

POPULATION DYNAMICS AND HABITAT ECOLOGY  
OF BLACK BEARS IN THE OUACHITA  
MOUNTAINS OF OKLAHOMA

By

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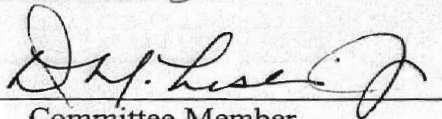
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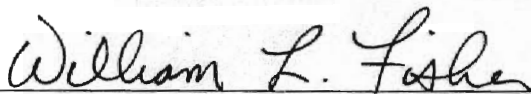
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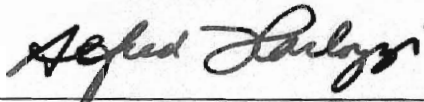
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## CHAPTER I

### POPULATION DYNAMICS OF BLACK BEARS IN THE OUACHITA MOUNTAINS OF OKLAHOMA

#### INTRODUCTION AND JUSTIFICATION

Understanding how wildlife populations expand to recolonize former habitats is important to management and conservation (Swenson et al. 1998). Population range expansion is influenced by individual dispersal, population growth, and variation in environmental factors (Lubina and Levin 1988). Dispersal is 1-way movement with no predetermined direction in the population as a whole (Sinclair 1986). These movements are distinct from emigration (directional 1-way movement; Sinclair 1992) and migration (regular round-trip traveling; Stenseth and Lidicker 1992). Factors that influence dispersal include competition for mates, avoidance of inbreeding, and competition for resources (Sinclair 1992).

Presaturation dispersal (occurs when ecological carrying capacity has not been reached), saturation dispersal (occurs when ecological carrying capacity has been reached), and natal dispersal (occurs before individuals reach breeding age; Sinclair 1992) also are important. Large mammals are similar to non-cycling rodent populations in that there is little evidence for presaturation dispersal (Sinclair 1992), although it has been reported in badgers (*Meles meles*; Cheeseman et al. 1988), roe deer (*Capreolus capreolus*; Wahlstrom and Liberg 1995), and brown bears (*Ursus arctos*; Swenson et al. 1998). Badger and roe deer populations exhibiting presaturation dispersal had low

densities and positive growth rates (Cheeseman et al. 1988, Wahlstrom and Liberg 1995). In an expanding wolf (*Canis lupus*) population in northwestern Montana and southeastern British Columbia, the sex ratio was female-biased, and survival rates of pups and adults were high compared with other wolf populations (Pletscher et al. 1997).

Dispersal patterns may differ between established continuous populations and those at the edge of an expanding population (Swenson et al. 1998). Availability of space and food resources, which significantly influences bear (*Ursus*) movement and female reproduction, may be greater in areas supporting expanding populations (Beckmann and Berger 2003). Subadult males disperse shorter distances in expanding populations because territory unoccupied by adult males is more abundant in peripheral areas (Swenson et al. 1998). In continuous populations, subadult males must disperse farther to reach unoccupied territory. Adult females are likely to shift their home ranges to allow space for female offspring (Rogers 1987) in an expanding population where space is limited. However, female offspring may disperse greater distances in a continuous population where competition for resources is high.

Core and peripheral areas are more distinct in expanding populations than in continuous populations. In an expanding population of brown bears (*Ursus horribilis*), total bear density decreased from core areas to peripheral areas (Swenson et al. 1998). Sex ratio in core areas was female-biased and in peripheral areas was male-biased (Swenson et al. 1998). Age structure between core and peripheral areas also differed for males (there were not enough data collected to detect differences in female age structure); more subadult males were harvested in peripheral areas, whereas more adults were harvested in core areas (Swenson et al. 1998).

The geographic range of American black bears (*U. americanus*) historically included the Ozark and Ouachita mountains of Arkansas and Oklahoma (Clark and Smith 1994). Loss of habitat and unregulated hunting by settlers in the late 19<sup>th</sup> century resulted in their decline and extirpation from Oklahoma by 1915 (Smith and Clark 1994). Arkansas Game and Fish Commission (AGFC) closed the bear-hunting season in Arkansas in 1927, when the only population remaining in Arkansas by the 1940s was located near White River National Wildlife Refuge (Oli et al. 1997, Clark and Smith 1994). Between 1958 and 1968, AGFC captured about 250 black bears (*U. a. americanus*) in northern Minnesota and Manitoba, Canada, and released them in the Ozark and Ouachita mountains (Interior Highlands) of Arkansas (Clark and Smith 1994). Translocations were successful in reestablishing black bear populations in Arkansas, and AGFC reopened the bear-hunting season in 1980 (Clark and Smith 1994, Smith and Clark 1994). Black bear populations have continued to increase, and in autumn 2001, AGFC allowed baiting on private lands in selected hunting zones (R. Eastridge, AGFC, personal communication).

Distribution and abundance of black bears in the Ouachita Mountains of southeastern Oklahoma are expanding as the species recolonizes its former range. Visitation by black bears at bait stations increased since initiation of bait-station lines in 1989-1997 (Skeen 1997). Since 1997, bait-station visitation by bears has fluctuated and has decreased slightly in recent years (J. Hemphill, Oklahoma Department of Wildlife Conservation [ODWC], personal communication). In addition, the number of bears causing nuisance activity (e.g., crop damage) in Oklahoma has increased since the late 1980s (J. Hemphill, ODWC, personal communication). The ODWC considers the

Ouachita National Forest critical to supporting Oklahoma's black bear population. However, little is known about the ecology or population dynamics of this species in Oklahoma. Information on demographic characteristics of the black bear population centered in the Ouachita Mountains of Oklahoma is essential to ODWC in developing a statewide management plan for bears.

## **OBJECTIVES AND PREDICTIONS**

My objective was to assess demographic characteristics of the black bear population in the Ouachita National Forest of Oklahoma. These data serve 2 purposes: to provide essential background information for effective management of black bears in Oklahoma and to provide insight into characteristics of a colonizing population of large carnivores. Based on demography of other colonizing carnivores (Pletscher et al. 1997, Swenson et al. 1998), I predicted a female-biased adult sex ratio and an age distribution skewed to young individuals; high female and cub survival rates; and high female density in my study area.

## **MATERIALS AND METHODS**

### **Study Area**

I conducted the study in the Kiamichi and Choctaw districts of the Ouachita National Forest, LeFlore County, southeastern Oklahoma, in a 406-km<sup>2</sup> study area (Fig. I.1). East-west ridges characterize the Ouachita Mountains, with elevations from 400-813 m. Climate consisted of mild winters (average January temperature: 3.9°C) and hot, humid summers (average July temperature: 27.7°C); however, temperatures were lower in high elevations (Oklahoma Climatological Survey 2003). LeFlore County averaged 122 cm of rainfall annually (Oklahoma Climatological Survey 2003).

Rolley and Warde (1985) described three main cover types for the area: pine (*Pinus* spp.) forests (primarily on south-facing slopes), deciduous forests (primarily on north-facing slopes and creek bottoms), and mixed pine-deciduous forests. Pine forests were characterized by an overstory dominated by shortleaf pine (*P. echinata*); a midstory of winged elm (*Ulmus alata*), sparkleberry (*Vaccinium arboreum*), and low blueberry (*V. vacillans*); and an understory of greenbriar (*Smilax* spp.), poison ivy (*Rhus radicans*), and little bluestem (*Andropogon scoparius*). Deciduous forests included an overstory dominated by oaks (*Quercus* spp.) and hickories (*Carya* spp.); a midstory of flowering dogwood (*Cornus florida*), eastern redbud (*Cercis canadensis*), red maple (*Acer rubrum*), and St. Johnswort (*Hypericum* spp.); and an understory of sparglegrass (*Chasmanthium* spp.), panicum (*Panicum* spp.), and wildrye (*Elymus* spp.). Mixed pine-deciduous forests occurred primarily at lower elevations in transition zones between pine forests and deciduous forests (Rolley and Warde 1985).

### **Capture and Handling**

Black bears were captured by Aldrich spring-activated snares modified for bear safety (Johnson and Pelton 1980) and barrel traps in May–August and October–November, 2001–2002. I anesthetized most bears (trap:  $n = 66$ ; den:  $n = 28$ ) with Telazol (A.H. Robins Company, Richmond, Virginia), a combination of tiletamine hydrochloride and zolazepam hydrochloride, at a dosage rate of 4.8 mg/kg (Doan-Crider and Hellgren 1996). Seven bears were tranquilized with a 2:1 mixture of ketamine-xylazine (Clark and Smith 1994) at a rate of 6.6 mg/kg. Drugs were administered by jabstick (Clark 1991). Captured bears were marked with plastic ear tags and lip tattoos bearing corresponding identification numbers (Nasco, Fort Atkinson, Wisconsin). Adult females ( $\geq 36$  kg) were

fitted with radiocollars equipped with mortality sensors (Telonics, Mesa, Arizona). All collars included a cotton spacer (Hellgren et al. 1988). A first premolar tooth was extracted from each captured bear for age estimation by cementum annuli analysis at a commercial laboratory (Matson's Laboratory, Milltown, Montana). Captured bears were measured and weighed ( $\pm 1$  kg).

