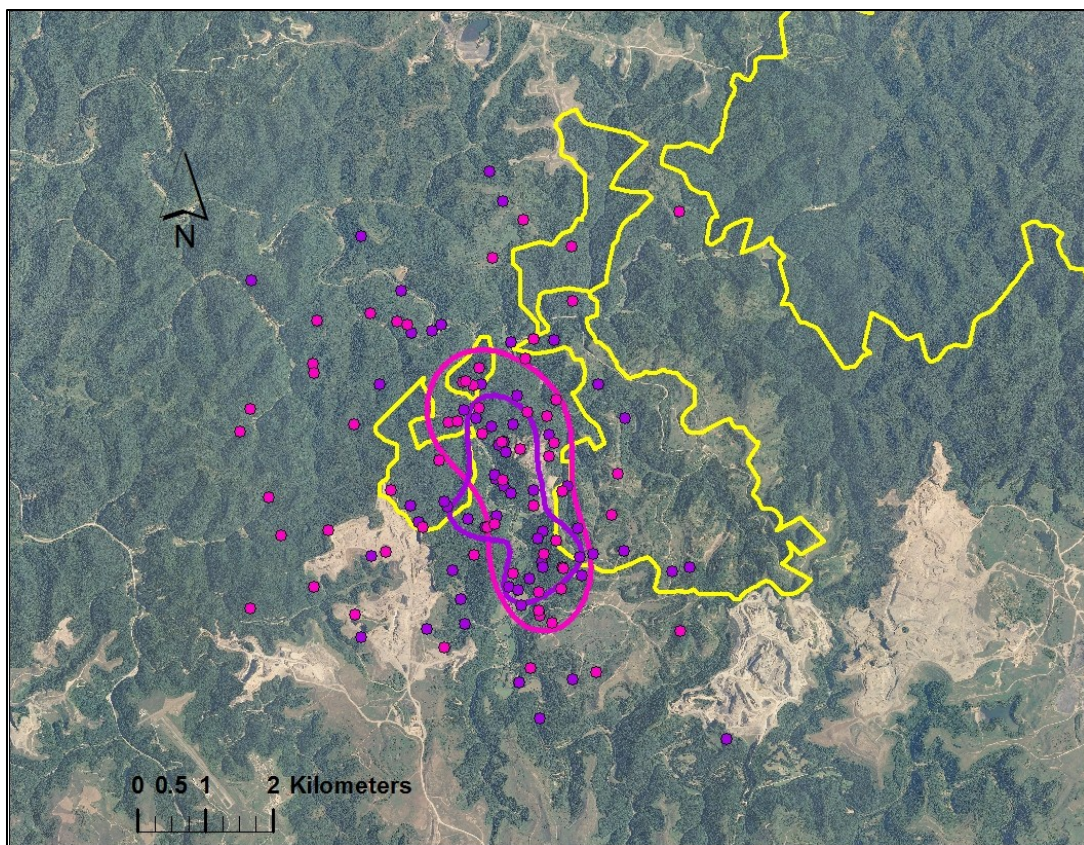


Figure A-42. The 50% adaptive kernel core area overlap for F2 and F3 with the associated locations.

F2 in purple and F3 in dark pink; the wildlife management area and Robinson Forest boundaries are represented in yellow.

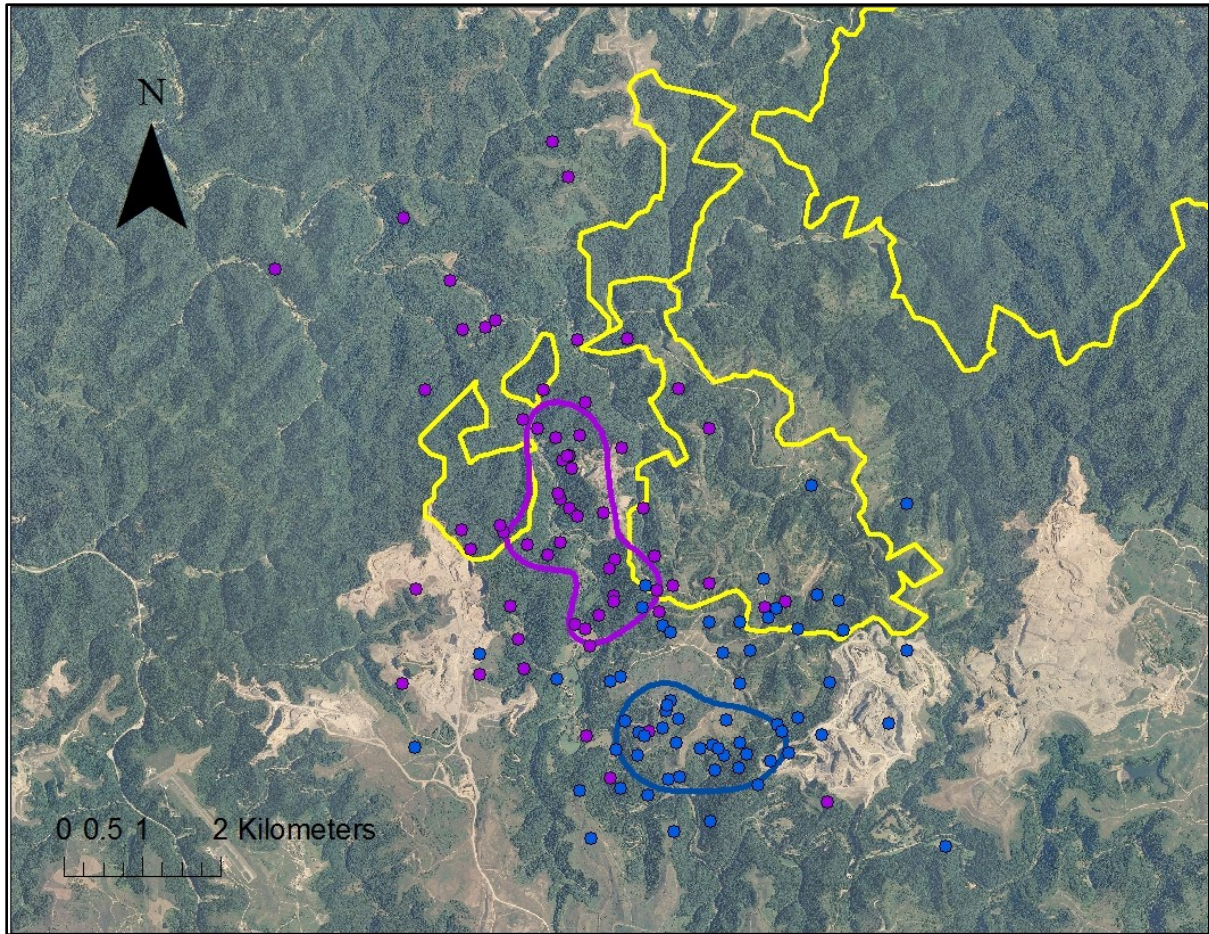


Figure A-43. The 50% adaptive kernel core area overlap for F2 and F4 with the associated locations.

F2 in purple and F4 in dark blue; the wildlife management area and Robinson Forest boundaries are represented in yellow.

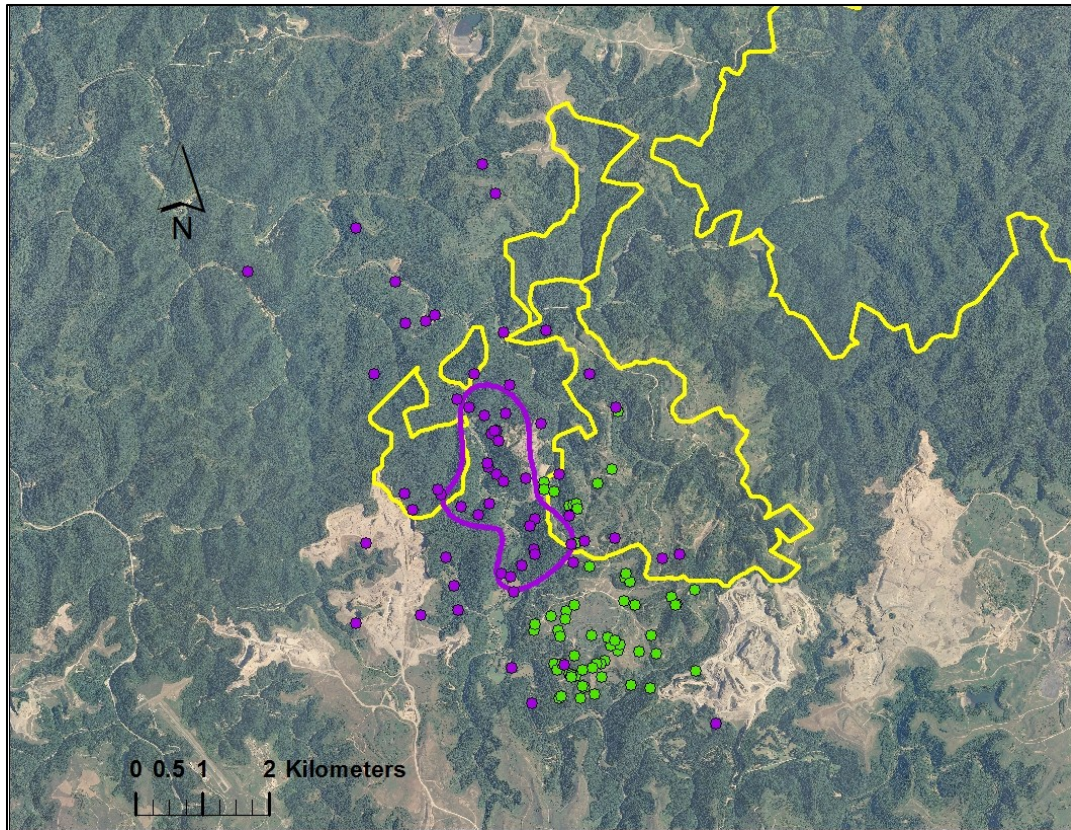


Figure A-44. The 50% adaptive kernel core area overlap for F2 and M1 with the associated locations.

F2 in purple and M1 in bright green; the wildlife management area and Robinson Forest boundaries are represented in yellow.

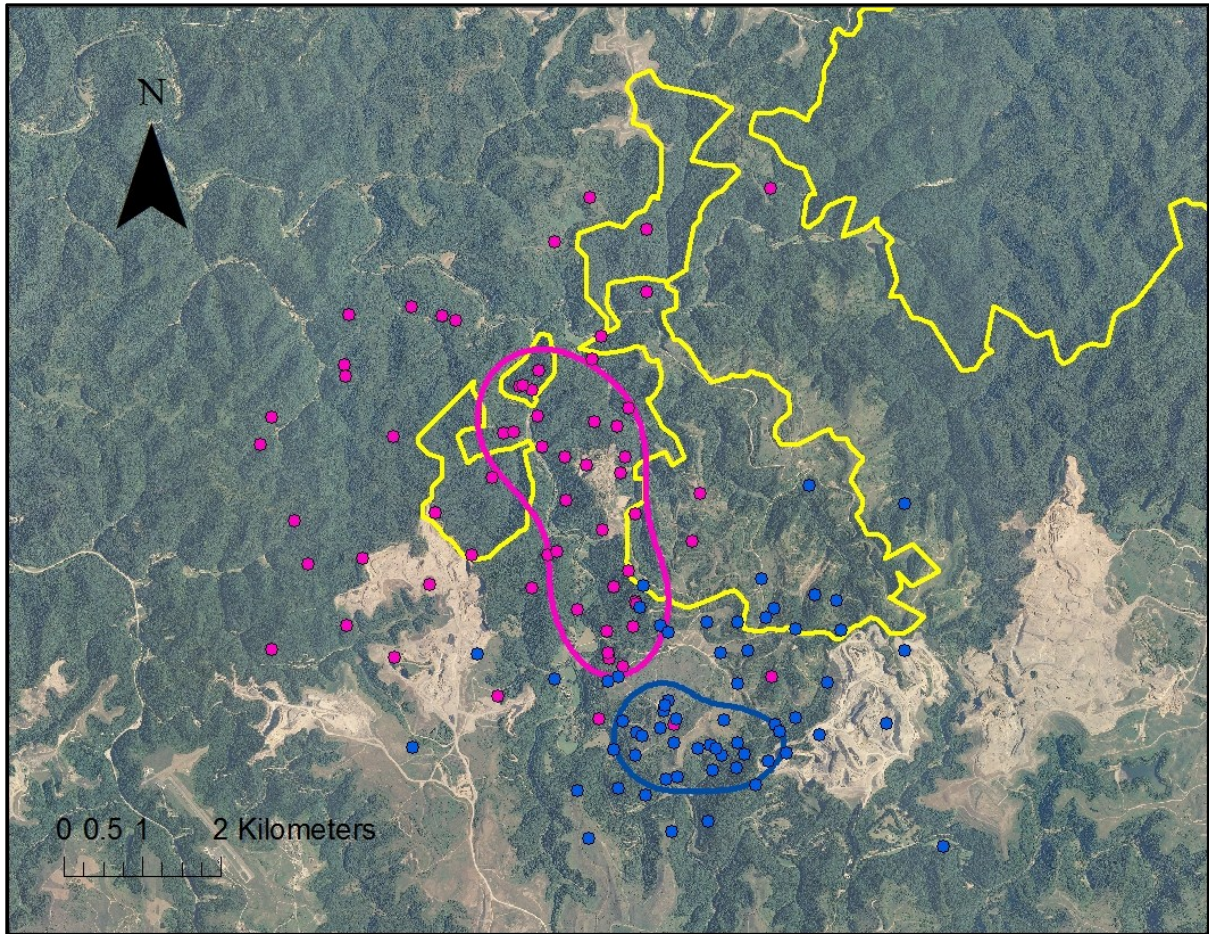


Figure A-45. The 50% adaptive kernel core area overlap for F3 and F4 with the associated locations.

F3 in dark pink and F4 in dark blue; the wildlife management area and Robinson Forest boundaries are represented in yellow.