Blood samples were collected by femoral venipuncture in Vacutainer tubes as soon as possible following immobilization. Each sample consisted of  $\geq 2$  10-mL tubes without anticoagulant for serum analyses (Hellgren et al. 1989) and 1 7-mL tube containing EDTA coagulant for genetic analysis. Samples were centrifuged, serum harvested, and samples stored at  $-20$  C for later analysis. Tubes containing EDTA also were centrifuged, white blood cells pipetted into a storage tube, and the storage tube frozen. Tissue samples were collected from ears of captured bears using disposable biopsy punches. Samples were stored in buffer solution out of direct sunlight for later analysis.

Heart rate, respiration, and temperature of captured animals were monitored during the handling process. Immobilized bears were placed in the shade with cloth covering their eyes at the conclusion of sampling procedures. If vital signs were stable and normal, handling personnel vacated the area to permit bear recovery. After 4–12 h, personnel returned to the site to verify that the bear had left the area.

Radiocollared females were tracked to winter dens in February–March 2002–2003 to determine presence of cubs or yearlings. If possible, females and yearlings were sedated as described above, and cotton spacers on radiocollars were replaced. One female was sedated with a dart pistol in 2002. Measurements and blood samples from



adult and yearling bears were collected if bears could be removed from dens. Cubs were sexed, measured, and weighed, and tissue samples were collected and preserved as described above. I tagged cubs with passive integrated transponder (PIT) tags (Biomark, Boise, Idaho) injected subcutaneously between the scapulae for future identification. Vital signs of sedated animals were monitored regularly, and yearlings and cubs were placed next to females in dens at the conclusion of the procedure. Personnel returned to the den area the following day to verify that sedated bears had recovered (via radio telemetry).

### **Demographic Characteristics**

*Population Composition*—Sex ratio of adults and cubs was calculated from the capture sample and from observations of newborn cubs in dens, respectively. I used Chi-square tests to determine if sex ratios of adults and cubs differed from 50:50 (Doan-Crider and Hellgren 1996).

I examined age structure using the captured-bear sample. Age classes were yearlings, subadults (2–3-yr-old), and adults ( $\geq$ 4-yr-old). Because of the non-normal distribution of the age structure, I used a Wilcoxon rank sum test to determine differences in median ages between males and females (Doan-Crider and Hellgren 1996). I used a Chi-square test to determine if age distribution of males and females differed.

*Reproduction and Survival*—Reproductive characteristics (i.e., litter size and fecundity) were estimated using data from monitoring radiocollared adult females and visiting dens of females in winter. Mean litter sizes of newborn cubs and yearlings were determined by observation of family groups in dens. Fecundity was calculated as the average litter size multiplied by the proportion of female newborn cubs divided by the

interbirth interval (Lancia et al. 1994). Interbirth interval was assumed to be 2 years based on observation of bears in my study and literature (Bunnell and Tait 1981, Powell et al. 1997).

Adult female survival is the dominant factor affecting population growth in long-lived, iteroparous species such as bears (Eberhardt 1990). To understand the population dynamics of the black bear population in the Ouachita Mountains, it is essential to estimate survival rates. Survival rates were estimated using Kaplan-Meier Limit Estimator (K-MLE) with the staggered-entry design (Pollock et al. 1989). K-MLE requires radiocollared individuals and allows newly collared individuals to be added to the sample at any time. Radiocollared individuals were censored (i.e., removed from the analysis) if radio contact was lost for  $\geq 14$  days or collars were dropped (Kasworm and Thier 1994). No radiocollared females died during the study; however, 1 individual's collar was found and she possibly was harvested illegally. Two survival estimates were generated; the first censored that female as with other dropped collars, and the second included her as a mortality.

Several assumptions exist for K-MLE. First, all animals of a particular cohort (e.g., sex or age class) have been sampled randomly. Second, survival times are independent for different animals. This assumption does not cause bias, but does make estimates appear to have smaller variances. I placed radiocollars on adult females to reduce possible violation of this assumption. Third, capturing the animal or having it carry a radiocollar does not influence its future survival. Fourth, the censoring mechanism is random or not related to an animal's fate. Fifth, newly tagged animals have the same survival function as previously tagged animals (Pollock et al. 1989).

*Density and Population Growth*—Population abundance ( $N_C$ ) was estimated using the Lincoln-Petersen (L-P) equation with Chapman Correction (Clark 1991):

$$N_C = [(n_1+1)(n_2+1)/(m_2+1)] - 1$$

where  $n_1$  = total captured, marked, and released in first sampling period;  $n_2$  = total captured, marked, and released in second sampling period; and  $m_2$  = number marked individuals captured in second sampling period. Population density was estimated by dividing  $N_C$  by study-area size. The variance equation for L-P estimator was

$$\text{var}(N_2) = [(n_1+1)(n_2+1)(n_1-m_2)(n_2-m_2)] / [(m_2+1)^2(m_2+2)].$$

This is a closed model and assumptions are negligible births, deaths, emigration, or immigration; equal catchability among individuals; and identification marks are not lost, gained, or overlooked. If the probability of emigration is equal for marked and unmarked individuals, the assumption of no emigration can be relaxed. To minimize the effect of births and deaths, yearlings captured during the second sampling period and bears known to have died during the interval between the 2 periods were not included in the analysis. Although the closed population assumption may have been violated because sampling efforts spanned 2 years, an open population model such as the Jolly-Seber could not be used because I had only 2 sampling periods (Lancia et al. 1994). Capture heterogeneity due to sex and age likely occurs in the Oklahoma black bear population (Lancia et al. 1994). Response to trapping pressure may increase (“trap-happy” individuals) or decrease (“trap-shy” individuals) capture probabilities (Lancia et al. 1994). I used a double-marking scheme to avoid violations of the tag-loss assumption (Lancia et al. 1994). Because of the possible violations of the assumptions of the L-P

estimator, population abundance and density estimates should be interpreted conservatively.

Study-area size was determined using 2 methods. First, I created a 95% minimum convex polygon for all radiolocations of adult females used in home-range analyses. Second, I buffered the 100% minimum convex polygon for trapping locations with the approximate radius of the average 95% minimum convex polygon home range of adult females ( $n = 13$ ) using ArcView (ESRI, Redlands, California). The resulting polygons were used to delineate the study-area boundary.

An age-structured, 5 x 5 Leslie matrix was used to calculate a stable age distribution to approximate population growth rate ( $\lambda$ ). Assumptions associated with the Leslie matrix are a closed population, no genetic variation in population affecting birth and death rates, and continuous growth with no time lags (Gotelli 1995). Vital rates included in the matrix were fecundity (F), proportion of cubs surviving to become yearlings ( $P_{CY}$ ) in the second sampling period, proportion of female yearlings surviving to become 2 yr-olds ( $P_{Y2}$ ) in the second sampling period, proportion of female 2 yr-olds that become 3 yr-olds ( $P_{23}$ ) in the second sampling period, proportion of female 3 yr-olds that become adults ( $P_{3A}$ ) in the second sampling period, and proportion of adult female survival ( $S_A$ ) estimated using the K-MLE. I assumed that survival rates of yearling and subadult females were equivalent to that of adult females (Clark and Smith 1994).

## RESULTS

### Population Composition

I made 77 captures of 51 black bears during 1,495 trapnights (5.1% trapping success) in 2001 and 2002. An additional 2 yearlings and 29 cubs of radiocollared adult females were captured and tagged in dens and were never subsequently recaptured.

The oldest male and female bears were 10- and 11-years old, respectively. Subadults composed 43% of males and 38% of females. Median age of males (2 yr,  $SE = 0.45$ ,  $n = 21$ ) was lower (Wilcoxon rank sum,  $Z = -2.16$ ,  $P = 0.03$ ) than that of females (3 yr,  $SE = 0.39$ ,  $n = 29$ ). Age distributions of males and females differed ( $\chi^2 = 7.38$ ,  $P = 0.025$ ), with more males in younger age categories (Fig. 1.2). Body mass averaged 100 kg ( $SE = 8.03$ ,  $n = 12$ , range: 77.1-158.8 kg) for adult males and 63 kg ( $SE = 2.74$ ,  $n = 27$ , range: 38.6-112 kg) for adult females.

The cub sex ratio (20F:9M) for 2 years combined was female-biased ( $\chi^2 = 4.172$ ,  $P = 0.041$ ). The sex ratio of captured bears  $\geq 1$  year of age (29F:22M) for 2 years combined did not differ from 1:1 ( $\chi^2 = 0.961$ ,  $P = 0.327$ ).

### Reproduction and Survival

Twenty-five radiocollared female bears were monitored 8,949 radiodays (24.5 radioyears) from 22 May 2001 through 19 March 2003. During that time, 1 possible mortality (due to poaching) may have occurred (OBB15). The annual survival rate estimates including OBB15 as a mortality and as a censored individual were  $0.9 \pm 0.1$  (95% CI) and  $1.0 \pm 0.0$ , respectively.

Litter size of newborn cubs and yearlings for both years combined averaged  $2.2 \pm 0.4$  (95% CI;  $n = 13$ ) and  $1.8 \pm 0.6$  (95% CI;  $n = 9$ ), respectively. Litter survival rates

were available for 8 litters. Fourteen of 19 (73.7%) cubs redenned with their mothers as yearlings. One entire litter of 2 cubs was lost. No adult females tracked to winter dens (n = 13) missed a reproductive cycle. The fecundity estimate for the study period was 0.77 female young/female/year.

### **Density and Population Growth**

The Lincoln-Peterson population estimate including known yearlings and excluding cubs was  $85 \pm 30$  (95% CI). Twenty-nine females and 22 males were captured during the 2 years of study. Lincoln-Peterson abundance estimates for females and males were  $43 \pm 4$  (95% CI) and  $39 \pm 5$  (95% CI), respectively.

The study-area size calculated using the composite home range of adult females was 406 km<sup>2</sup>. As a result, estimates of female and male densities were 0.11 bear/km<sup>2</sup> and 0.10 bear/km<sup>2</sup>, respectively. The density estimate for all bears was 0.21 bear/km<sup>2</sup>. The study-area size calculated using buffered trapping locations was 321 km<sup>2</sup>. As a result, estimates of female and male densities were 0.13 bear/km<sup>2</sup> and 0.12 bear/km<sup>2</sup>, respectively. The density estimate for all bears was 0.27 bear/km<sup>2</sup>.

Age groups used in the Leslie Matrix model were cubs (<1 year), yearlings (1-2 years), subadults (2-3 years), and adults (>3 years). Vital rates included F (0.77), P<sub>CY</sub> (0.74), P<sub>YS</sub> (0.9), P<sub>SS</sub> (0.9), P<sub>SA</sub> (0.9), and S<sub>A</sub> (0.9). Estimated population growth rate ( $\lambda$ ) was 1.14.

### **DISCUSSION**

Composition of this population of black bears was indicative of a colonizing population, and the proportion of males in the adult sample (0.76M:1F) was low among studied populations. Garshelis (1991) summarized data from 10 un hunted populations,

reporting an average ratio of 0.99M:1F. Female home ranges are typically smaller than male home ranges, and male home ranges may encompass several female home ranges. Therefore, larger male home ranges increase the probability of a male-biased sex ratio because males have a greater chance of encountering trap sites. However, Clark and Smith (1994) found a similar female-biased ratio (0.59M:1F) in the Ouachita Mountains of Arkansas. A colonizing population of black bears in Big Bend National Park had a sex ratio of 0.33M:1F (Onorato 2003). An expanding population of brown bears in Sweden had a female-biased sex ratio in core areas (Swenson et al. 1998). A female-biased sex ratio was also characteristic of a colonizing wolf population in Montana (Pletscher et al. 1997).

Because large dominant males can displace females from prime habitats (Beckman and Berger 2003), sex ratio and population density play a role in reproductive efforts. Female condition or social dominance status have been found to affect offspring sex ratios in cervids and primates (Van Schaik and Hrdy 1991, Kojola 1997). Cub sex ratio in the Ouachita Mountains of Oklahoma was female-biased. Female cubs are less energetically expensive to produce and may have a higher probability of reproducing after maturation in an expanding population (Clutton-Brock 1988). Adult females are not limited by space in expanding populations; therefore, they may shift their home ranges to accommodate female offspring without consequences of competition for resources. Female carnivores are likely to breed at first estrus when space is not a limiting factor (Pletscher et al. 1997, Swenson et al. 1998).

Male cubs are more energetically expensive to produce, experience higher potential mortality during dispersal, and must compete with other males to reproduce

(Clutton-Brock 1988). Subadult males must **disperse** to avoid territories occupied by adult males. Swenson et al. (1998) found that **core** areas of an expanding brown bear population had higher female densities. **Therefore**, subadult males dispersing in expanding populations may occupy habitats **with few adult females**, thereby reducing their chances of future reproduction. **In continuous** populations, space is a limiting factor. **Therefore**, female offspring must **compete** with their mothers for vital resources (space and food), possibly reducing reproductive potential of mother and offspring. However, subadult males in continuous populations likely will settle in areas previously occupied by adult females, increasing their reproductive potential. **Therefore, a female's** physiological response to produce female cubs **in an expanding population** has a greater chance of benefiting her future reproductive success.

The Oklahoma bear population is young compared with other populations (Great Dismal Swamp, Virginia: Hellgren and Vaughan 1989, Ouachita Mountains; Arkansas: Clark and Smith 1994; Smoky Mountains, Tennessee: McLean and Pelton 1994), and its age distribution may be another sign of its **expanding nature**. Unhunted populations, such as in Oklahoma, typically have older age distributions than hunted populations (Bunnell and Tait 1981). Mean ages in other unhunted black bear populations (Coahuila, Mexico: Doan-Crider and Hellgren 1996; Great Dismal Swamp NWR, Virginia: Hellgren and Vaughan 1989) were older than the Oklahoma population. Although Clark and Smith (1994) described a rapidly expanding population, the mean ages of both Arkansas populations were older than the mean age of the Oklahoma population. In the Ouachita Mountains of Arkansas, mean age for both sexes was 4.3 years, and the oldest individual was a 17-year-old female (Clark 1991). In the Ozark Mountains of Arkansas, mean age



for both sexes was 5.2 years, and the oldest individual was a 26-year-old male (Clark 1991).

Although there is no legal hunting pressure on the Oklahoma population, poaching is a possible source of mortality. Poaching activities were reported to ODWC at the beginning of the study (R. Fennel, ODWC, personal communication); however, surveillance by ODWC game wardens and presence of bear project personnel and radiocollared bears appeared to deter illegal activities for the duration of the study (R. Fennel, ODWC, personal communication). Alternatively, it is possible that undocumented illegal baiting and harvest may have reduced the number of older bears in this population, resulting in a younger age distribution.

Survival rate of adult females in my study area (0.9) was comparable to estimates from populations in Virginia (0.87: Hellgren and Vaughan 1989), Arkansas (0.98: Clark and Smith 1994), and Tennessee (0.83: McLean and Pelton 1994). Average litter sizes in the Ouachita National Forest, Oklahoma (2.2 cubs/female, 1.8 yearlings/female) were comparable to, but slightly lower than, other populations ( $x = 2.4$  cubs/female: Clark and Smith 1994;  $x = 2.6$  cubs/female,  $x = 2.2$  yearlings/female: Eiler and Pelton 1989).

The density estimate using composite home ranges for estimating study-area size lies within the range of those calculated by other researchers in the Interior Highlands of Arkansas (Ozark,  $0.17/\text{km}^2$ ; Ouachita,  $0.23/\text{km}^2$ ; Clark and Smith 1994), Arkansas Delta ( $0.17\text{--}0.42/\text{km}^2$ ; Smith 1985), Great Smoky Mountains National Park ( $0.29/\text{km}^2$ ; McLean and Pelton 1994), and mountains of North Carolina ( $0.16\text{--}0.23/\text{km}^2$ ; Warburton 1984). Those estimates were calculated using composite home ranges to estimate study-area size; therefore, estimates may be high and biased toward better habitats (Clark 1991).

The total density estimate using buffered trapping locations was higher than those in the Interior Highlands of Arkansas (Ozark, 0.08/km<sup>2</sup>; Ouachita, 0.09/km<sup>2</sup>; Clark and Smith 1994).

Black bears were reintroduced to the Interior Highlands of Arkansas between 1958 and 1968 (Smith and Clark 1994). In the late 1980s, population growth rate was estimated to be 1.26 on the Dry Creek study area in the Ouachita Mountains of Arkansas (Clark and Smith 1994). Although the Dry Creek estimate was higher than the growth rate estimate for the Oklahoma population (1.14), both estimates indicated expanding populations. Growth rate of the Arkansas population likely have slowed since the late 1980s study period.

The combination of topography, geographic location, habitat suitability, and the absence of black bears for decades in the Ouachita Mountains of Oklahoma lends itself to successful expansion from the population in Arkansas. These mountains are composed of east-west trending mountain ridges located on the Arkansas-Oklahoma border and largely consist of pine-hardwood forest. East-west ridges, valleys, and farmland may hinder north-south dispersal (Clark and Smith 1994) but may encourage east-west dispersal. Bears can travel along ridges or valleys to explore new, unoccupied habitat.

Smith and Clark (1994) published a map of relative density estimates for the Ozark and Ouachita Mountains based on bait station surveys. This map indicated the part of the Ouachita National Forest in Oklahoma including my study area supported the highest density of bears in the State. I predict that areas to the north, west, and south currently support lower densities of black bears because habitat to the west of the study

area is less suitable than that included in the study (Hellgren et al. 1998) and those areas are farther from the originating population described by Clark and Smith (1994).

### **MANAGEMENT IMPLICATIONS**

My data are evidence for a rapidly expanding population; however, managers must interpret population-dynamics estimates conservatively due to the short duration and limited spatial extent of this study. My study area encompassed about 40% of the available black bear habitat in the Ouachita National Forest, Oklahoma. Because this area is potentially the first area colonized as bears move into Oklahoma from Arkansas, the population density estimates cannot be extrapolated to areas west of the study site in the remaining 60% of the Ouachita National Forest. Future research should focus on intensive mark-recapture to calculate reliable population abundance and density estimates for the entire Ouachita National Forest.

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Figure I.1. Map of black bear study area (polygon outline) within the Ouachita National Forest, LeFlore County, Oklahoma.