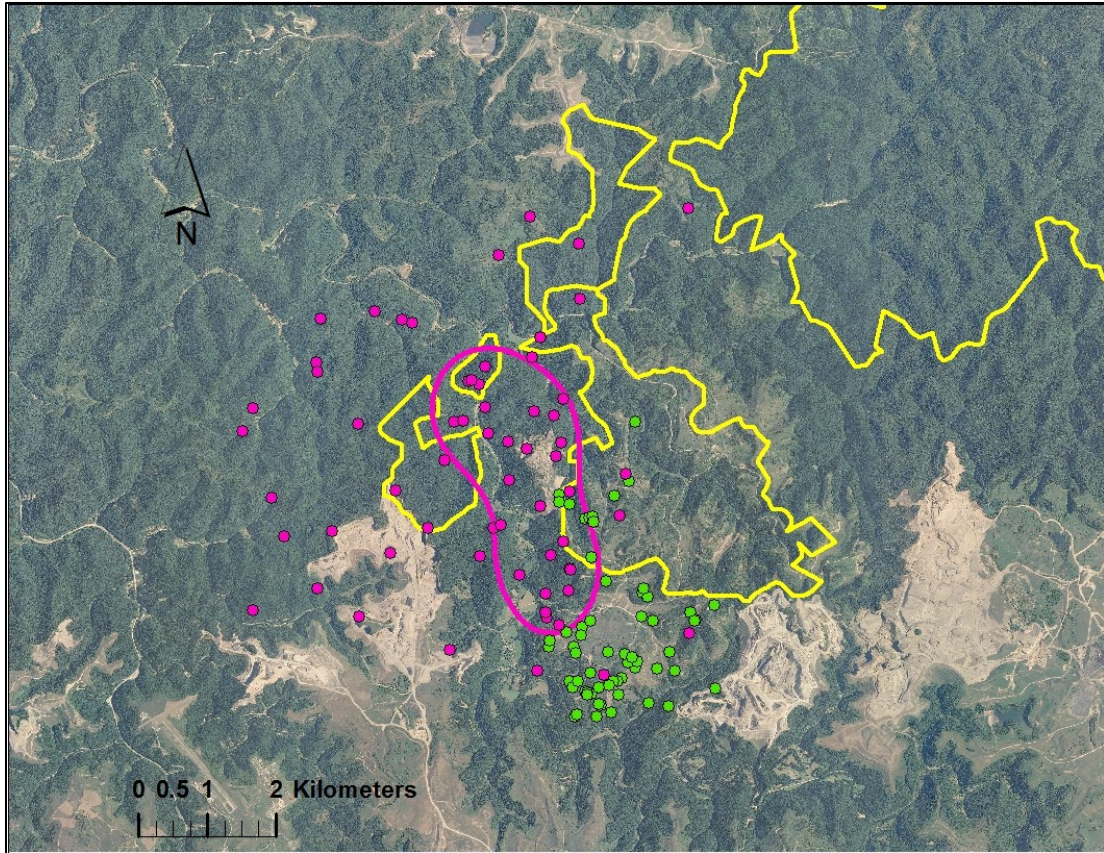


Figure A-46. The 50% adaptive kernel core area overlap for F3 and M1 with the associated locations.

F3 in dark pink and M1 in bright green; the wildlife management area and Robinson Forest boundaries are represented in yellow.

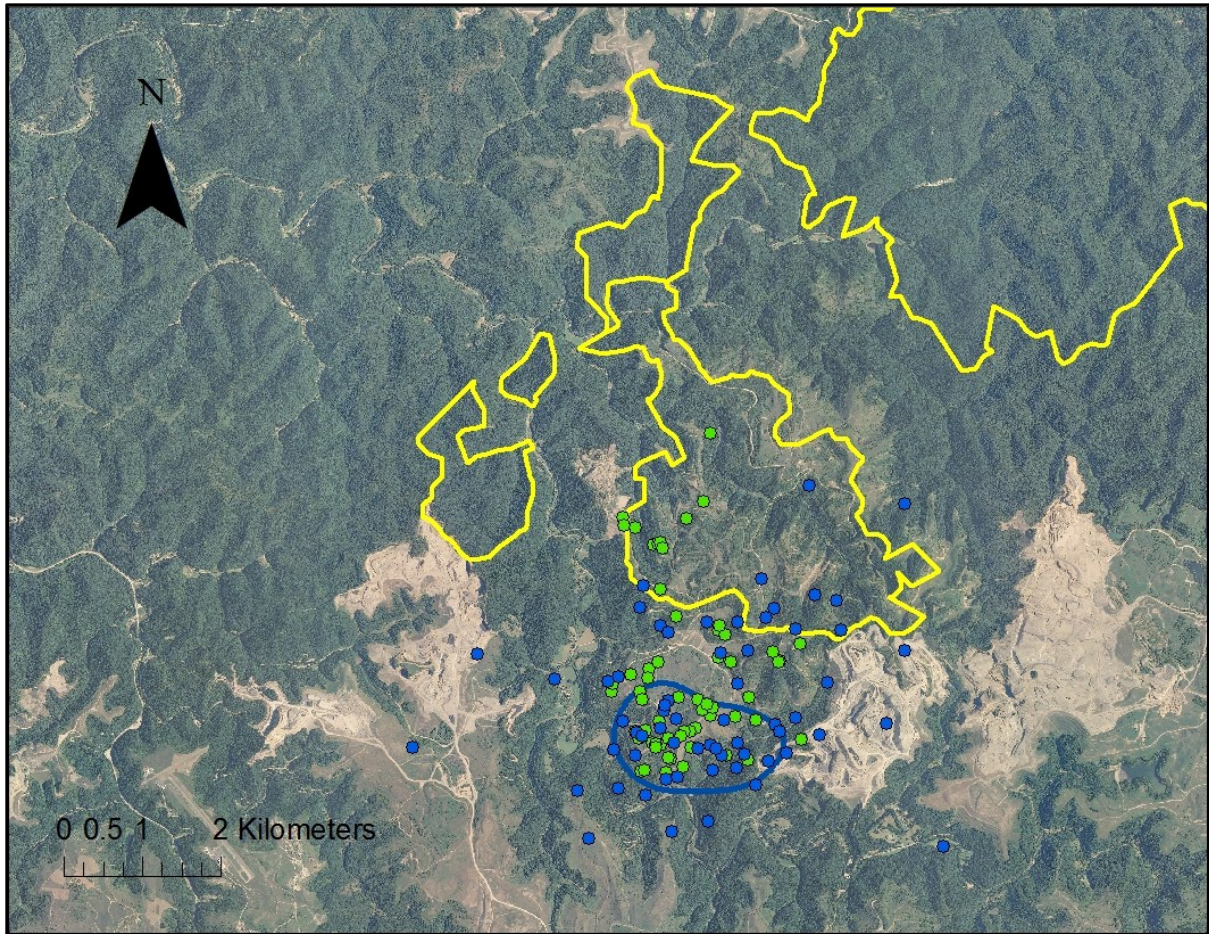


Figure A-47. The 50% adaptive kernel core area overlap for F4 and M1 with the associated locations.

F4 in dark blue and M1 in bright green; the wildlife management area and Robinson Forest boundaries are represented in yellow.

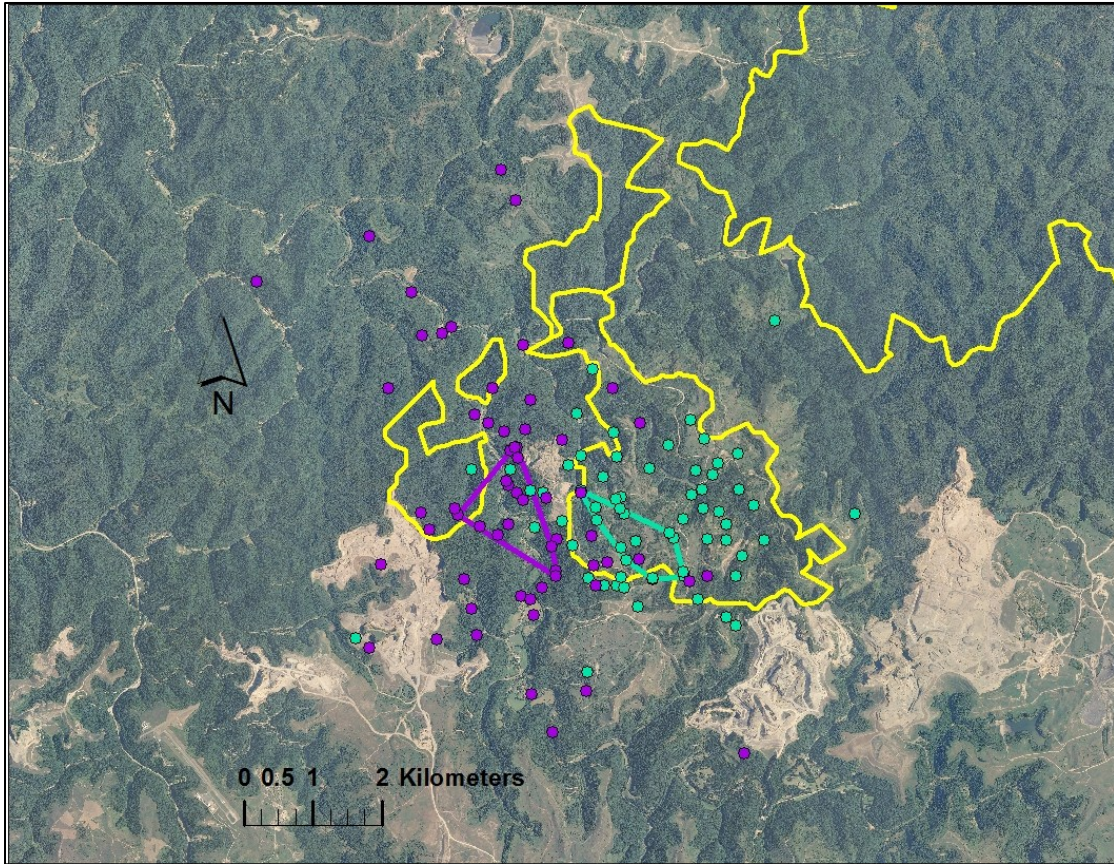


Figure A-48. The 25% minimum convex polygon core area overlap for F1 and F2 with the associated locations.

F1 in teal and F2 in purple; the wildlife management area and Robinson Forest boundaries are represented in yellow.

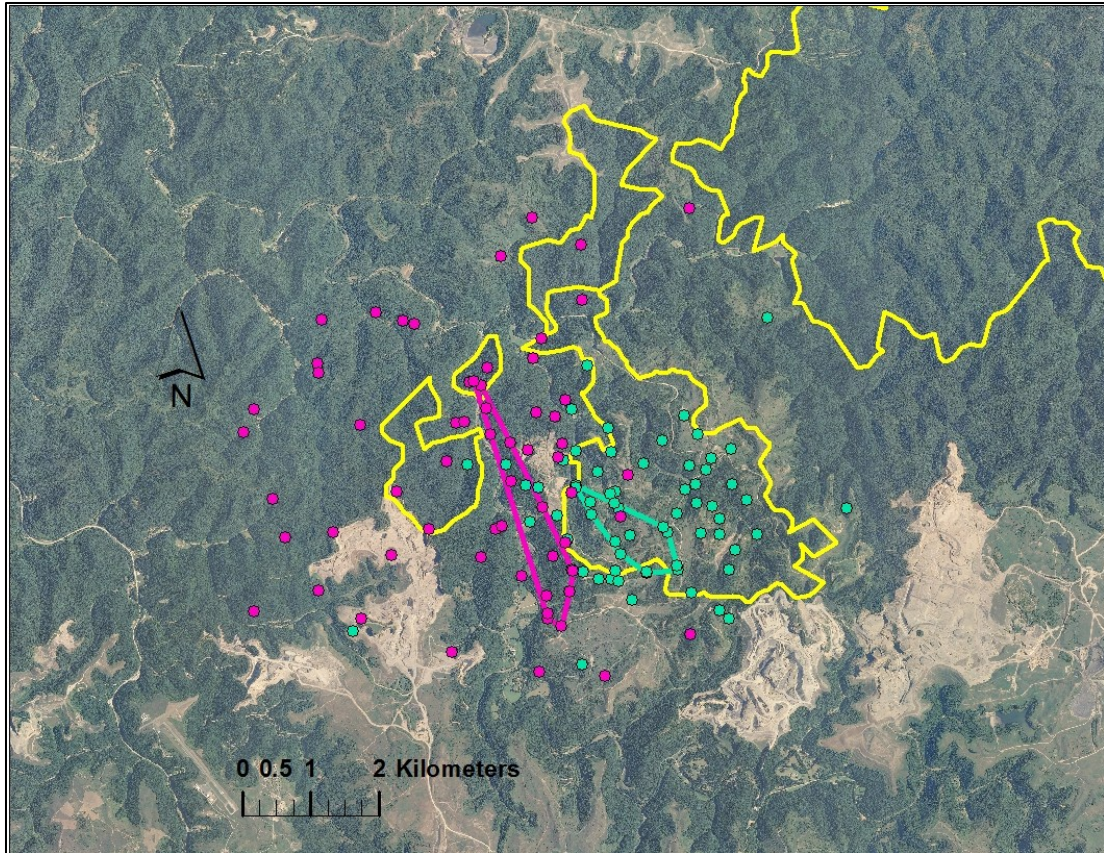


Figure A-49. The 25% minimum convex polygon core area overlap for F1 and F3 with the associated locations.

F1 in teal and F3 in dark pink; the wildlife management area and Robinson Forest boundaries are represented in yellow.

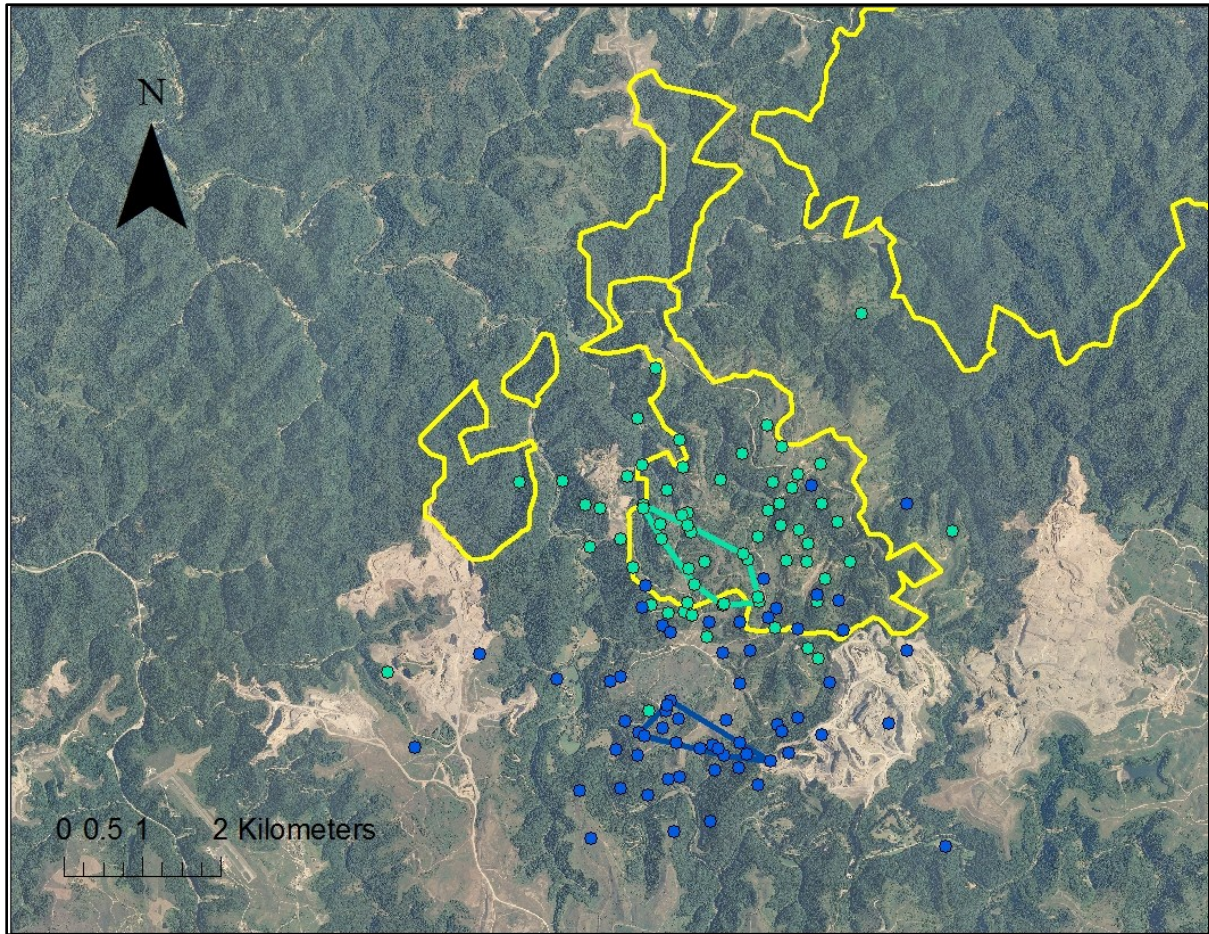


Figure A-50. The 25% minimum convex polygon core area overlap for F1 and F4 with the associated locations.