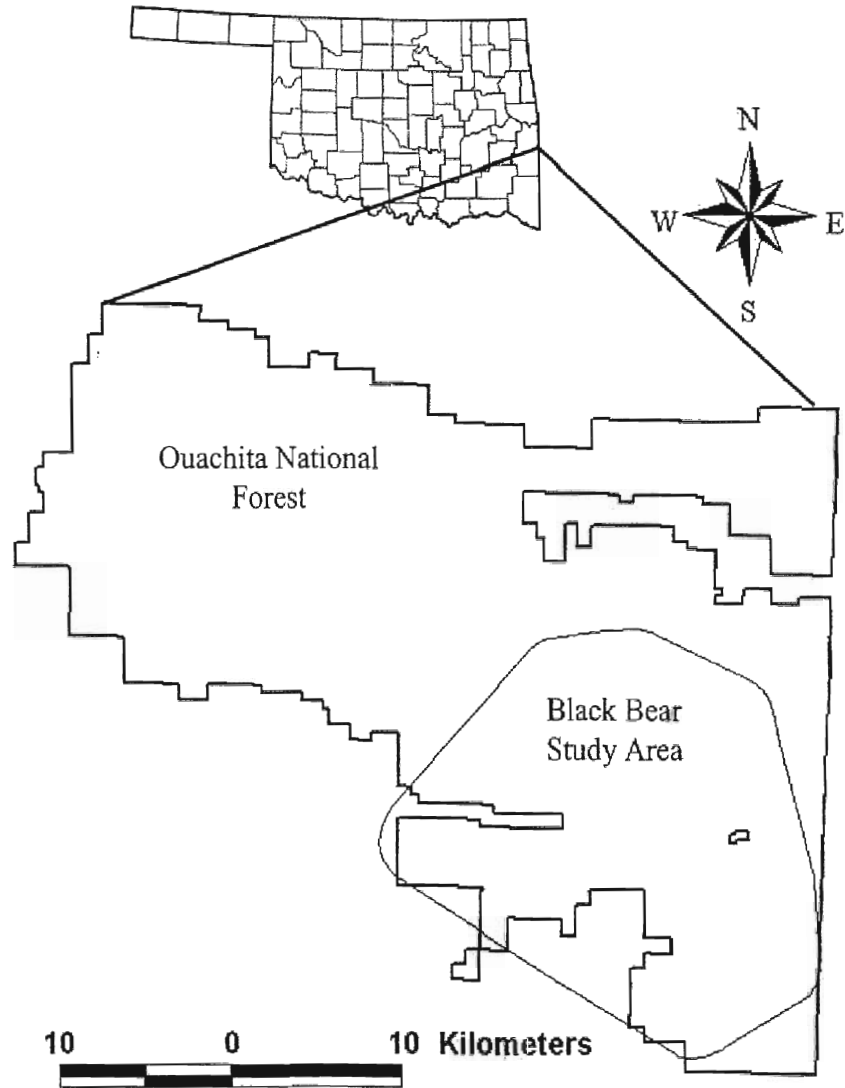
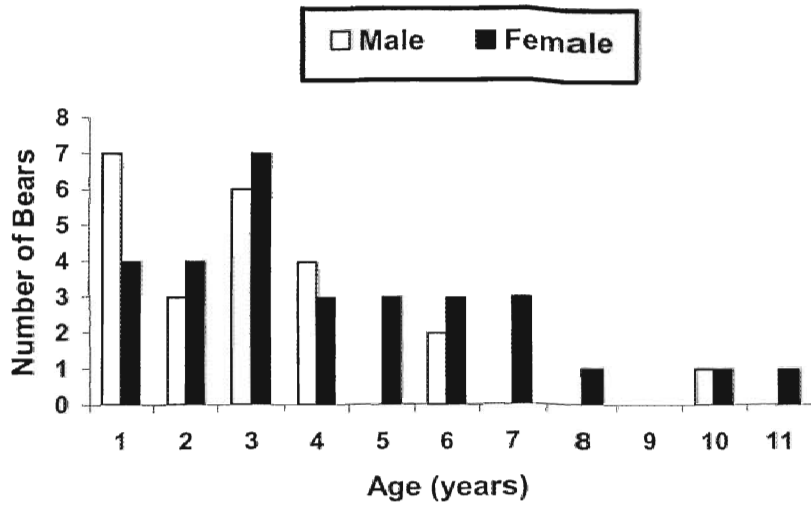


Figure I.2. Age structure of male and female black bears captured in  
Ouachita National Forest, LeFlore County, Oklahoma, in  
2001–2002.



## CHAPTER II

### HABITAT ECOLOGY OF BLACK BEARS IN THE OUACHITA MOUNTAINS OF OKLAHOMA

#### INTRODUCTION AND JUSTIFICATION

Successful wildlife management depends on managers' abilities to assess and understand wildlife-habitat relationships (van Manen 1997). Predictive models of wildlife habitat use commonly are developed to aid effective management of wildlife populations, but many are based on a limited number of habitat variables (van Manen 1997). It is necessary to use multivariate methods to develop effective models at landscape scales (van Manen 1997) because multivariate statistical models account for the multidimensional nature of wildlife habitats and landscapes (Clark et al. 1993a, van Manen 1997, Hellgren et al. 1998). An invaluable tool for modeling wildlife-habitat relationships on a landscape scale is a Geographic Information System (GIS), which is designed to store large amounts of data (Donovan 1987, van Manen 1997, Radeloff et al. 1999).

Geographic Information Systems are used to manage, analyze, and manipulate spatial data (van Manen 1994, Davis 2001). GIS databases typically contain data about broadly defined areas and variables; therefore, multivariate GIS models are most effective for wildlife species with generalized habitat requirements (Clark et al. 1993a, van Manen 1994, van Manen 1997). Radeloff et al. (1999) used GIS to model habitat and population dynamics of German roe deer (*Capreolus capreolus*) over time and

evaluate density dependence of roe deer. Gurnell et al. (2002) used a habitat suitability model for red squirrels (*Sciurus vulgaris*) in GIS to evaluate the relationship between forest management and population estimates for red squirrels. Knick and Dyer (1997) used GIS to develop multivariate descriptions (based on a collection of habitat characteristics) of habitat for black-tailed jackrabbit (*Lepus californicus*) using the Mahalanobis distance statistic.

Efforts toward habitat management of bears are most effective at the landscape scale due to the large home ranges, omnivorous habits, and seasonal use of habitat (Clark et al. 1993a). GIS facilitates the complex multivariate calculations on a landscape scale that are required to develop models that predict bear habitat suitability (Clark et al. 1993a). Kobler and Adamic (2000) developed a habitat suitability model for brown bears (*Ursus arctos*) using a raster (grid-based) GIS. GIS spatial representation of the Kobler and Adamic (2000) model identified habitat fragmentation that would have otherwise gone unnoticed. Gaines et al. (1994) used LANDSAT multispectral scanner imagery and GIS to evaluate suitability of the North Cascades Grizzly Bear Ecosystem (NCGBE) to support grizzly bears (*U. a. horribilis*).

Models can be used to evaluate potential effects of forest management and other habitat changes on bears (van Manen and Pelton 1997). van Manen and Pelton (1997) used GIS, radiolocations, and habitat data to develop predictive habitat models of habitat use by black bears in eastern Tennessee. Clark et al. (1993a) also incorporated GIS in the development of multivariate models of habitat use by black bears in the Interior Highlands of Arkansas. Hellgren et al. (1998) developed a multivariate GIS model of

habitat use by black bears for the Ouachita National Forest in Oklahoma using the paradigm of Clark et al. (1993a) to test the Arkansas model.

Distribution and abundance of black bears in the Ouachita Mountains of southeastern Oklahoma have expanded in recent years. Visitation by black bears at bait stations increased since initiation of bait station lines in 1989–1997 (Skeen 1997). Since 1997, bait station visitation has fluctuated, and it has decreased slightly in recent years (J. Hemphill, Jr., Oklahoma Department of Wildlife Conservation [ODWC], personal communication). Number of bears causing nuisance activity (e.g., crop damage) in Oklahoma also has increased since the late 1980s (J. Hemphill, Jr., ODWC, personal communication). The ODWC considers the Ouachita National Forest critical to supporting Oklahoma's black bear population. However, little is known about the ecology or range dynamics of this species in Oklahoma. Due to the lack of historical information about the black bear population in Oklahoma, it also is important to assess the relationship between home-range dynamics of black bears and land-use practices. Information on range dynamics of the black bear population centered in the Ouachita Mountains of Oklahoma is essential to ODWC in developing a distribution-wide management plan for bears.

Little research has been conducted to determine if models created for 1 area can be useful in predicting habitat selection for other, independent areas. Model validation by field-testing is often impossible because researchers are limited by time and funding (Roloff et al. 2001). Availability of models by Clark et al. (1993a) and its descendent (Hellgren et al. 1998; Fig. II.1), which are based on the Mahalanobis distance statistic (a

multivariate measurement of dissimilarity between points), provided a unique opportunity to test a published, multivariate, GIS model on black bear habitat use.

## **OBJECTIVES AND PREDICTIONS**

The objectives of my research were to test a multivariate GIS model of black bear habitat use at the landscape scale and to describe the home range dynamics and habitat use of black bears in the Ouachita Mountains of Oklahoma. Black bears rely heavily on soft and hard mast during summer and autumn, respectively (Pelton 1982); therefore, I predicted that bears would use regeneration or open-canopy stands more than expected during summer and hardwood stands more than expected during autumn. In an ecological sense, humans interact with bears as predators and/or competitors (Schoen 1989); therefore, I predicted that bears would avoid areas with high human disturbance (van Manen and Pelton 1997). In expanding populations, preferred habitats should be settled first (Garshelis 2000); therefore, I predicted that bear locations would correspond with a higher proportion of low Mahalanobis distance values than the model distribution.

## **MATERIALS AND METHODS**

### **Study Area**

I conducted the study in the Kiamichi and Choctaw districts of the Ouachita National Forest, LeFlore County, southeastern Oklahoma, in a 406-km<sup>2</sup> study area (Fig. I.1). East-west ridges characterize the Ouachita Mountains, with elevations from 400-813 m. Climate consisted of mild winters (average January temperature: 3.9°C) and hot, humid summers (average July temperature: 27.7°C); however, temperatures were lower in high elevations (Oklahoma Climatological Survey 2003). LeFlore County averaged 122 cm of rainfall annually (Oklahoma Climatological Survey 2003).



Rolley and Warde (1985) described three main cover types for the area: pine (*Pinus* spp.) forests (primarily on south-facing slopes), deciduous forests (primarily on north-facing slopes and creek bottoms), and mixed pine-deciduous forests. Pine forests were characterized by an overstory dominated by shortleaf pine (*P. echinata*); a midstory of winged elm (*Ulmus alata*), sparkleberry (*Vaccinium arboreum*), and low blueberry (*V. vacillans*); and an understory of greenbriar (*Smilax* spp.), poison ivy (*Rhus radicans*), and little bluestem (*Andropogon scoparius*). Deciduous forests included an overstory dominated by oaks (*Quercus* spp.) and hickories (*Carya* spp.); a midstory of flowering dogwood (*Cornus florida*), eastern redbud (*Cercis canadensis*), red maple (*Acer rubrum*), and St. Johnswort (*Hypericum* spp.); and an understory of sparglegrass (*Chasmanthium* spp.), panicum (*Panicum* spp.), and wildrye (*Elymus* spp.). Mixed pine-deciduous forests occurred primarily at lower elevations in transition zones between pine forests and deciduous forests (Rolley and Warde 1985).

### **Capture and Handling**

I captured 51 black bears during 1,495 trapnights by Aldrich spring-activated snares modified for bear safety (Johnson and Pelton 1980) and by barrel traps during May–August and October–November, 2001–2002. I anesthetized most bears (trap:  $n = 66$ ; den:  $n = 28$ ) with Telazol (A.H. Robins Company, Richmond, Virginia), a combination of tiletamine hydrochloride and zolazepam hydrochloride, at a dosage rate of 4.8 mg/kg (Doan-Crider and Hellgren 1996). Seven bears were tranquilized with a 2:1 mixture of ketamine-xylazine (Clark and Smith 1994) at a rate of 6.6 mg/kg. Drugs were administered by jabstick. Twenty-eight adult females ( $\geq 36$  kg) were fitted with

radiocollars equipped with mortality sensors (Telonics, Mesa, Arizona). All collars included a cotton spacer (Hellgren et al. 1988).

### **Landscape Analyses**

*Home Range*—Home ranges were estimated by the convex polygon (Mohr 1947) and adaptive kernel (Worton 1989) methods using Animal Movements extension (Hooge and Eichenlaub 1997) in ArcView (ESRI, Redlands, California). The convex polygon model was not constrained by assumptions of distribution and independence of observations; however, it was sensitive to sample size, had a convex shape, and may have overestimated home range due to the presence of outliers (van Manen 1994). The kernel estimator was a nonparametric, scaled-down probability density function placed over each data point to develop the estimator (Worton 1989). It had a higher density where there was a concentration of location points than where there were fewer points (Worton 1989). Home ranges were estimated only for bears with  $\geq 20$  radiolocations.

*Habitat Analyses*—I relocated radiocollared bears 5–10 times monthly using triangulation (azimuths obtained in  $< 50$  min) by ground telemetry with receivers and hand-held H-type antennas. Telemetry station Universal Transverse Mercator (UTM) coordinates, azimuth, and time of reading were recorded. Location estimates of radiocollared bears were assigned UTM coordinates with LOCATE software (Pacer Computer Software, Truro, Nova Scotia, Canada; Nams 1990). Radiocollared females were tracked to winter dens in February–March 2002–2003 to determine den types and site characteristics. To determine triangulation error, test collars were placed in topographic positions and distances from the observer consistent with typical bear radiolocations by assisting personnel (Clark 1991). Test collars were located using the

same methods as for bear locations. System error was determined by calculating average azimuth error using SAS (SAS Institute Inc., 1999–2001, Cary, North Carolina).

Observer error was determined by calculating the average distance from true locations to test locations (Clark 1991) using SAS (SAS Institute Inc., 1999–2001, Cary, North Carolina).

Habitat selection was determined at 2 spatial scales (2nd and 3rd order selection; Johnson 1980): the level of the study area and the level of the home range. In the 1st analysis, availability was considered the habitat composition of the composite home range of radiocollared individuals of a given cohort (i.e., females), and use was considered the vegetative types that composed an individual's home range. In the 2nd analysis, the vegetation-type composition of the home range was considered available habitat for a given individual and the specific types used by that individual were considered used habitat. The U.S. Forest Service provided forest cover maps and stand data for the Kiamichi and Choctaw districts as 1:24,000 ArcInfo coverages developed through their Continuous Inventory Stand Condition management system. I combined coordinates for bear locations with the vegetation stand-type layer used by Hellgren et al. (1998) using the geoprocessing extension for ArcView (ESRI, Redlands, California). The study-area polygon and home-range polygons for 13 adult female bears were intersected with the same vegetation stand-type layer using the geoprocessing extension for ArcView (ESRI, Redlands, California). Areas for individual stand-type patches within study area and home ranges were determined using Patch Analyst (Elkie et al. 1999) for ArcView (ESRI, Redlands, California). Fifteen stand types were combined into 6 vegetation categories for analysis (Table II.1). Condition classes have site-specific

definitions in which “immature” and “mature” refers to the age of the stand and “poletimber” and “sawtimber” refers to the size of the trees. In the Ouachita National Forest, Oklahoma, immature poletimber includes trees < 24.4 cm diameter at breast height (dbh) and < 70 years old; mature poletimber includes trees < 24.4 cm dbh and > 70 years old; immature sawtimber includes trees > 24.4 cm dbh and < 70 years old; and mature sawtimber includes trees > 24.4 cm dbh and > 70 years old (R. L. Bastarache, U. S. Forest Service, personal communication).

Data were analyzed by compositional analysis (Aebischer et al. 1993), which is a multivariate, rank-based method for determining preference. Aebischer et al. (1993) recommended a minimum sample of 10 and preferably > 30 individuals to represent a population adequately when using compositional analysis. Individual animals were considered replicates. Compositional analysis allows the incorporation of other variables, such as sex and season, into the multivariate model to account for different landscape or habitat availabilities that occur among those variables.

Distribution of distances of bear locations from all roads and from paved roads was compared with a distribution of random points to assess effects of land use on bear range dynamics using Chi-square analyses. Random points were generated within composite home range using Random Point Generator v. 1.1 (Jeness Enterprises, Flagstaff, Arizona) in ArcView (ESRI, Redlands, California). Distance to paved roads distributions were grouped into 4 categories ( $\leq 4,000$ ; 4,001-8,000; 8,001-12,000;  $\geq 12,001$ ) and distance to all roads distributions were grouped into 4 categories ( $\leq 500$ ; 501-1,000; 1,001-1,500;  $\geq 1,501$ ) for Chi-square analyses.

*Denning Ecology*—Radiocollared females were tracked to winter dens in January–March, 2002–2003. Den type and site characteristics (i.e., elevation, percent slope, aspect) were recorded. Due to field conditions and time constraints, all den-site characteristics were not recorded for all 29 dens. Estimated den entrance (2002–2003) and exit (2002) dates were recorded. Due to budgetary and time constraints, personnel were unable to monitor dened females to determine den exit dates for 2003.

*Model Validation*—Methods for habitat model development and data acquisition were described in Hellgren et al. (1998) and Clark et al. (1993a). The habitat model was based on the Mahalanobis distance statistic, which is approximately distributed as Chi-square with  $n-1$  degrees of freedom ( $n$  being the number of map layers; Clark et al. 1993a).

Mahalanobis distance is a measurement of dissimilarity and represents the standard squared distance between a set of sample variates ( $\underline{x}$ ) and an ideal habitat ( $\underline{\hat{u}}$ ; Clark et al. 1993a). An inverse relationship exists between Mahalanobis distance value and similarity of a site to the ideal habitat (Hellgren et al. 1998). Hellgren et al. (1998) used the mean vector of habitat characteristics (forest cover type, forest cover type diversity, distance to roads and streams, elevation, aspect, and slope) estimated from Dry Creek radiolocations ( $\underline{\hat{u}}$ ), and the estimated covariance matrix from Dry Creek radiolocations (Clark 1991) to calculate Mahalanobis distances for areas on the Kiamichi and Choctaw districts.

I intersected coordinates of bear radiolocations collected during my study with the 30 x 30-m pixel model of Hellgren et al. (1998) using ArcInfo (ESRI, Redlands, California). To incorporate telemetry error, I created buffers with radii equal to average

error distance around each bear location in ArcView (ESRI, Redlands, California). I used the Random Point Generator v. 1.1 extension (Jeness Enterprises, Flagstaff, Arizona) for ArcView (ESRI, Redlands, California) to generate random points within each buffered zone. Those points also were intersected with the Hellgren et al. (1998) model in ArcInfo (ESRI, Redlands, California). The area of a circle ( $\pi r^2$ ) with a 300-m radius (average error distance) is 282,600 m<sup>2</sup>. Each pixel in a 30 x 30-m pixel model was 900 m<sup>2</sup>; therefore, there were about 314 pixels in the buffer zones. I sorted random-buffered points into 350 sets that included 1 random location per bear location. I created cumulative frequency distributions of Mahalanobis distance values from the Ouachita National Forest (ONF) model, my study area, bear locations, and sets of random-buffered points. I compared the ONF model and study-area distribution to the distributions of sets of random-buffered points. I concluded that distributions differed if the distribution for ONF model or study area fell outside distributions of sets of random-buffered points.