F1 in teal and F4 in dark blue; the wildlife management area and Robinson Forest boundaries are represented in yellow.

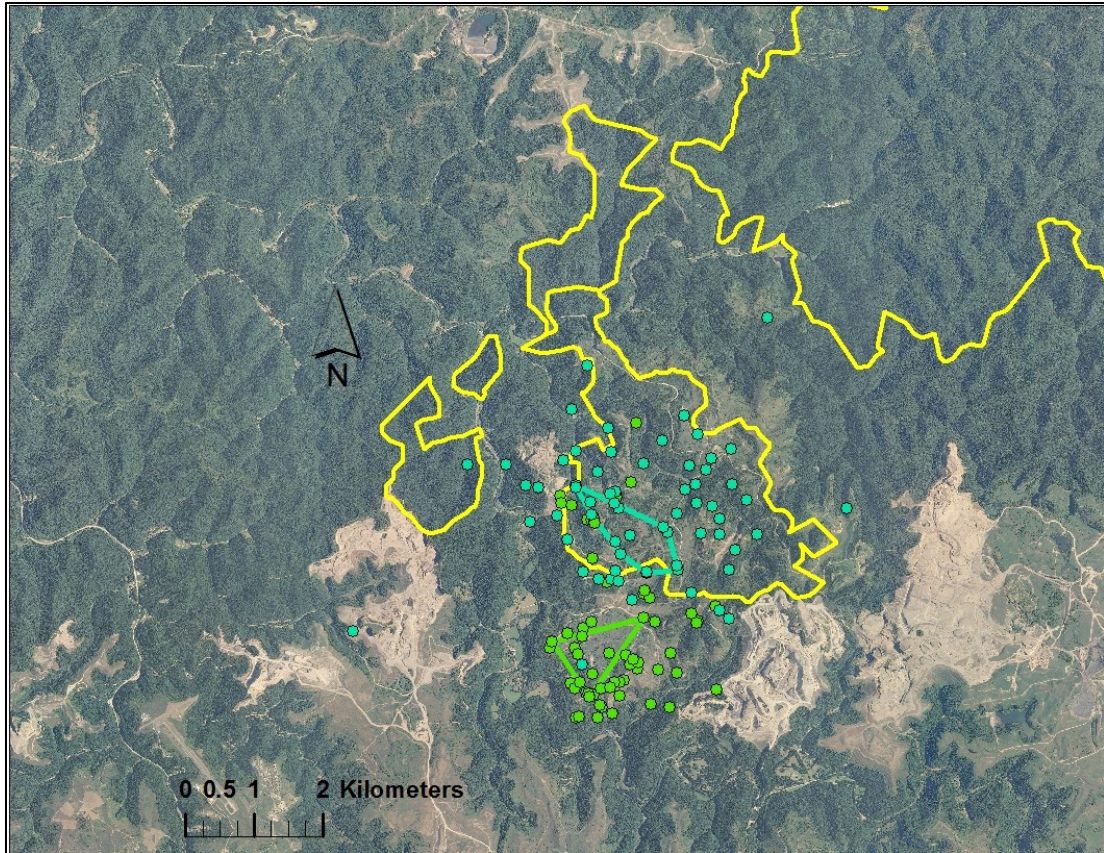


Figure A-51. The 25% minimum convex polygon core area overlap for F1 and M1 with the associated locations.

F1 in teal and M1 in bright green; the wildlife management area and Robinson Forest boundaries are represented in yellow.

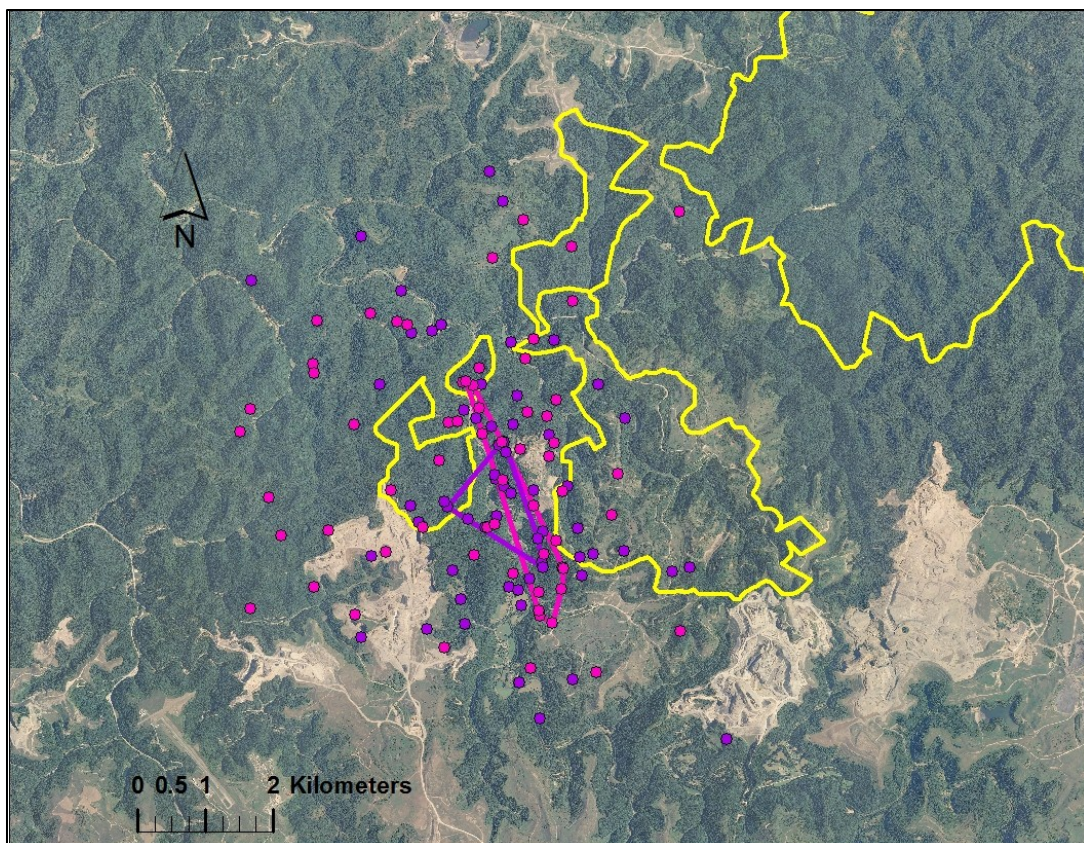


Figure A-52. The 25% minimum convex polygon core area overlap for F2 and F3 with the associated locations.

F2 in purple and F3 in dark pink; the wildlife management area and Robinson Forest boundaries are represented in yellow.

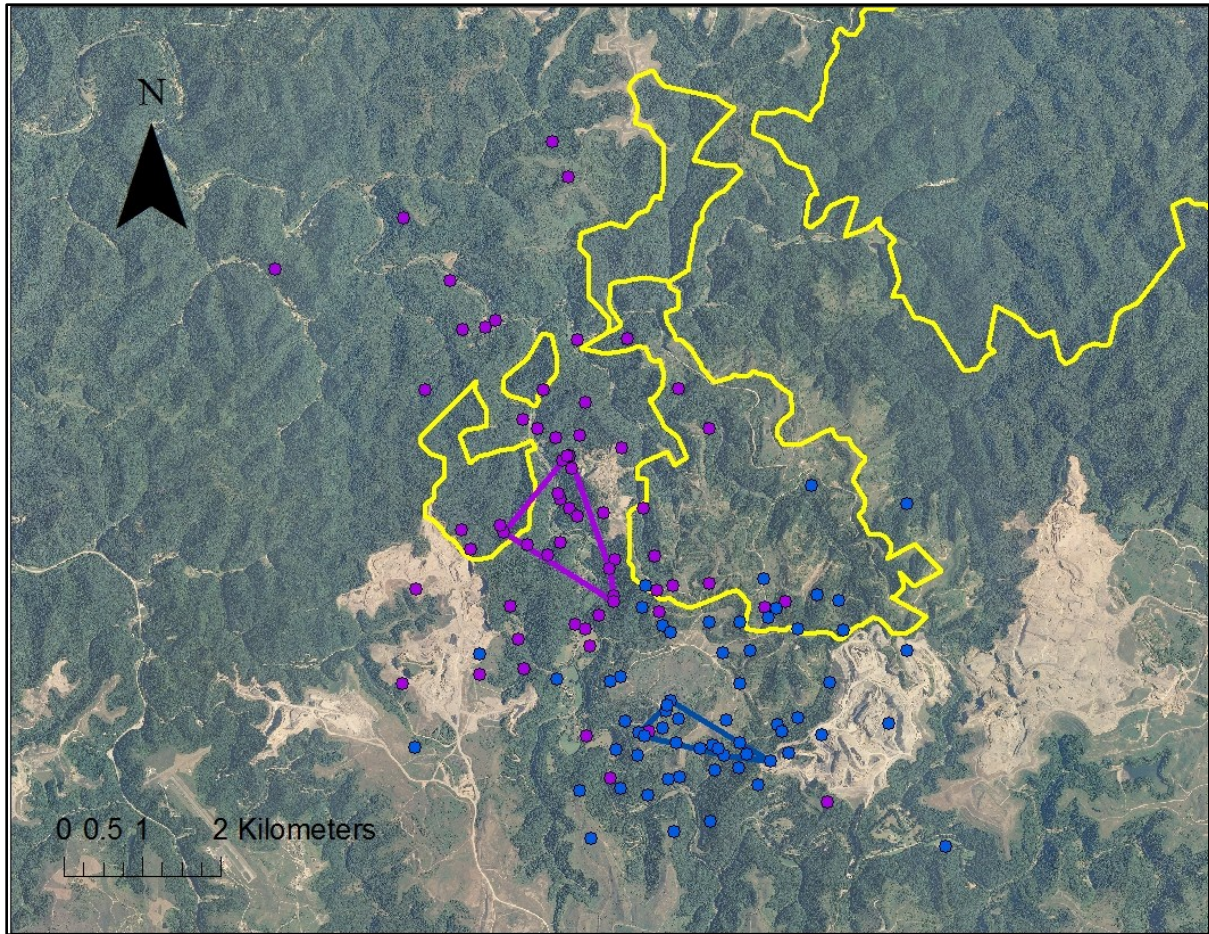


Figure A-53. The 25% minimum convex polygon core area overlap for F2 and F4 with the associated locations.

F2 in purple and F4 in dark blue; the wildlife management area and Robinson Forest boundaries are represented in yellow.

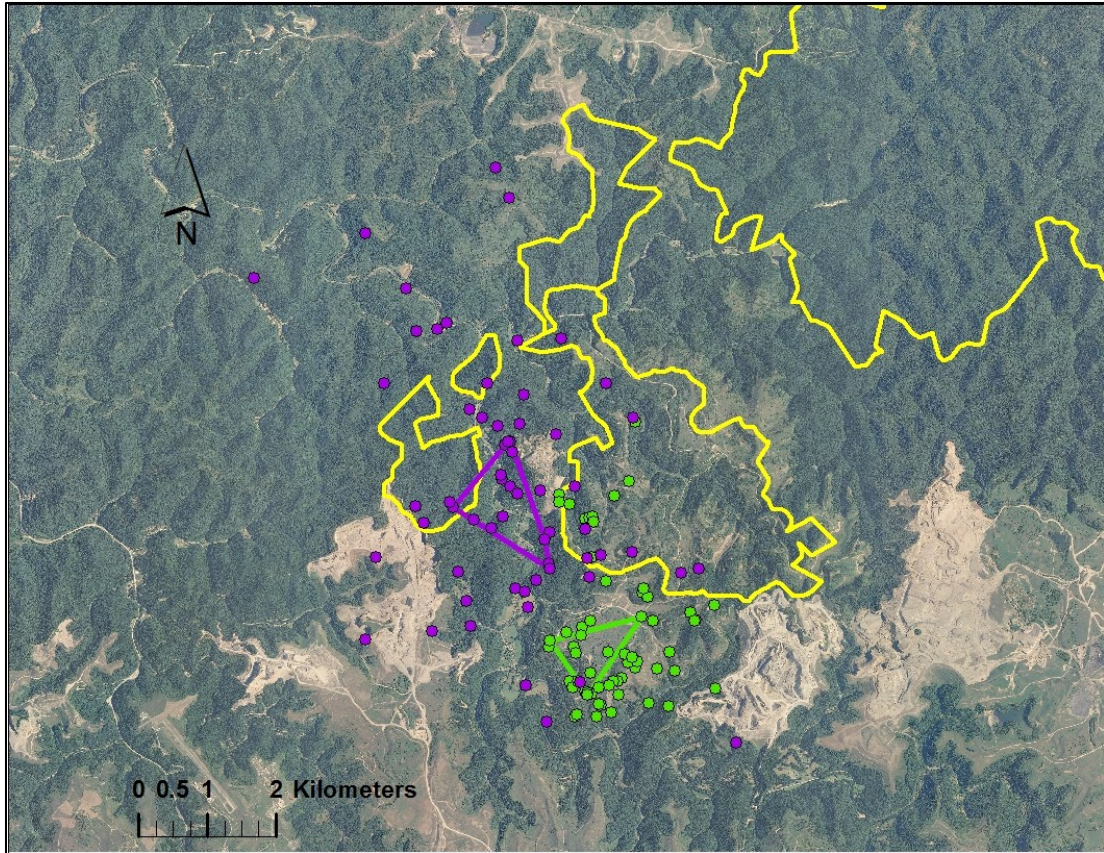


Figure A-54. The 25% minimum convex polygon core area overlap for F2 and M1 with the associated locations.

F2 in purple and M1 in bright green; the wildlife management area and Robinson Forest boundaries are represented in yellow.