## RESULTS

*Home Range*—A total of 824 radiolocations was made during 8,949 radio-days on 28 female black bears from July 2001 to January 2003. Of those locations, 686 remained (6,949 radio-days) after eliminating individual bears with < 20 radiolocations. All radiolocations were obtained during daylight hours (0700–1900 h). Mean annual home range estimates for females using 100% and 95% minimum convex polygon methods were 24.9 ( $SE = 4.3, n = 13$ ) km<sup>2</sup> and 14.5 ( $SE = 2.7, n = 13$ ) km<sup>2</sup>, respectively (Table II.2; Fig. II.2). The estimate of mean annual home range using the 95% adaptive kernel method was 21.0 ( $SE = 4.3, n = 13$ ; Table II.2). Using 100% minimum convex polygon, 95% minimum convex polygon, and 95% adaptive kernel methods, respectively, mean

summer (May–Aug) home-range estimates averaged 13.3 ( $SE = 2.2$ ,  $n = 10$ ), 11.9 ( $SE = 2.4$ ,  $n = 10$ ), and 21.3 ( $SE = 3.0$ ,  $n = 10$ )  $\text{km}^2$  (Table II.3). Using 100% minimum convex polygon, 95% minimum convex polygon, and 95% adaptive kernel methods, respectively, mean autumn (Sep–Dec) home-range estimates were 16.7 ( $SE = 3.7$ ,  $n = 13$ ), 11.1 ( $SE = 2.1$ ,  $n = 13$ ), and 15.7 ( $SE = 3.0$ ,  $n = 13$ )  $\text{km}^2$  (Table II.3).

*Habitat Analyses*— Four personnel conducted radio telemetry; however, only 2 (myself and 1 technician) had the opportunity to track enough test collars ( $n > 10$ ) to calculate reliable error estimates. Telemetry system error averaged  $1.8^\circ$  ( $SD = 10.7$ ) and  $-3.0^\circ$  ( $SD = 7.1$ ) for 2 observers. Observer error averaged 311.2 m ( $SE = 81.9$ ) and 278.1 m ( $SE = 104.9$ ) for 2 observers. I conducted telemetry with the other 2 observers and believe that the 2-error estimates calculated were representative of the telemetry error of all observers.

Habitat use was nonrandom for 2<sup>nd</sup>-order analysis on an annual basis ( $\chi^2 = 23.8$ ,  $df = 5$ ,  $P < 0.001$ ; Table II.4). Pine-hardwood poletimber differed in rank from oak-hardwood sawtimber ( $P = 0.007$ ) but did not differ in rank from the other 4 habitat types ( $P \geq 0.11$ ). Home ranges of female bears had a lower proportion of oak-hardwood sawtimber than the composite range of all monitored bears, and oak-hardwood sawtimber ranked lower ( $P \leq 0.007$ ) than all other habitat types.

Habitat use was nonrandom for 2<sup>nd</sup>-order analysis in summer ( $\chi^2 = 20.0$ ,  $df = 5$ ,  $P < 0.05$ ; Table II.4). Again, pine-hardwood poletimber ranked highest among all vegetation types. However, its selection did not vary from oak-hardwood poletimber, shortleaf-pine sawtimber, or regeneration ( $P \geq 0.12$ ). Pine-hardwood sawtimber ranked lower than oak-hardwood poletimber and pine-hardwood poletimber ( $P \leq 0.04$ ). Oak-

hardwood sawtimber differed in rank from all other vegetation types ( $P \leq 0.04$ ). Those results indicated that summer home ranges of female bears had a lower proportion of pine-hardwood sawtimber and oak-hardwood sawtimber than the composite range of all monitored bears.

Habitat use was nonrandom for 2<sup>nd</sup>-order analysis in autumn ( $\chi^2 = 17.4$ ,  $df = 5$ ,  $P < 0.05$ ; Table II.4). Regeneration and shortleaf-pine sawtimber ranked 1<sup>st</sup> and 2<sup>nd</sup>, respectively, among all vegetation types although their selection did not vary statistically from oak-hardwood poletimber and pine-hardwood poletimber ( $P \geq 0.15$ ). Pine-hardwood sawtimber was lower in rank than shortleaf-pine sawtimber and regeneration ( $P \leq 0.02$ ). Oak-hardwood sawtimber also was lower in rank than oak-hardwood poletimber, pine-hardwood poletimber, shortleaf-pine sawtimber, and regeneration ( $P \leq 0.03$ ). Fall home ranges of female bears had a lower proportion of oak-hardwood sawtimber than the composite range of all monitored bears.

Habitat use was nonrandom for 3<sup>rd</sup>-order analysis on an annual basis ( $\chi^2 = 19.1$ ,  $df = 5$ ,  $P < 0.05$ ; Table II.4). Shortleaf-pine sawtimber, regeneration, and oak-hardwood sawtimber ranked 1<sup>st</sup>, 2<sup>nd</sup>, and 3<sup>rd</sup>, respectively. However, selection for those types only varied statistically from pine-hardwood sawtimber ( $P \leq 0.04$ ). Selection of the other 3 types were ranked, in order, oak-hardwood poletimber, pine-hardwood poletimber, and pine-hardwood sawtimber. Radiolocations of bears were less likely to be in pine-hardwood sawtimber relative to its availability within an individual home range than any other vegetation type.

Habitat use was nonrandom for 3<sup>rd</sup>-order analysis in summer ( $\chi^2 = 19.9$ ,  $df = 5$ ,  $P < 0.05$ ; Table II.4). Radiolocations of bears in summer were more likely to be in



shortleaf-pine sawtimber relative to its availability within individual home range than any other vegetation type, and this habitat ranked highest among all vegetation types. However, its selection did not vary statistically from the next 2 highest-ranked types, oak-hardwood sawtimber ( $P = 0.15$ ) and regeneration ( $P = 0.22$ ).

Habitat use was nonrandom for 3<sup>rd</sup>-order analysis in autumn ( $\chi^2 = 24.9$ ,  $df = 5$ ,  $P < 0.001$ ; Table II.4). Shortleaf-pine sawtimber and oak-hardwood sawtimber ranked 1<sup>st</sup> and 2<sup>nd</sup>, respectively, among all vegetation types. However, their selection only differed from pine-hardwood sawtimber ( $P = 0.02$ ). Radiolocations of bears in autumn were less likely to be in pine-hardwood sawtimber relative to its availability within individual home range than shortleaf-pine sawtimber or oak-hardwood sawtimber.

Distances to roads of black bear locations and random locations were compared for paved roads and all roads (including paved and unpaved). Average distances of bear and random locations from nearest paved road were 3,018 m (SE = 98,  $n = 815$ ) and 2,674 m (SE = 103,  $n = 815$ ), respectively (Fig. II.3). Distributions of distances of bear radiolocations and random locations to paved roads differed ( $\chi^2 = 88.6$ ,  $df = 3$ ,  $P < 0.0001$ ), with bears less likely to be found  $\leq 4,000$  m from paved roads than random points. Average distances of bear and random locations from nearest road were 399 m (SE = 10,  $n = 815$ ) and 546 m (SE = 16,  $n = 815$ ), respectively (Fig. II.4). Distributions of distances of bear radiolocations and random locations to all roads differed ( $\chi^2 = 88.0$ ,  $df = 3$ ,  $P < 0.0001$ ), with bears more likely to be found  $\leq 500$  m from roads than random points.

*Denning Ecology*—I monitored 20 radiocollared females for 29 den-years. Nine bears were monitored for 2 consecutive years, and 11 were monitored for only 1 year ( $n =$

2 in 2002,  $n = 9$  in 2003; Fig. II.5). All radiocollared females entered dens by 1 January 2002–2003 and exited dens by 1 May 2002. Den types included hollow bases of hardwood trees ( $n = 8$ ), ground excavations ( $n = 8$ ), elevated hardwood tree cavities ( $n = 5$ ), rock cavities ( $n = 4$ ), and ground nests ( $n = 4$ ). I documented 2 cases of den site reuse: 1 radiocollared female denned in the same ground excavation in 2002 and 2003, and 1 denned in the same elevated tree cavity in 2002 and 2003. Elevation at winter den sites averaged 543 m (SE = 24,  $n = 29$ , range = 177–592 m). Percent slope averaged 19% (SE = 2,  $n = 15$ , range = 6–33%). The majority of dens ( $n = 9$ ) were located on north-facing slopes, with others on northeast-facing ( $n = 2$ ), south-facing ( $n = 2$ ), west-facing ( $n = 2$ ), northwest-facing ( $n = 2$ ), southeast-facing ( $n = 1$ ), and east-facing ( $n = 1$ ) slopes.

*Model Validation*—Distribution of Mahalanobis distance values for bear locations was within the range of distributions of distance values for sets of random points in the buffered zone surrounding bear locations. Distribution of modeled Mahalanobis distance values for the Ouachita National Forest and study area were shifted to the right of the distribution of distance values for sets of random-buffered locations (Fig. II.6).

Distribution of Mahalanobis distance values for the study area was shifted to the left of the distribution of distance values for the entire National Forest. These differences indicated that the study area was comprised of a higher proportion of ideal habitat than the entire National Forest and that bears were selecting points closer to the ideal habitat (e.g., the multivariate mean habitat vector of bear locations) than expected based on modeled suitability.

## DISCUSSION

*Home Range*— Home-range estimates for adult females in Oklahoma were similar to those reported for other areas (range = 8–49 km<sup>2</sup>; Wooding and Hardisky 1994). Black bear home-range size is influenced by factors such as age, sex, habitat quality, and population density (Pelton 1982). Clark (1991) reported that home ranges of adult females averaged 34.7 km<sup>2</sup> in the Interior Highlands of Arkansas, which were similar in habitat composition to the Ouachita National Forest, Oklahoma. Home ranges of adult females in eastern deciduous forests ranged from 8 to 18 km<sup>2</sup> (Garshelis and Pelton 1981, Warburton and Powell 1985, Powell et al. 1997). Old-growth and mature, late-successional forests provide most of the habitat requirements for black bears in the southern Appalachians (Powell et al. 1997); therefore, bears typically occupy small home ranges in that region. Bears travel long distances to find productive patches during food shortages (Powell et al. 1997, Garshelis and Pelton 1981), or in areas where resources are more widely distributed annually (Edwards 2002). At Osceola National Forest, Florida, where resources are scattered across the landscape, mean annual home range for adult females was 62 km<sup>2</sup> (Wooding and Hardisky 1994).

Climate and topography influence quantity, quality, and distribution of black bear foods, which probably set constraints on the size of bear home ranges (Amstrup and Beecham 1976, Rudis and Tansey 1995). Female black bears typically have smaller home ranges than male black bears. Females may increase fitness by using areas just large enough for self-maintenance and rearing cubs (Smith and Clark 1994). Female bears increase their long-term efficiency in exploiting the environment by leaving productive patches to explore new patches, thereby improving overall fitness (Amstrup

and Beecham 1976). The range of annual home-range estimates in the present study (2.9–35.2 km<sup>2</sup>,  $n = 13$ ) indicated that some females were located in areas where resources were clumped whereas others were located where resources were more widely distributed. Overall, it appeared that females in my study area occupied suitable habitat and did not require large areas to meet their requirements.

*Habitat Analyses*—Morrison et al. (1992) defined habitat as an area with resources and environmental conditions combined to support survival and reproduction of individuals (or a population) of a given species. Habitat quality or suitability is defined by Garshelis (2000: 113) “as the ability of the habitat to sustain life and support population growth.” Whittaker et al. (1973) and Morrison et al. (1992) suggested that habitat quality is related to the survival and reproductive ability of a species or population and to the stability of habitat through time. Animals should prefer habitats of high quality, and fitness should be higher for animals subsisting in preferred habitats (Garshelis 2000). Reproductive and survival data may be a better reflection of influences of habitat on demographic characteristics of species (Whittaker et al. 1973, Morrison et al. 1992, Garshelis 2000); however, data on habitat-specific demography are difficult to obtain.

Compositional analysis revealed only weak patterns of habitat selection at both scales. The small **number of bears involved** in the analysis ( $n = 13$ ) and subsequent low power may have contributed to the inability to detect clear selection among habitats. Nevertheless, I was surprised that pine and mixed pine-hardwood stands ranked high at both scales. In most studies of habitat selection by black bears, hardwood habitats, especially those dominated by oak, are highly preferred and pine stands are relatively

avoided (Hellgren et al. 1991, Clark et al. 1993a). Habitat selection is often tied to stands of abundant soft and hard mast-bearing shrubs and trees, especially oaks (*Quercus* spp.; Hellgren et al. 1991). Productive stands of mast-bearing trees are important habitat components in autumn, whereas older clearcuts may provide important soft mast in summer (e.g., *Vaccinium* spp., *Prunus serotina*, *Rubus* spp.; Hellgren et al. 1991).

Clark et al. (1994) used 2<sup>nd</sup>-order selection in their evaluation of bear habitat selection in the Ouachita Mountains of Arkansas. They found that bear use of immature poletimber (site-specific definition: trees with small diameter at breast height) stands of white oak (*Q. alba*)–red oak (*Q. rubra*)–hickory (*Carya* spp.) was greater than expected in early and late summer. Clark et al. (1994) incorporated abundance of food plants, a measurement of yearly fruit production, and a measurement of relative frequency of food items in bear diets to develop food value indices (FVI) for habitat types in the Interior Highlands of Arkansas. Bears fed on pokeweed (*Phytolacca americana*), blackberry (*Rubus* spp.), and blueberry (*Vaccinium* spp.) in early summer (Clark et al. 1987, 1994). During early summer, white oak–red oak–hickory poletimber and sawtimber had the highest FVI values (Clark et al. 1994). Pokeweed, black cherry (*Prunus serotina*), blackberry, and blueberry made up the majority of food items in bear diets in late summer (Clark et al. 1994). The habitat with highest FVI value was shortleaf-pine regeneration during late summer (Clark et al. 1994). Results from the present study were similar to those reported by Clark et al. (1994). Female bears in Oklahoma used pine-hardwood and oak-hardwood poletimber stands more than expected during summer. Both studies found that shortleaf-pine habitats were ranked lower than mixed oak and hardwood stands.

In autumn, immature poletimber and sawtimber (site-specific definition: trees with larger diameter at breast height) of white oak–red oak–hickory stands were used more than expected by bears in Arkansas (Clark et al. 1994). Although, these stands ranked high in soft mast production in summer, their use by bears in autumn was related to hard-mast production. Clark et al. (1987) found that acorns were most common in diets but use of acorns and hickory nuts were related to availability. Important hard mast-producing habitats in autumn were white oak–red oak–hickory low-quality poletimber and shortleaf-pine mature sawtimber (Clark et al. 1994). Most pine types in the Ouachitas of Arkansas were used less than expected (Clark et al. 1993*a*, 1994). Results from the present study differed from those reported by Clark et al. (1994) because bears in Oklahoma used oak-hardwood poletimber and sawtimber stands less than expected during autumn. Another difference between the 2 Ouachita studies was that female bears in Oklahoma used regeneration and shortleaf-pine sawtimber stands more than expected, whereas bears in Arkansas used regeneration and shortleaf-pine stands less than expected during autumn.

My study indicated that female black bears in the Ouachita National Forest, Oklahoma, used oak-hardwood sawtimber less than expected in 2<sup>nd</sup>-order analyses. This result differs from most published evaluations of black bear habitat use (Powell et al. 1997) and contrasts with results of Clark et al. (1994) in the same ecoregion. Oak-hardwood stands composed only 17.5 % of the Ouachita National Forest, Oklahoma, and 13 % of the present study area (Fig. II.7). Upland hardwood forest types composed 17.5 % (Clark et al. 1993*b*) of the Arkansas study area of Clark et al. (1994), a proportion equal to the oak-hardwood composition of the Ouachita National Forest in Oklahoma. I

believe that low availability of oak-hardwood habitat partially explains lack of selection for this vegetation type at the home-range scale. It also is likely that widespread but sparse distribution of this vegetation type within my study areas had a negligible influence on habitat selection by black bears at the home-range scale. If available oak-hardwood stands were grouped into smaller areas, bears could more easily select home ranges that include those stands. However, it may be more difficult to include in home ranges oak-hardwood stands that are more widely distributed over the landscape.

Pine and regeneration vegetation types were used more than expected in 3<sup>rd</sup>-order analyses. Forest management techniques on the National Forest may provide an explanation for some of the disparities between previous work and my study. The Ouachita National Forest in Oklahoma has an intensive pine management program (R. L. Bastarache, U. S. Forest Service, personal communication). All timber harvesting occurs within pine or pine-hardwood stands, with the exception of occasional wildlife stand improvements. There are no commercial timber sales within hardwood stands. Harvested areas are prescribe-burned within 5 years of harvest, which reduces standing dead vegetation and ground litter accumulation that can inhibit herbaceous vegetation growth (Masters et al. 1993). Thinning of stands and fire opens the overstory and reduces midstory vegetation, resulting in warmer soil temperatures, increased nitrogen availability, and increased surface light intensity (Masters et al. 1993). These management practices may encourage earlier growth and increased production of herbaceous and soft-mast producing vegetation (Masters et al. 1993, Perry et al. 1999). Total production of soft mast was greater in harvested stands than in unharvested stands in the Ouachita Mountains of Arkansas and Oklahoma (Perry et al. 1999). Clark et al.

(1994) found that bear habitat use was related to soft mast production in the Ouachita Mountains of Arkansas. This may explain our findings that black bears in Oklahoma use pine stands more than expected.