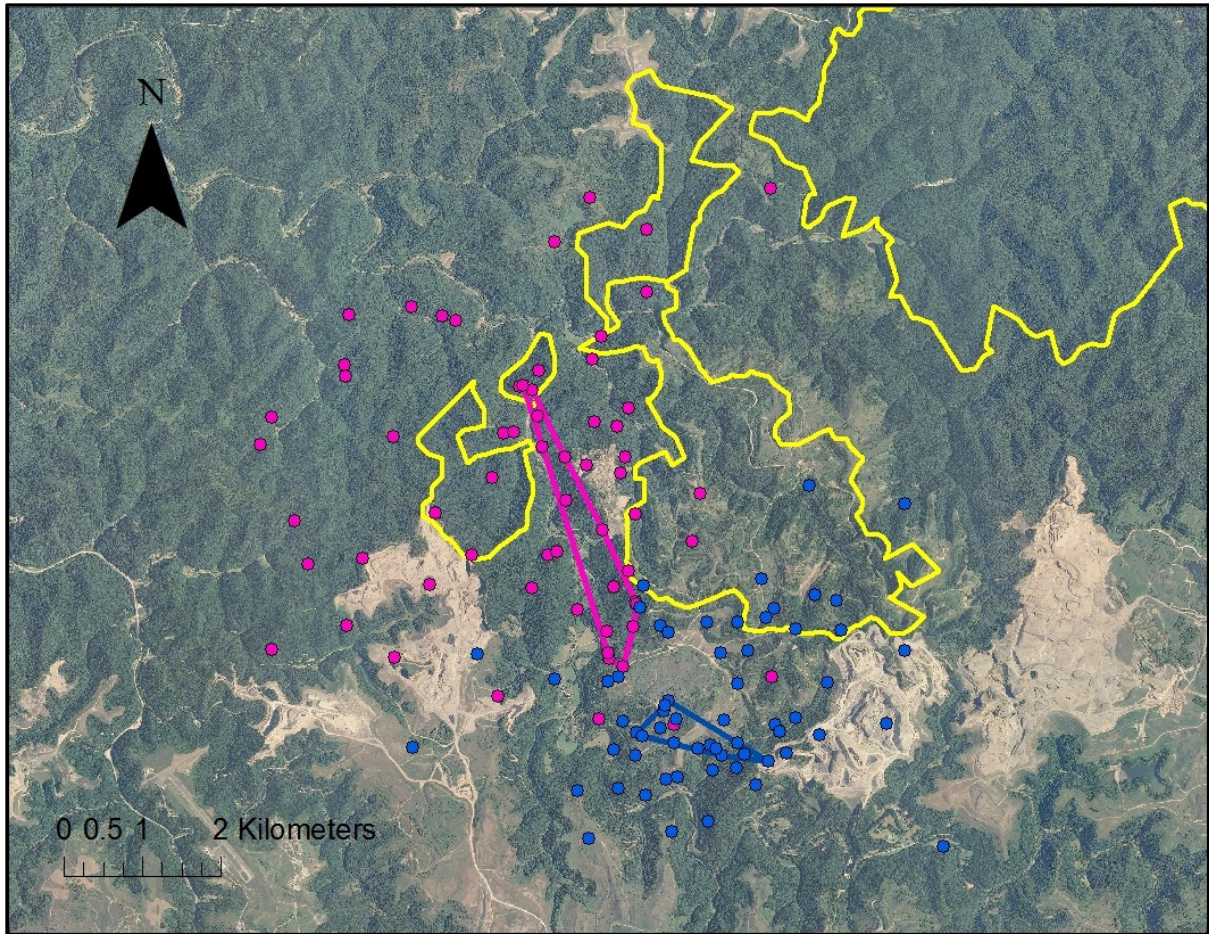


Figure A-55. The 25% minimum convex polygon core area overlap for F3 and F4 with the associated locations.

F3 in dark pink and F4 in dark blue; the wildlife management area and Robinson Forest boundaries are represented in yellow.

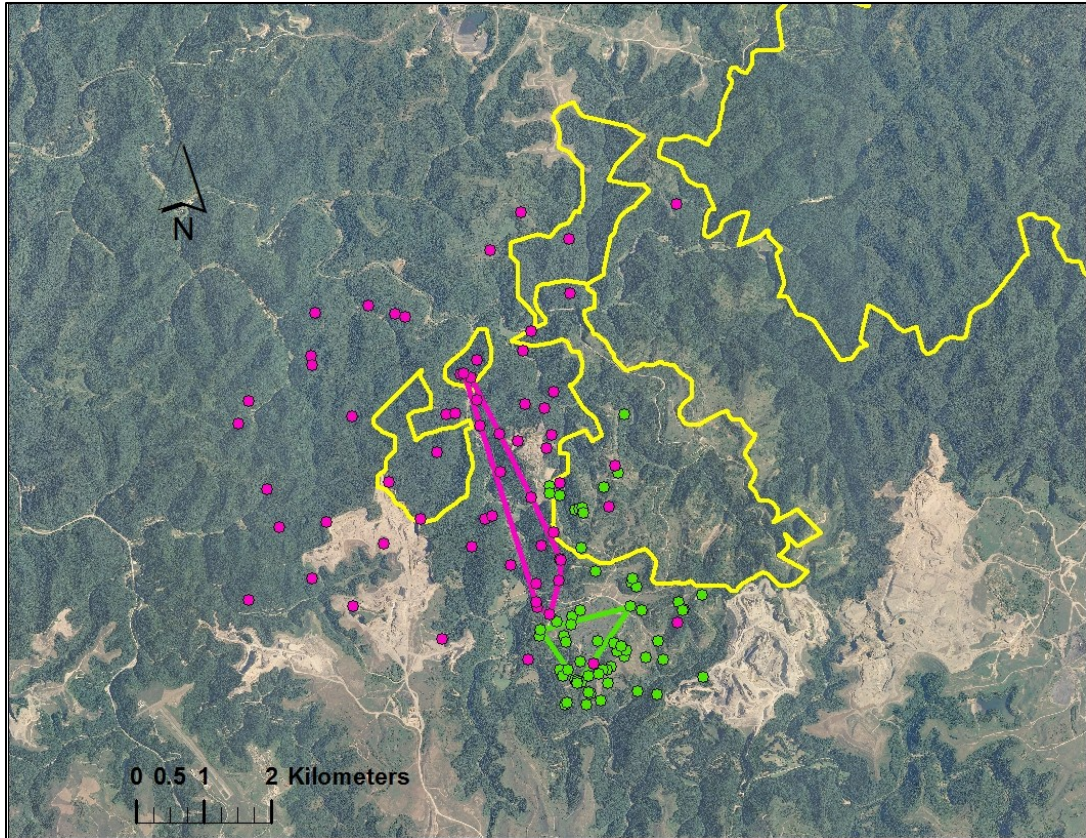


Figure A-56. The 25% minimum convex polygon core area overlap for F3 and M1 with the associated locations.

F3 in dark pink and M1 in bright green; the wildlife management area and Robinson Forest boundaries are represented in yellow.

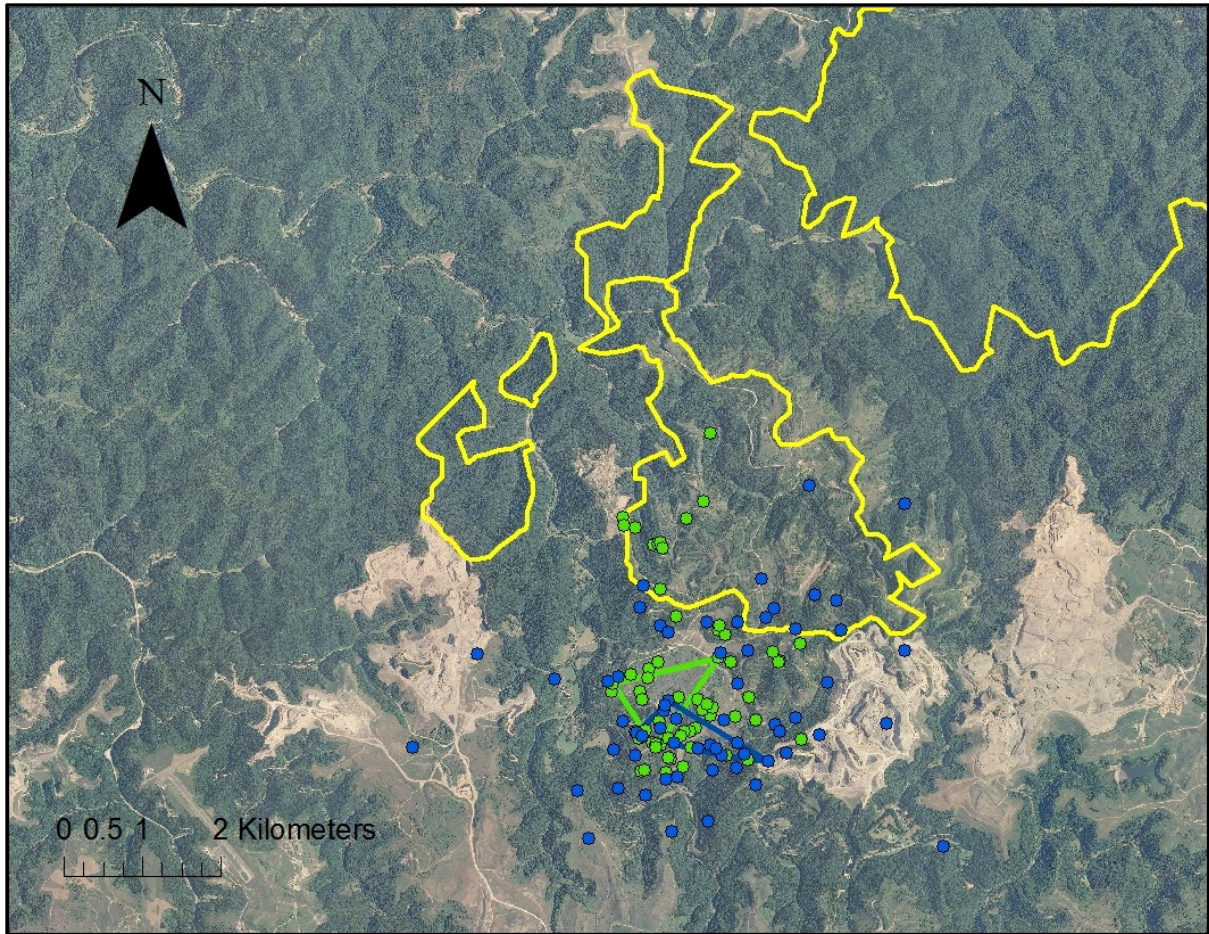


Figure A-57. The 25% minimum convex polygon core area overlap for F4 and M1 with the associated locations.

F4 in dark blue and M1 in bright green; the wildlife management area and Robinson Forest boundaries are represented in yellow.

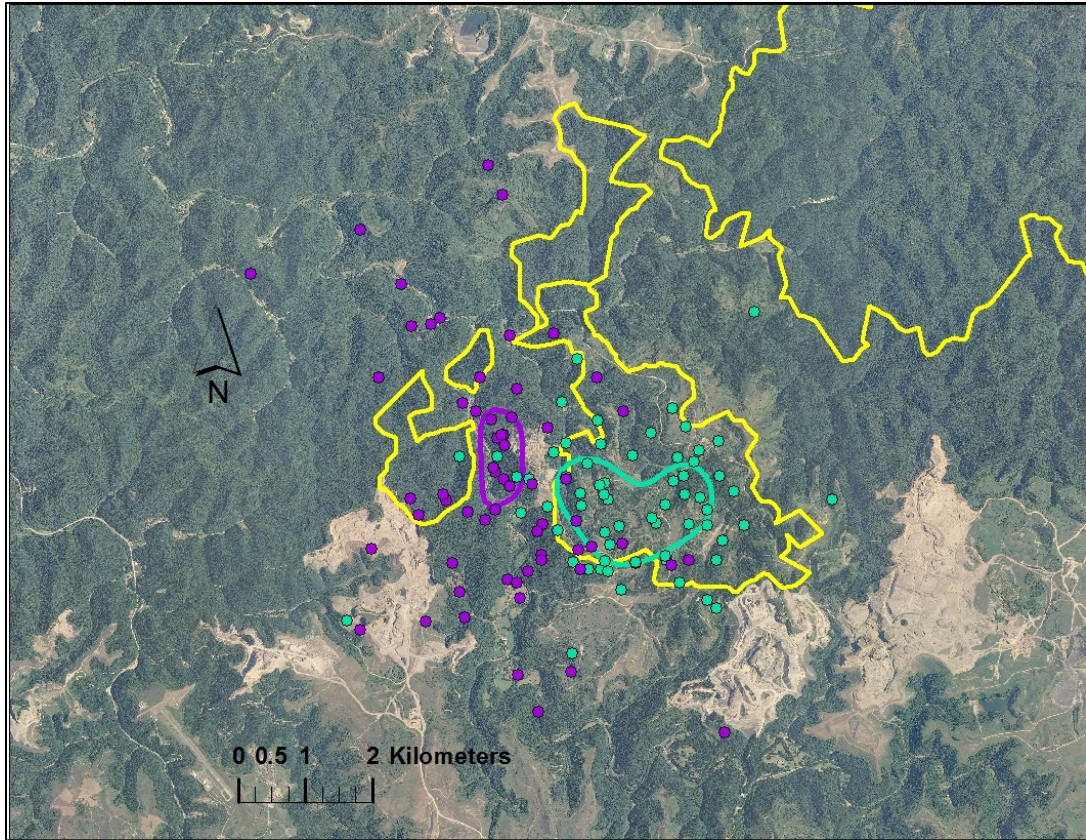


Figure A-58. The 25% adaptive kernel core area overlap for F1 and F2 with the associated locations.

F1 in teal and F2 in purple; the wildlife management area and Robinson Forest boundaries are represented in yellow.

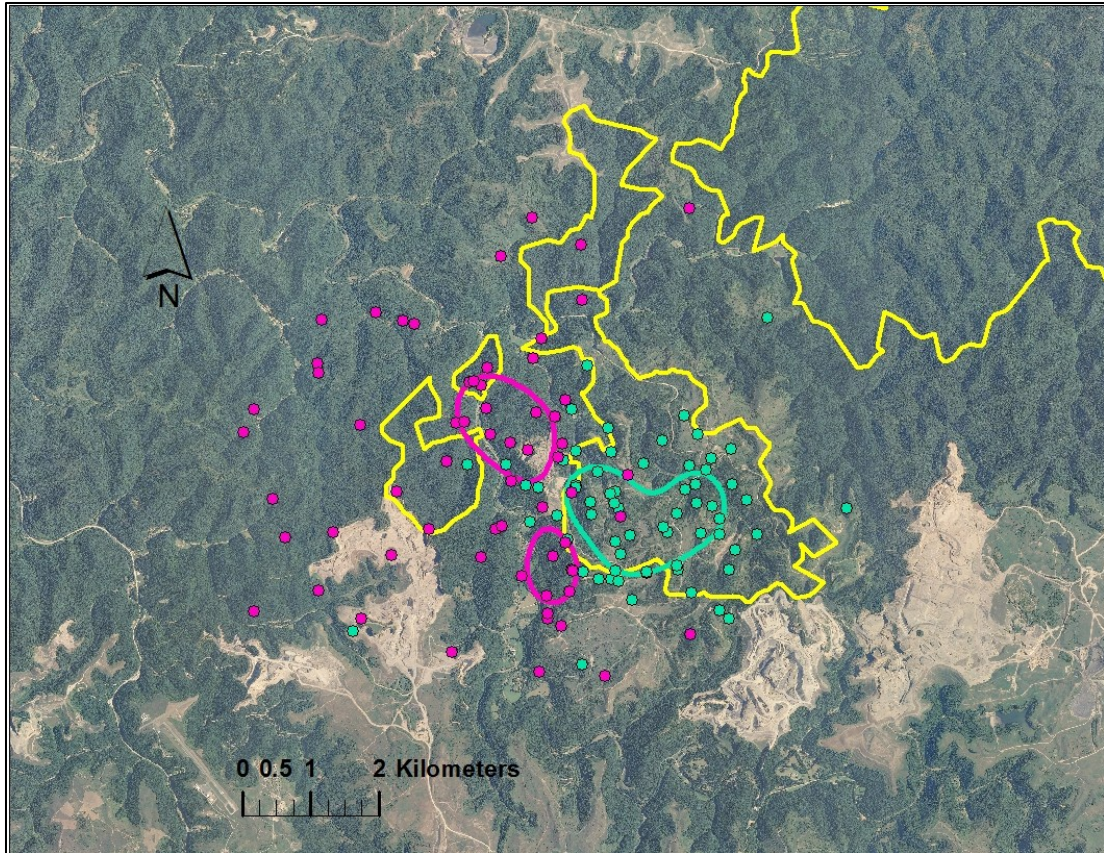


Figure A-59. The 25% adaptive kernel core area overlap for F1 and F3 with the associated locations.

F1 in teal and F3 in dark pink; the wildlife management area and Robinson Forest boundaries are represented in yellow.

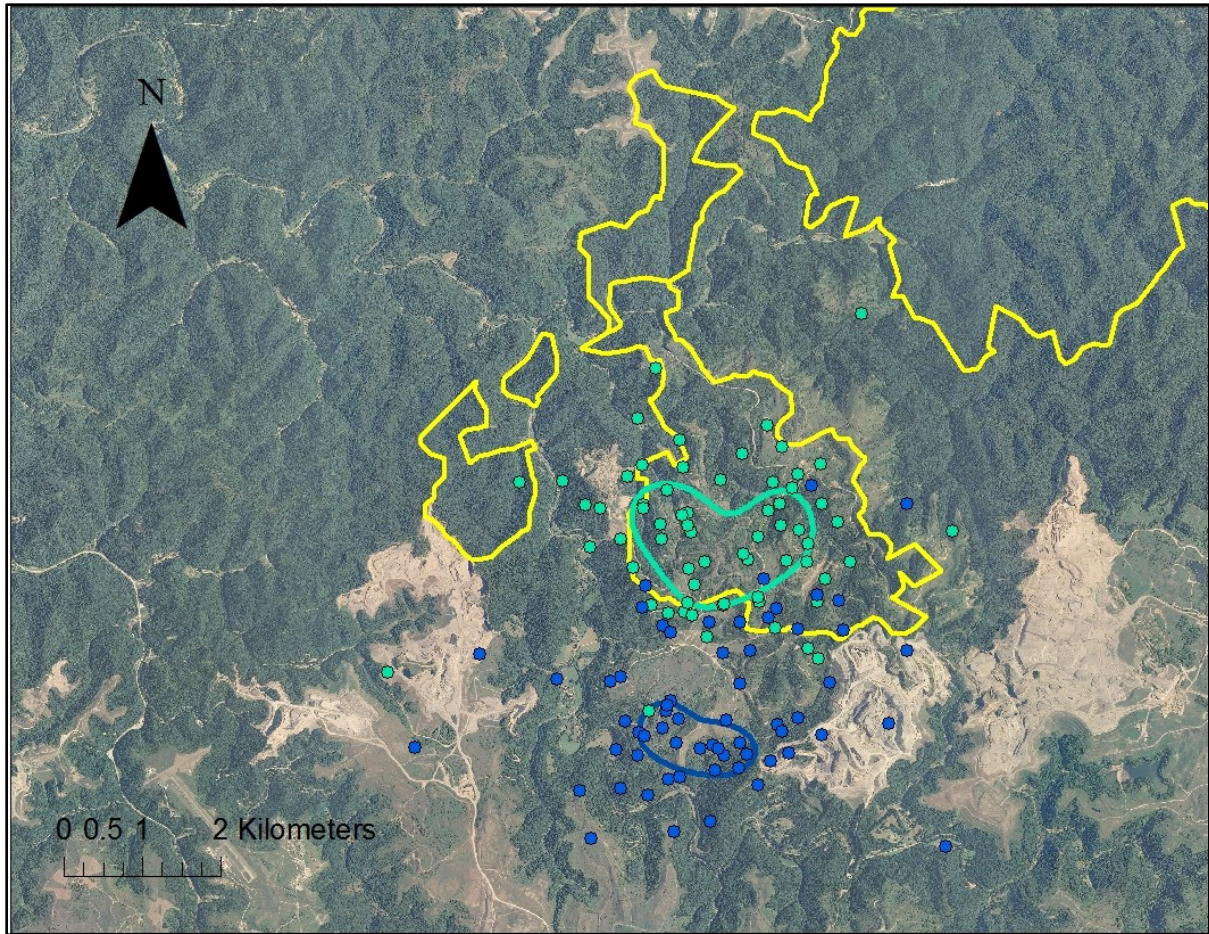


Figure A-60. The 25% adaptive kernel core area overlap for F1 and F4 with the associated locations.

F1 in teal and F4 in dark blue; the wildlife management area and Robinson Forest boundaries are represented in yellow.

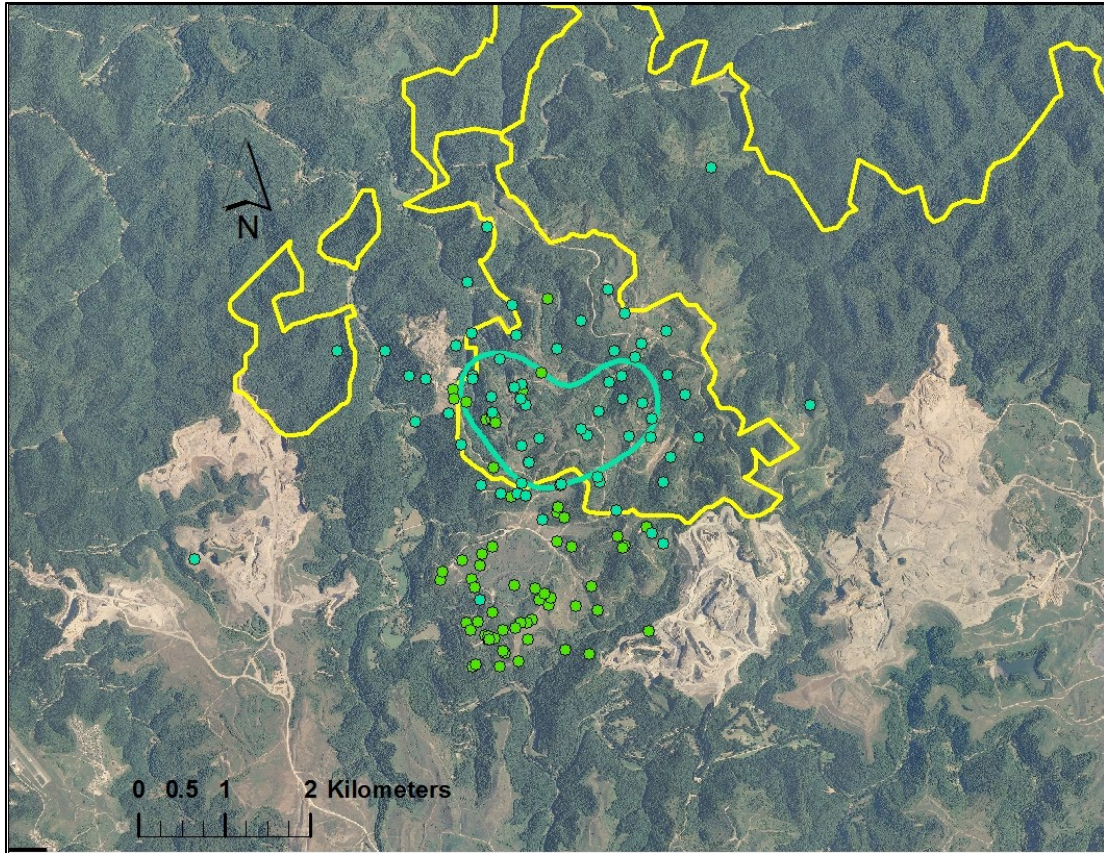


Figure A-61. The 25% adaptive kernel core area overlap for F1 and M1 with the associated locations.

F1 in teal and M1 in bright green; the wildlife management area and Robinson Forest boundaries are represented in yellow.

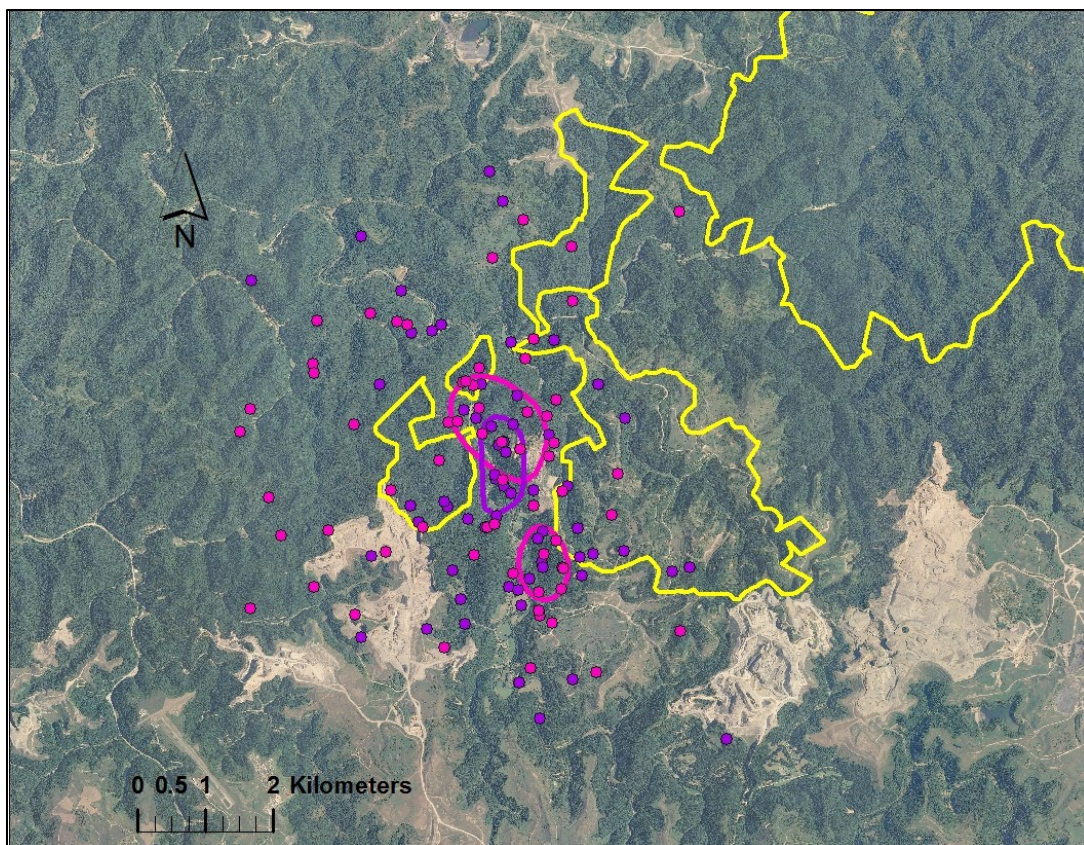


Figure A-62. The 25% adaptive kernel core area overlap for F2 and F3 with the associated locations.

F2 in purple and F3 in dark pink; the wildlife management area and Robinson Forest boundaries are represented in yellow.

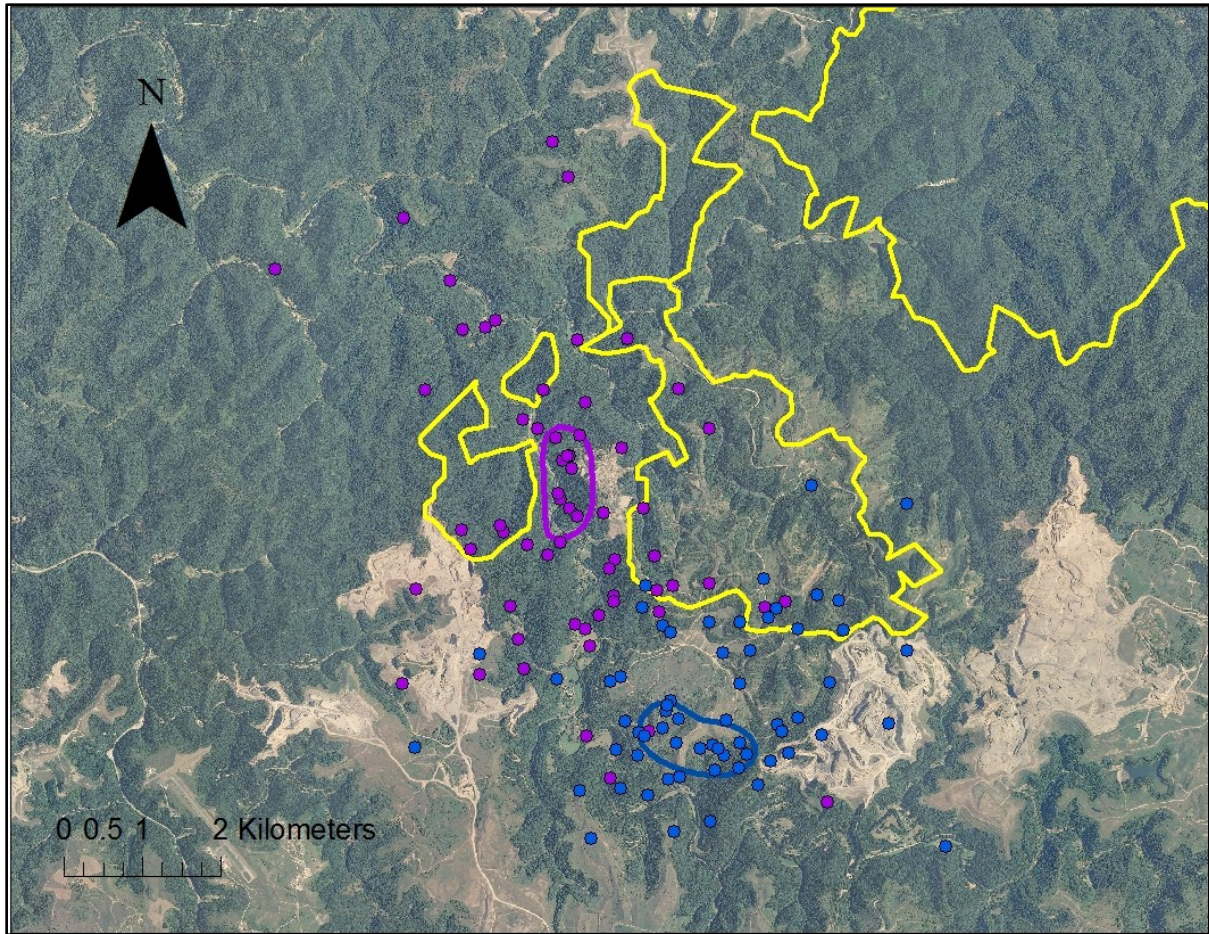


Figure A-63. The 25% adaptive kernel core area overlap for F2 and F4 with the associated locations.

F2 in purple and F4 in dark blue; the wildlife management area and Robinson Forest boundaries are represented in yellow.

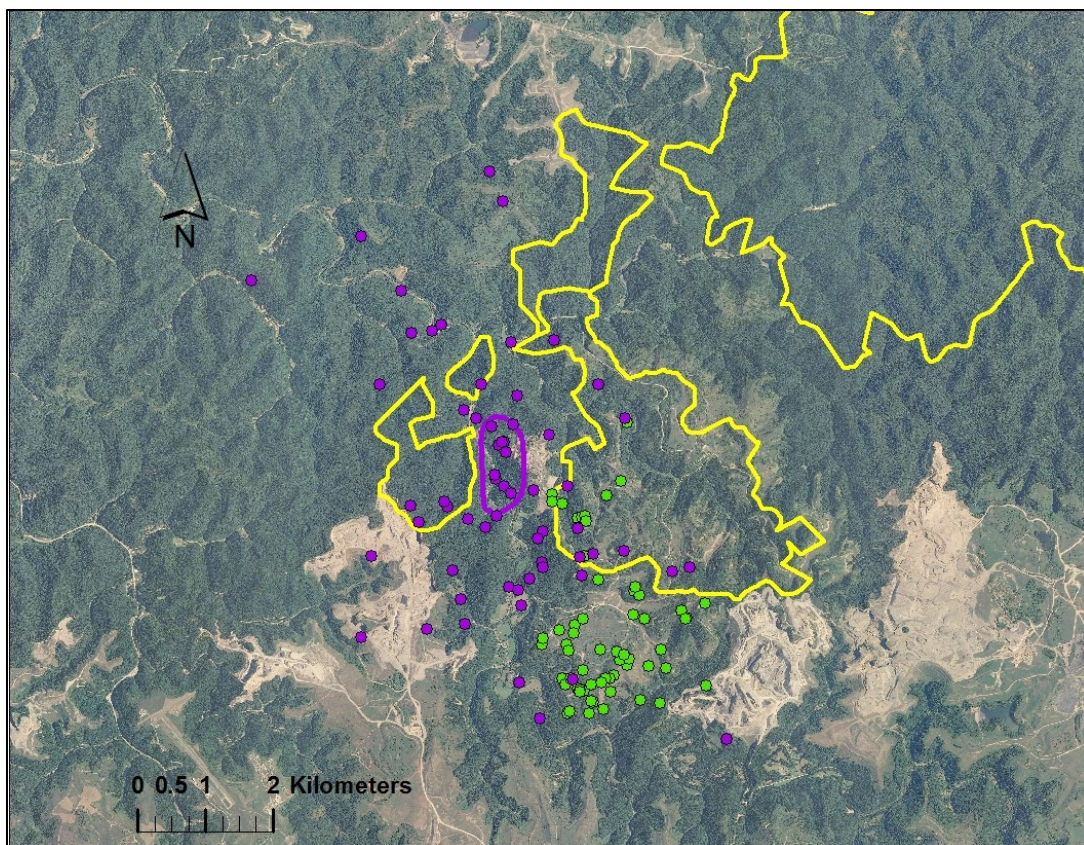


Figure A-64. The 25% adaptive kernel core area overlap for F2 and M1 with the associated locations.

F2 in purple and M1 in bright green; the wildlife management area and Robinson Forest boundaries are represented in yellow.

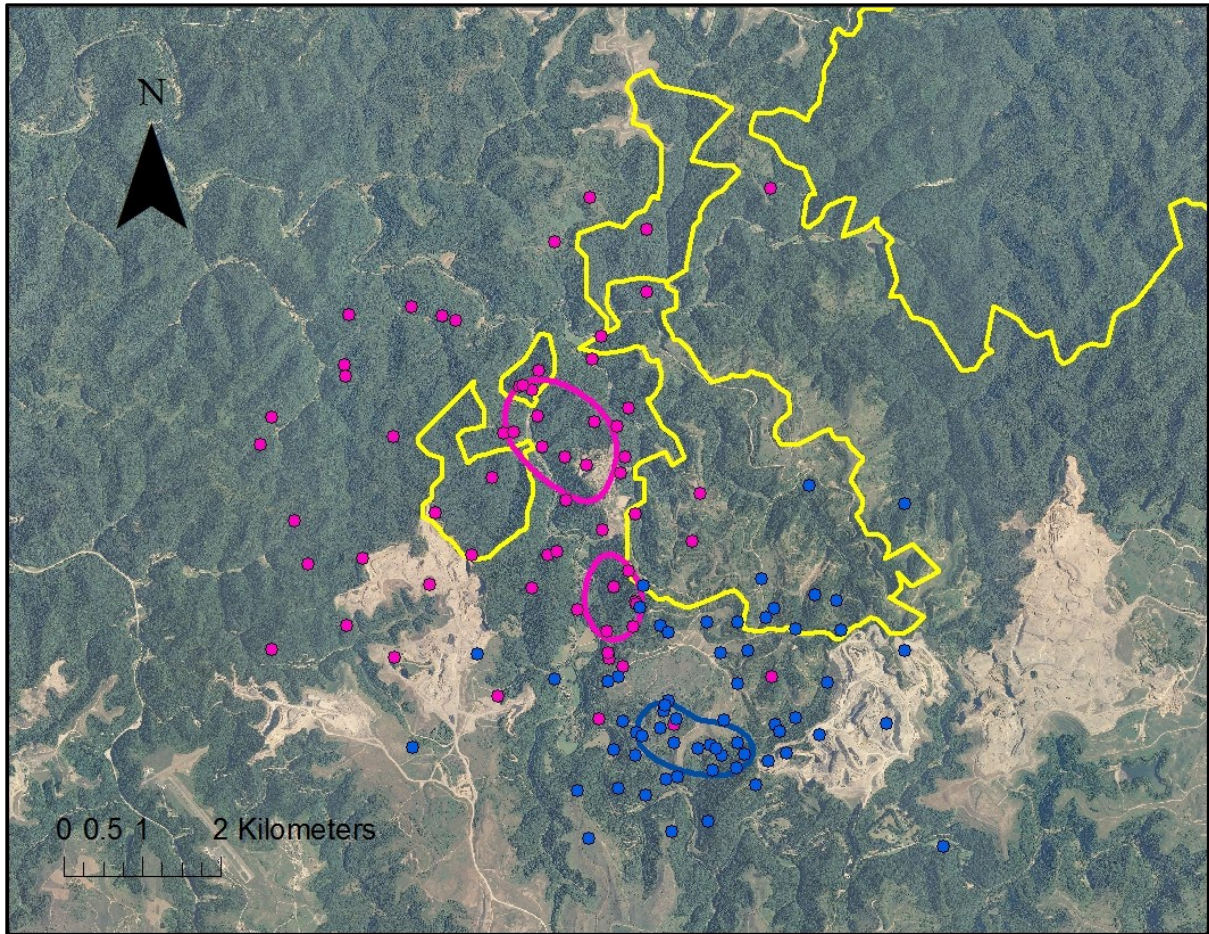


Figure A-65. The 25% adaptive kernel core area overlap for F3 and F4 with the associated locations.

F3 in dark pink and F4 in dark blue; the wildlife management area and Robinson Forest boundaries are represented in yellow.

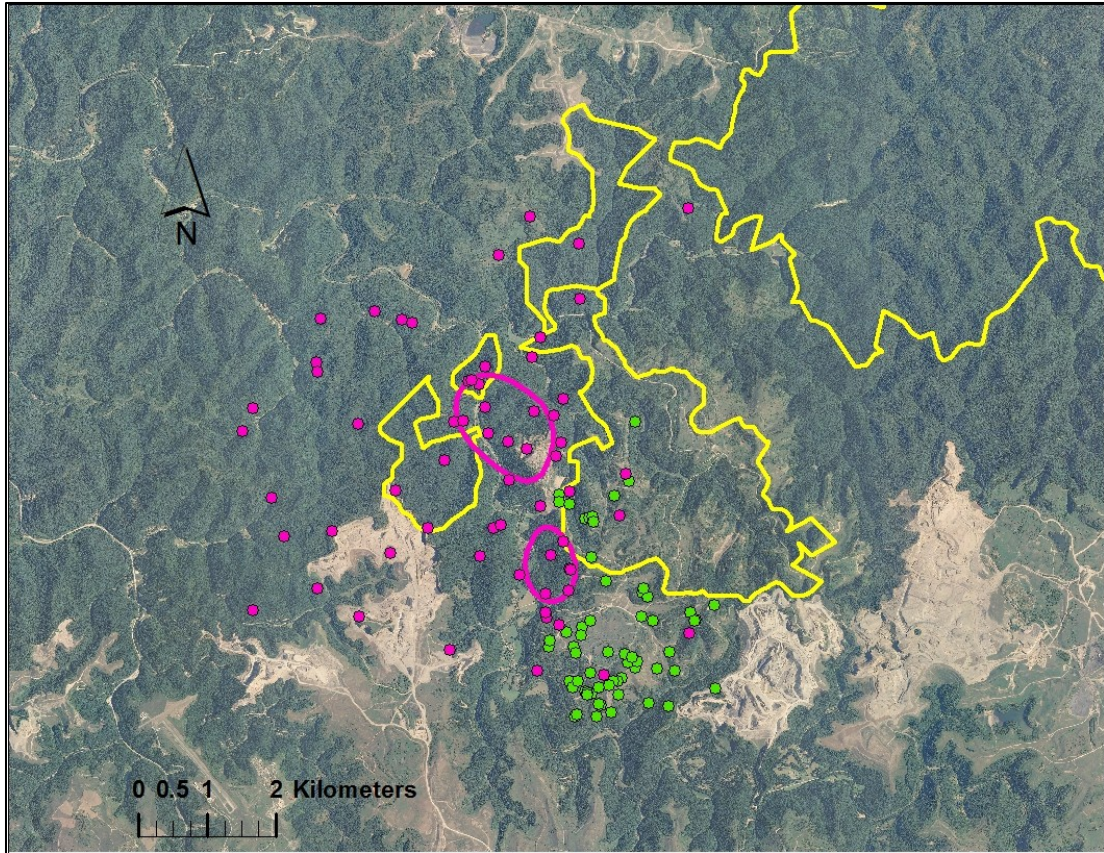


Figure A-66. The 25% adaptive kernel core area overlap for F3 and M1 with the associated locations.

F3 in dark pink and M1 in bright green; the wildlife management area and Robinson Forest boundaries are represented in yellow.

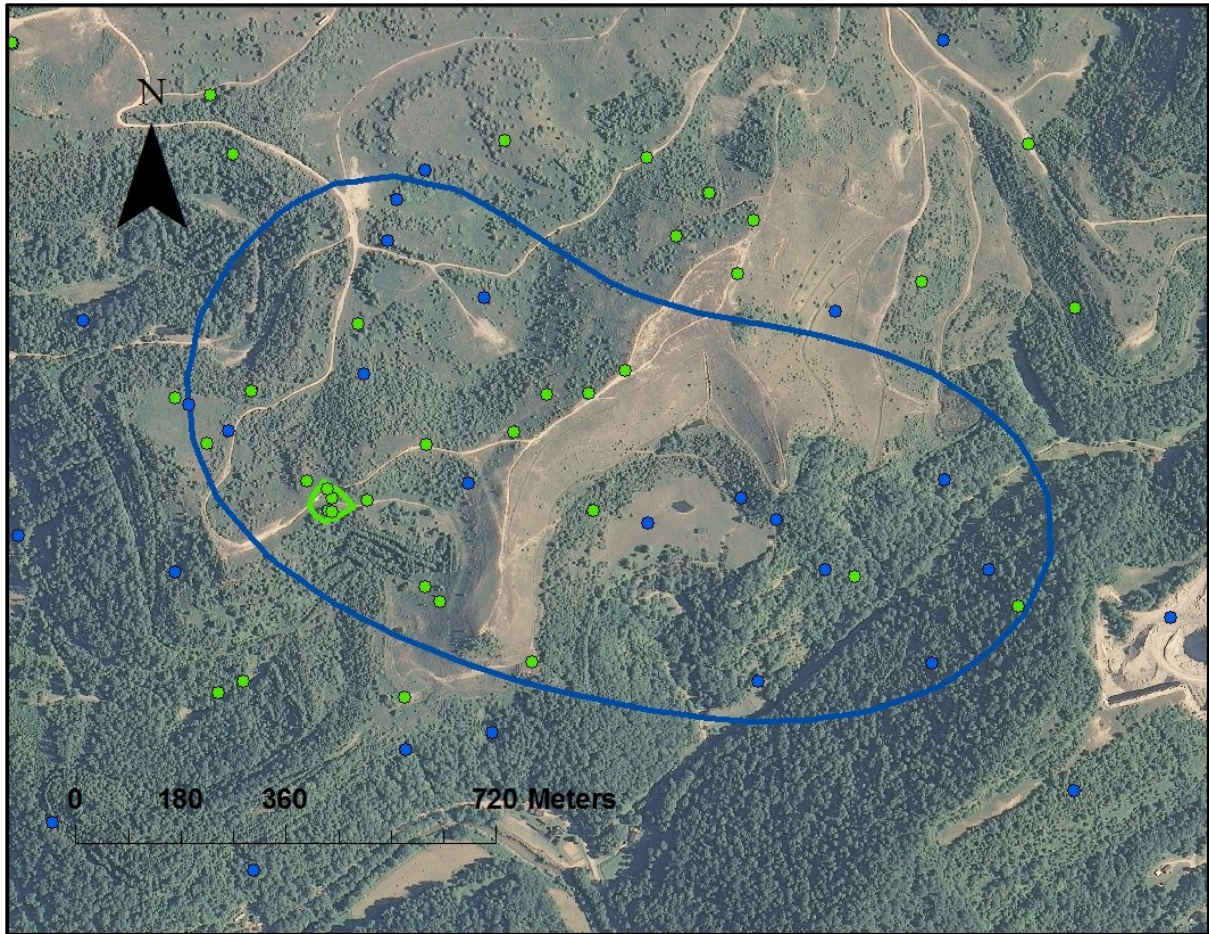


Figure A-67. The 25% adaptive kernel core area overlap for F4 and M1 with the associated locations.

F4 in dark blue and M1 in bright green; the wildlife management area and Robinson Forest boundaries are represented in yellow.

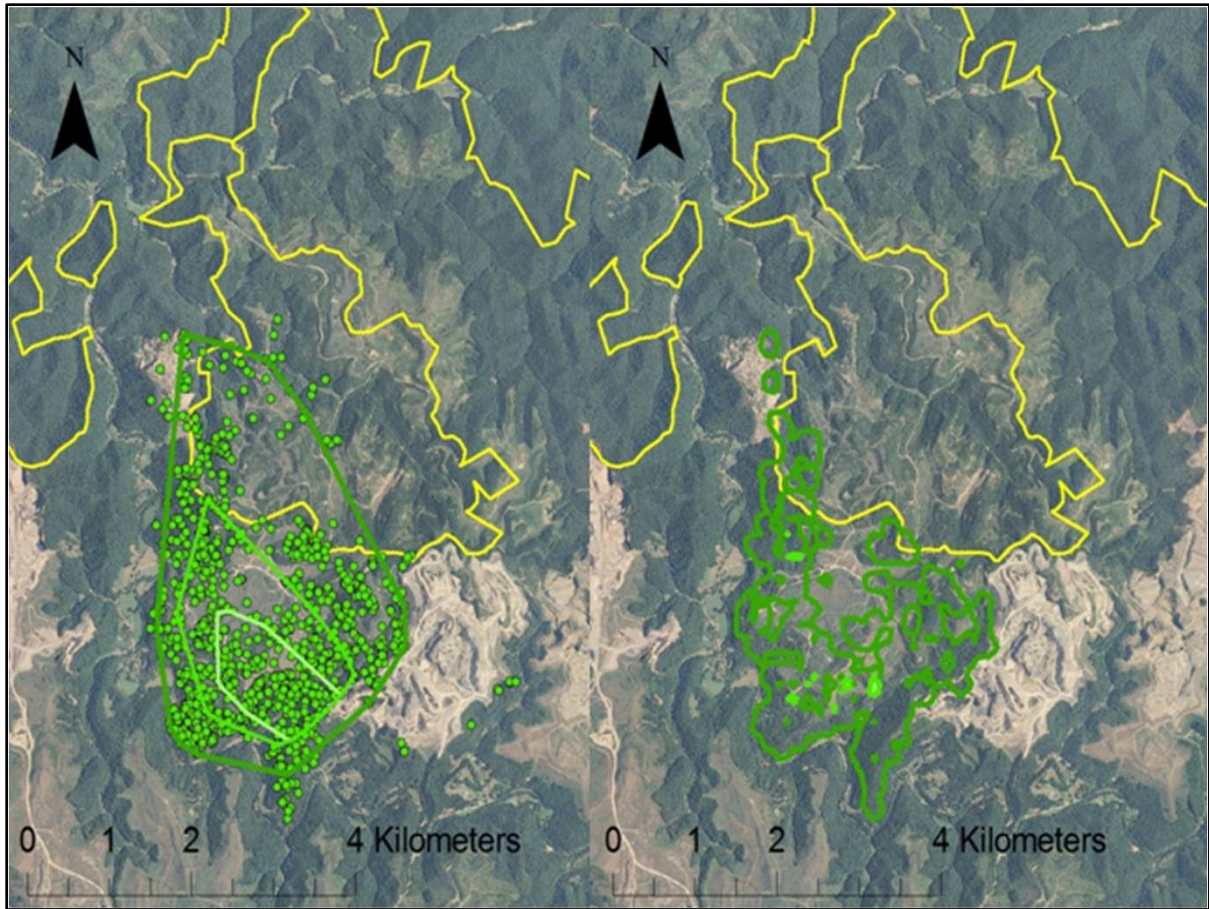


Figure A-68. The 95%, 50%, and 25% annual minimum convex polygons (at left) and adaptive kernels (at right) for M1.

Each scale is in a progressively lighter shade of green, with the associated locations, the wildlife management area and Robinson Forest boundaries are represented in yellow.

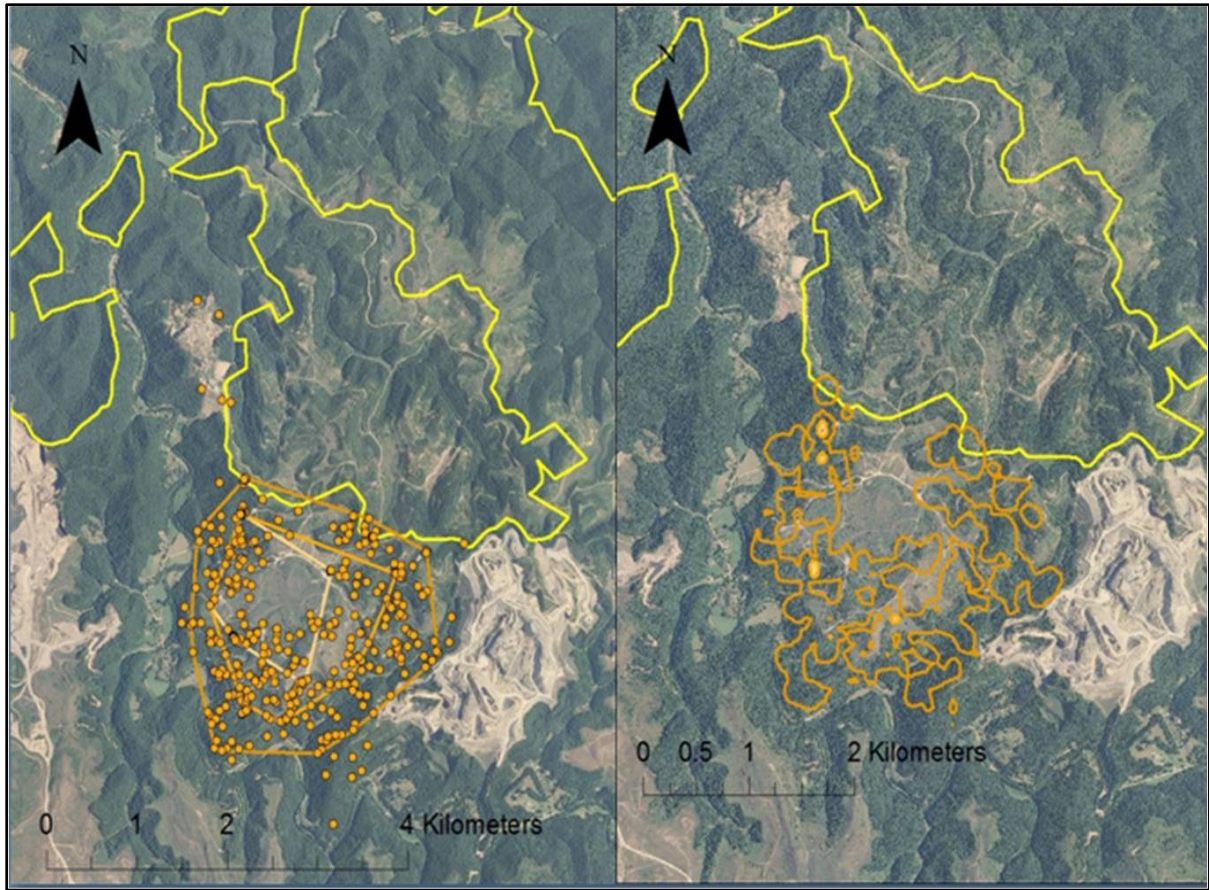


Figure A-69. The 95%, 50%, and 25% minimum convex polygons (at left) and adaptive kernels (at right) for M1.

Each scale is in a progressively lighter shade of orange, with the associated locations for the fall season, the wildlife management area and Robinson Forest boundaries are represented in yellow.

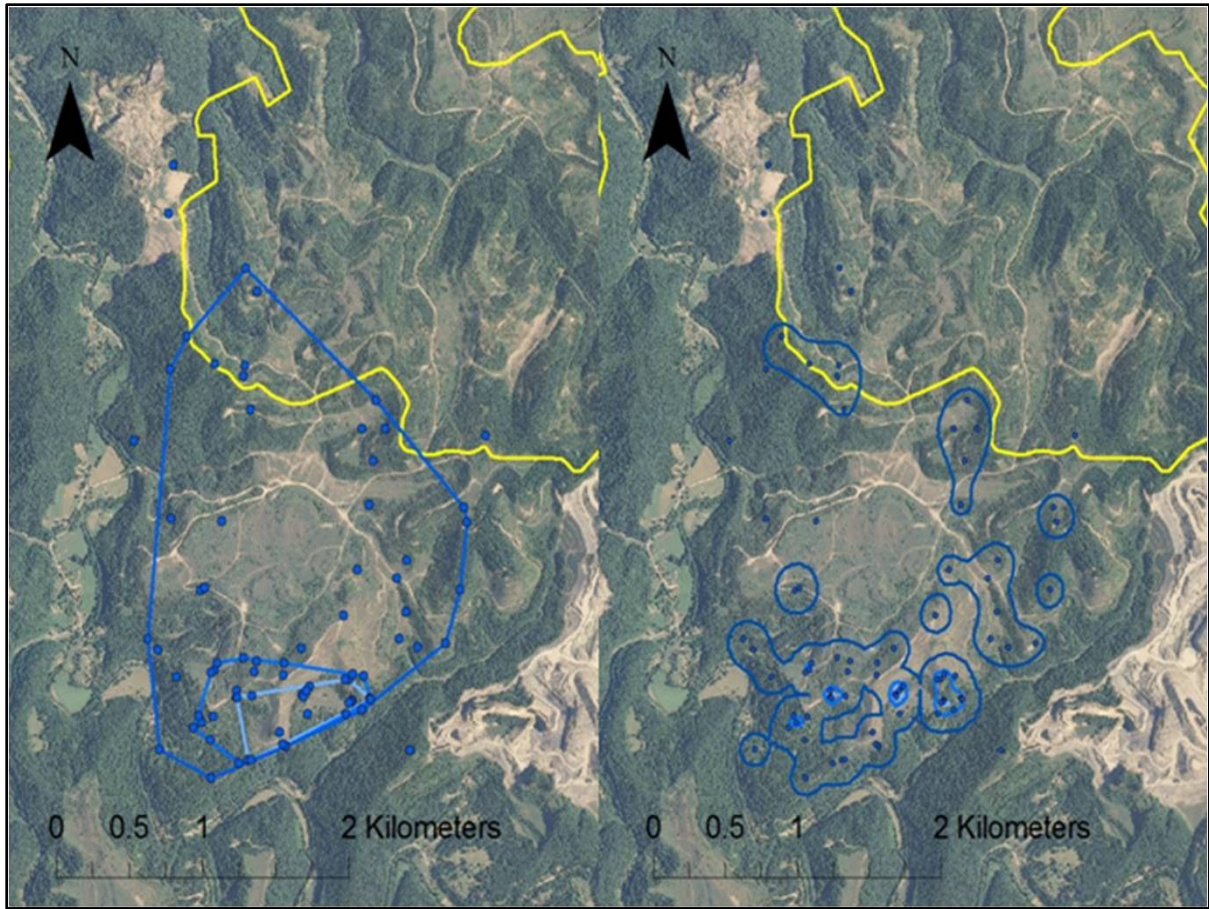


Figure A-70. The 95%, 50%, and 25% minimum convex polygons (at left) and adaptive kernels (at right) for M1.

Each scale is in a progressively lighter shade of blue, with the associated locations for the winter season, the wildlife management area and Robinson Forest boundaries are represented in yellow.

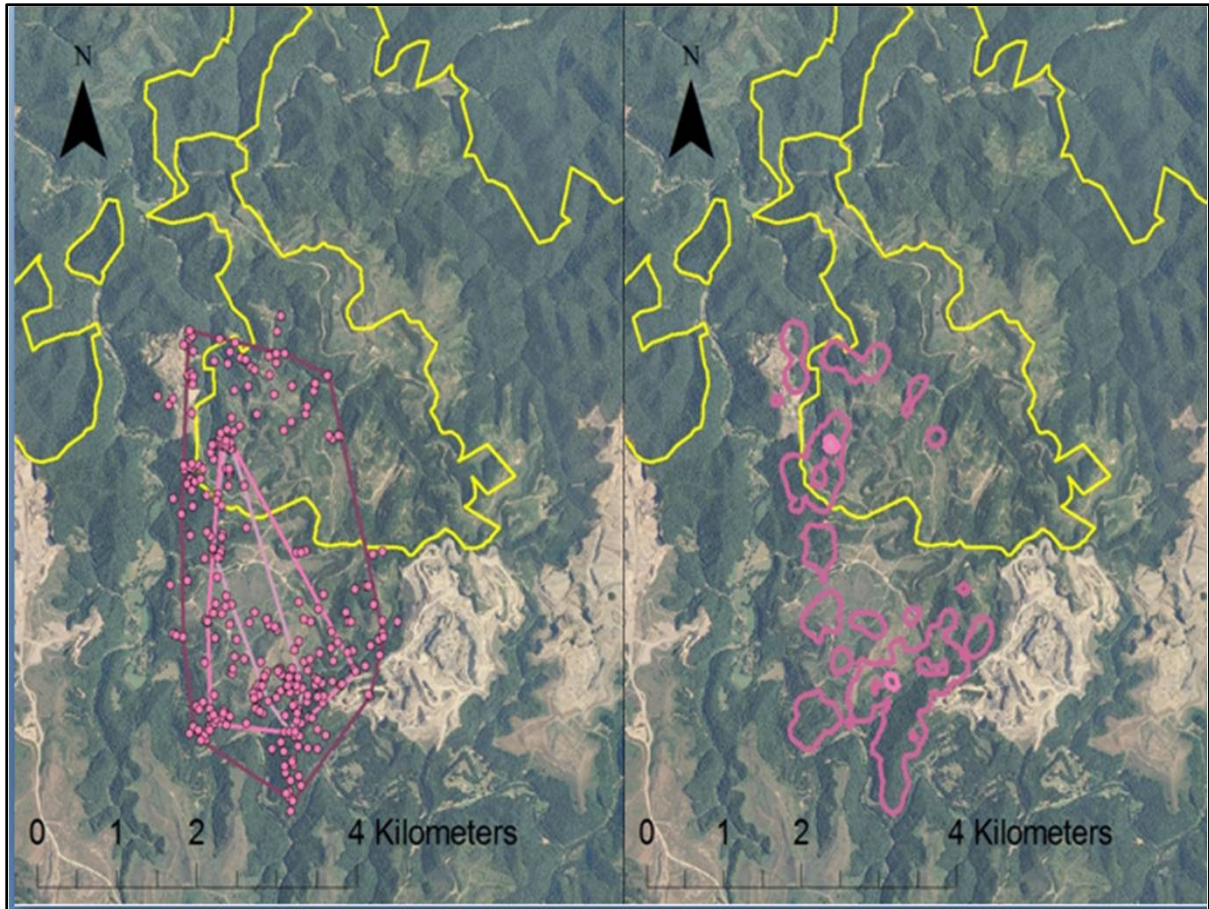


Figure A-71. The 95%, 50%, and 25% minimum convex polygons (at left) and adaptive kernels (at right) for M1.

Each scale is in a progressively lighter shade of pink, with the associated locations for the spring season, the wildlife management area and Robinson Forest boundaries are represented in yellow.

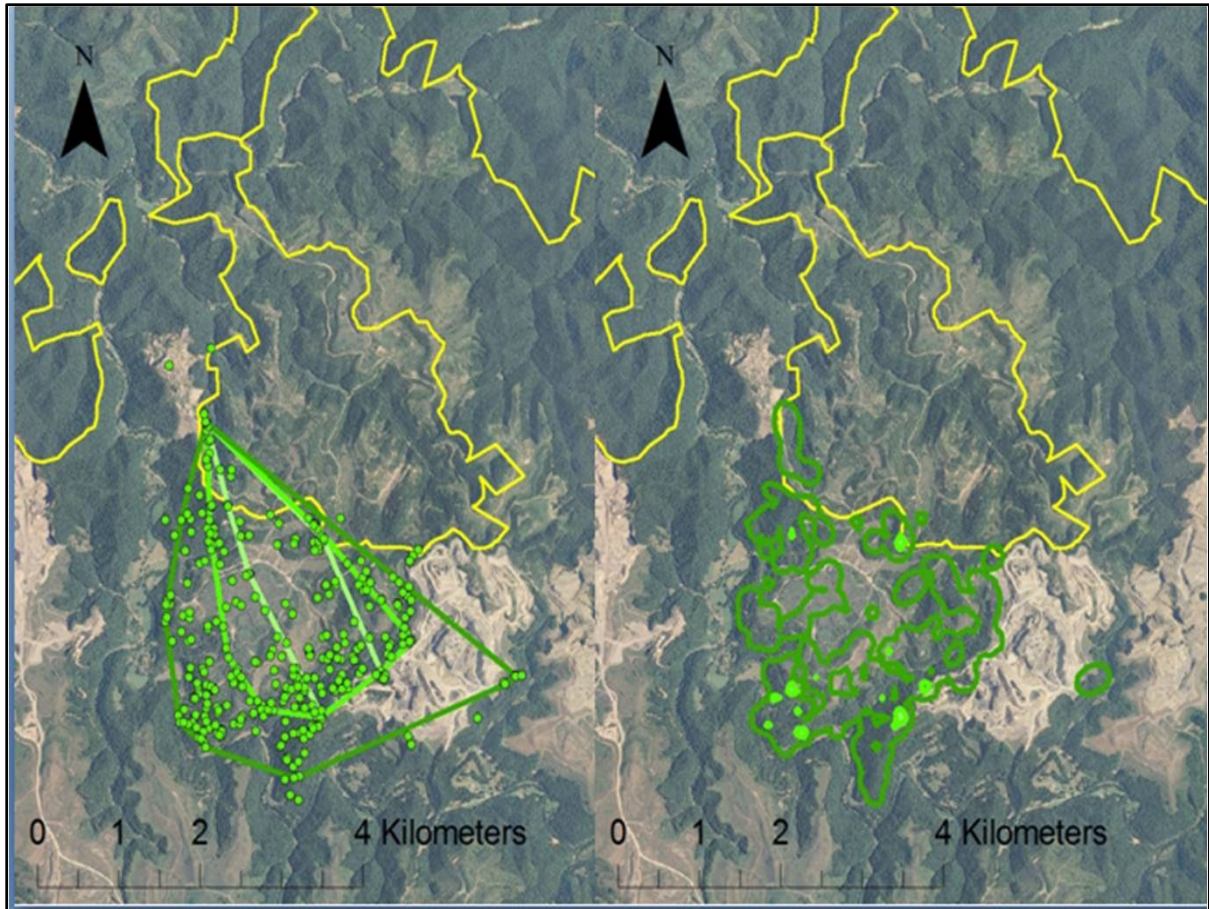


Figure A-72. The 95%, 50%, and 25% minimum convex polygons (at left) and adaptive kernels (at right) for M1.

Each scale is in a progressively lighter shade of green, with the associated locations for the summer season, the wildlife management area and Robinson Forest boundaries are represented in yellow.

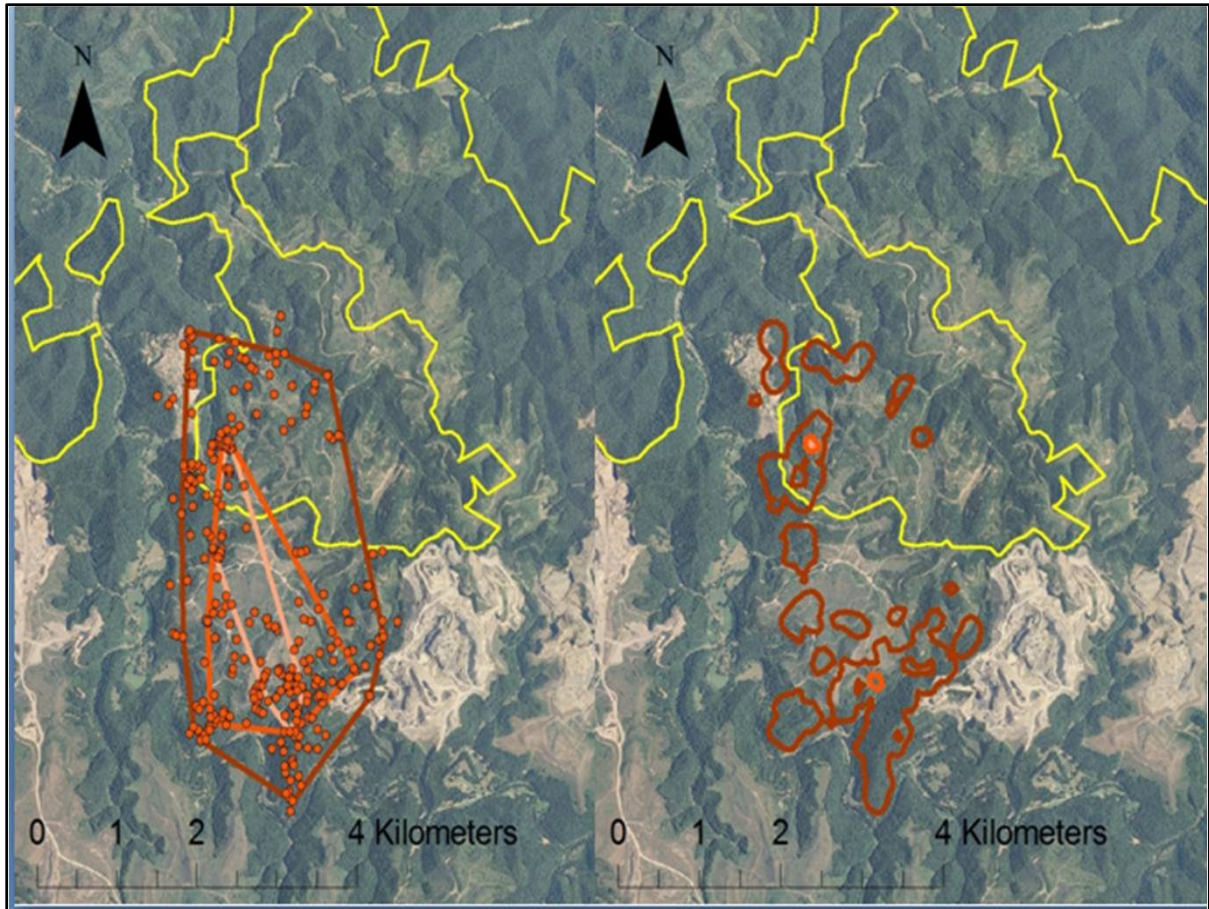


Figure A-73. The 95%, 50%, and 25% minimum convex polygons (at left) and adaptive kernels (at right) for M1.

Each scale is in a progressively lighter shade of brown, with the associated locations for the breeding season, the wildlife management area and Robinson Forest boundaries are represented in yellow.

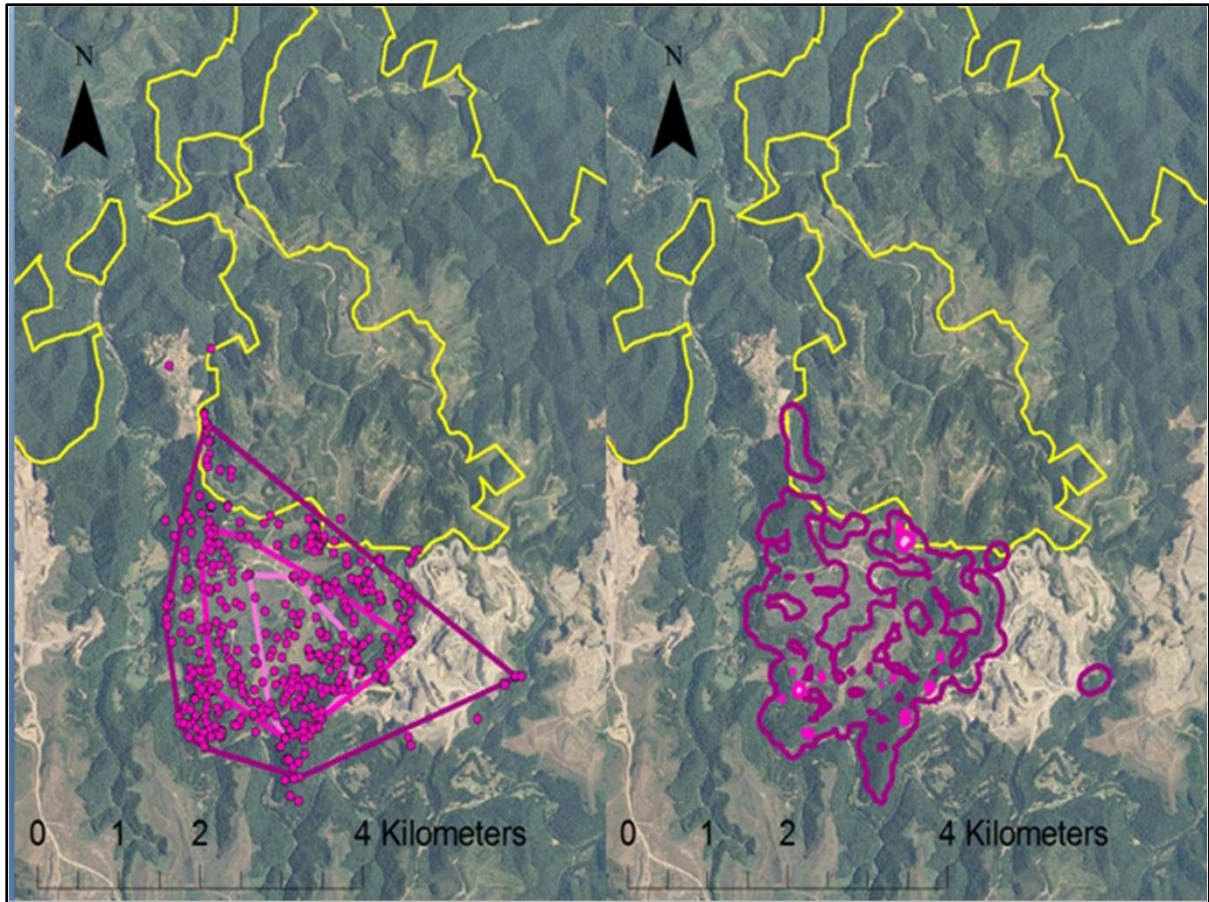


Figure A-74. The 95%, 50%, and 25% minimum convex polygons (at left) and adaptive kernels (at right) for M1.

Each scale is in a progressively lighter shade of pink, with the associated locations for the rearing season, the wildlife management area and Robinson Forest boundaries are represented in yellow.

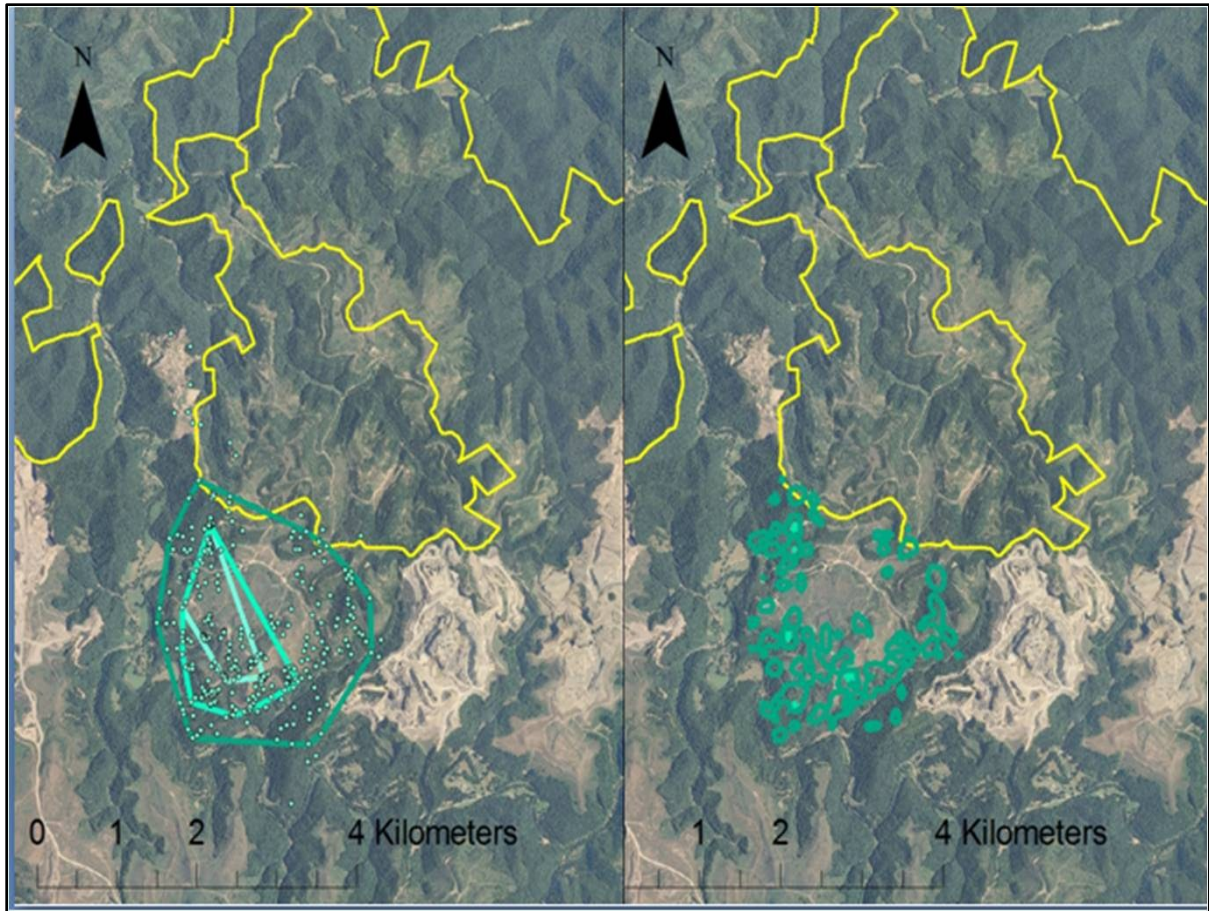


Figure A-75. The 95%, 50%, and 25% minimum convex polygons (at left) and adaptive kernels (at right) for M1.

Each scale is in a progressively lighter shade of blue, with the associated locations for the 3-season-winter season, the wildlife management area and Robinson Forest boundaries are represented in yellow.

VITA

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