Management practices also may explain the high ranking of regeneration areas in 2<sup>nd</sup>- and 3<sup>rd</sup>- order habitat analyses of habitat selection for bears in the present study. Clearcuts and regeneration stands produce more soft mast than unharvested areas and areas managed using other silvicultural techniques in the Ouachita Mountains of Arkansas and Oklahoma (Perry et al. 1999) and other southeastern pine forests (Stransky and Roese 1984). Other studies of black bear habitat use in southeastern states have documented the importance of regenerating clearcuts to bears (Hellgren et al. 1991). Conversely, Clark et al. (1994) found that regeneration areas were used less than expected by female black bears during all seasons in the Ouachita Mountains of Arkansas, although white oak–red oak–hickory immature sawtimber and shortleaf-pine regeneration ranked highest for soft mast production during autumn (Clark et al. 1994). Important autumn soft mast was pokeweed, Carolina buckthorn (*Rhamnus caroliniana*), devil’s walking stick (*Aralia spinosa*), spicebush (*Lindera benzoin*), and greenbriar (*Smilax* spp.–Clark et al. 1994). It was suggested that the exclusion of female bears by male bears explained the presence of feeding sign in those areas but the relatively low numbers of radiolocations of females there (Clark et al. 1994). I documented female-biased adult and cub sex ratios and high female population density in this study (Chapter I). Furthermore, I found that female bears used highly suitable habitats based on model validation. These factors suggest limited exclusion of females from suitable habitats by



males and may explain the importance of regeneration areas to females in the Ouachita Mountains of Oklahoma.

Area characteristics (e.g., roads, human activity) also may influence habitat use by bears. Road density often is correlated with human activity, influencing hunter (illegal or legal) access in remote areas. Roads may attract bears in protected areas with no hunting season (Brody and Pelton 1989, Hellgren et al. 1991); however, bears may avoid roads in areas with unrestricted vehicle use and an open hunting season (Brody and Pelton 1989, Hellgren et al. 1991).

Bears appeared to avoid paved roads in my study but were closer to all roads (which were mostly graveled or dirt) than random points. Paved roads, especially high-speed or divided highways, have significant impacts on black bear survival (Edwards 2002). Frequency of road crossing or proximity to roads of an individual bear is a result of a trade-off between resource exploitation and potential mortality because crossing a paved road increases mortality risk (Brody and Pelton 1989). Paved roads in my study area are two-lane and have a 105 km/h speed limit, excluding areas of steep grades and sharp curves. Bear-vehicle collisions are reported each year (J. Hemphill, Jr., ODWC, personal communication). It is likely that bears in my study area avoided paved roads due to potential mortality. Bear-hunting season is closed in Oklahoma, but poaching was evident in parts of the study area at the outset of the study (R. Fennel, ODWC, personal communication). These activities were known to occur in areas not included in home ranges of radiocollared bears (trapping efforts were focused in other locations), and poaching appeared to slow throughout the duration of the study. Potential mortality due to illegal hunting may reduce the frequency of road travel by bears. However, graveled

and dirt roads have a lower traffic volume and vehicles must travel at lower speeds due to inhospitable road conditions. I observed bears traveling such roads during my study, and it appears that these roads provided important travel corridors with little threat of mortality.

*Denning Ecology*—Den site selection is vital to black bear survival and productivity (Johnson and Pelton 1981, Oli et al. 1997, Clark et al. 1998). Black bears do not eat, drink, urinate, or defecate during denning and can spend over one-half their lives in winter dens (Clark et al. 1998). Hibernation is a highly specialized and physiologically complex adaptation that restricts bear winter-feeding and movement (Johnson and Pelton 1980). Secure, well-insulated den sites are critical to females because parturition and the first 8–10 weeks of cub development occur in dens (Oli et al. 1997, Clark et al. 1998). Bears in southern regions typically den for shorter periods because winters are shorter and less severe (Hellgren and Vaughan 1989, Oli et al. 1997). Females enter dens first, followed by subadults and adult males (Johnson and Pelton 1980). Lack of insulating snow cover throughout southern habitats may make denning habitat especially vital (Johnson and Pelton 1981, Clark et al. 1998). Den sites selected include elevated tree cavities (Johnson and Pelton 1981), hollows at the base of live or dead trees, rock cavities (Johnson and Pelton 1981, Oli et al. 1997, Clark et al. 1998), brush piles and ground dens (Hellgren and Vaughan 1989), and human-made structures.

Characteristics of den chronology and den types of bears in Oklahoma were similar to those of bears in Arkansas (Hayes and Pelton 1994, Oli et al. 1997). However, frequency of den-type use differed between study areas, and differences in den selection between the 2 study areas may be due to differences in availability. Hayes and Pelton

(1994) reported that the majority of dens located in the Arkansas Interior Highlands were in rock cavities, whereas rock cavities were the least used (same proportion as open nests) in Oklahoma. Clark et al. (1998) suggested that rock cavities were the most desirable den type in the Arkansas Interior Highlands because they were thermally efficient, secure, long-lasting, and require little preparation. The majority of dens in the Ouachita Mountains of Oklahoma were in tree cavities, whereas Hayes and Pelton (1994) only reported 1 den in a tree cavity in the Ouachita Mountains of Arkansas. Tree dens provide protection from disturbance and energy savings relative to ground dens, and females may use this conserved energy for parturition and lactation (Johnson and Pelton 1981).

Pine and mixed pine-hardwood stands in the Ouachita National Forest are intensively managed for timber, leaving few large, decayed trees with cavities sufficient for den sites (White et al. 2001). Dens in tree cavities in the Ouachita Mountains of Oklahoma were located in hardwood-dominated stands managed primarily for wildlife. Availabilities of hardwood-dominated stands on the Arkansas and Oklahoma study sites were 17.5 % (Clark et al. 1993b) and 13 %, respectively. Although availability of tree cavities may be low, bears appeared to select for this den type, possibly as a result of increased protection and energy savings relative to other den types. Nevertheless, the variety of den types observed suggests that suitable den sites do not limit the black bear population in Oklahoma.

*Model Validation*— There are 5 basic steps of GIS habitat modeling: extraction of descriptive habitat data with GIS; statistical analysis outside GIS environment; spatial modeling in GIS based on statistical analysis; mapping and simulations; and model

testing (van Manen 1997). Hellgren et al. (1998) performed steps 1–4 to develop a multivariate model of habitat suitability for the Ouachita National Forest using the original model of Clark et al. (1993a). The final step, model testing or validation, is often conducted with the same data sets through techniques such as jackknifing and splitting of data sets (Cressie 1993). Availability of the Clark et al. (1993a) model provided me an opportunity to validate a published model with independent data.

Mahalanobis distance statistic should be used to describe habitat suitability when distribution of the habitat variable does not change, the landscape is thoroughly sampled to determine the mean habitat vector, and animals are distributed optimally (Podruzny et al. 2002). There were no large-scale changes in the landscape in my study area between model creation and collection of bear habitat-use data. The multivariate mean habitat vector was based on 1,395 locations from radiocollared female bears “sampling” a 518-km<sup>2</sup> area of the Ouachita Mountains in Arkansas (Clark et al. 1993a, Clark and Smith 1994). I assumed that animals were distributed optimally but was unable to test that assumption.

The Clark et al. (1993a) model was not invalidated. As predicted, female black bears used areas with lower Mahalanobis distance values with greater frequency than expected based on availability within the study area and Ouachita National Forest. Results of model validation indicate multivariate models of habitat suitability developed for 1 area can be used to predict habitat use in other, independent areas of similar habitat. However, it is imperative to assess each model independently. Differences in population characteristics and model variables may influence a model’s applicability to other areas (Mitchell et al. 2002). Habitat types with large Mahalanobis distance values also can

only be interpreted as less similar to average bear locations than types with small values. Highly suitable and unsuitable habitat types may have combinations of variables that are similarly distant in multivariate space (Podruzny et al. 2002). Professionals may manage for areas with low Mahalanobis distance values to provide suitable bear habitat; however, habitat types with high Mahalanobis distance values should be considered as well.

## MANAGEMENT IMPLICATIONS

It is essential to understand habitat needs of black bears to effectively conserve and manage populations. Habitat use analyses indicate that female black bears in Oklahoma are responding to the current management practices of the USFS on the Ouachita National Forest. Although pine and mixed pine-hardwood habitat appears to be selected for in this area, it is important to recognize the importance of hardwood stands, particularly those including oaks, to black bear productivity. Hard mast is an important component in bear-diets during fall, when females are preparing to enter dens and produce young (Pelton 1989). It is necessary to maintain a heterogeneous landscape that includes habitats that produce hard- and soft-mast.

The Clark et al. (1993a) model predicted habitat suitability for black bears, but only for the Ouachita National Forest. To develop a statewide black bear management plan in Oklahoma, it is important to provide a predictive model of habitat suitability for the entire state. Various methods have been used to map vegetation data for use in habitat models. Landsat Multispectral Scanner (MSS) and Thematic Mapper (TM) satellite data were used to map habitats of grizzly bear and caribou (*Rangifer tarandus*; Gaines et al. 1994, Hansen et al. 2000). Gaines et al. (1994) cited studies using aerial photographs and ground-truthing to identify vegetation types for grizzly bears. These

methods may be expensive and time-consuming, and vegetation types may vary based on researcher interpretation of images and photographs. The National Gap Analysis Program (GAP) was developed “to provide broad geographic information on the status of ordinary species and their habitats” (Merchant and Eve 1998:2–3). Accurate maps of landcover and nationwide cooperation by states are critical to achieve this goal. Protocols for landcover-mapping procedures have been developed to ensure that each state’s products are compatible with products developed across the U.S. (Merchant and Eve 1998). Data have been reported for much of the conterminous U.S. and are easily accessible, which may alleviate budgetary and time constraints, and variation in vegetation-type interpretation. GAP landcover maps for Oklahoma could be used in place of the U. S. Forest Service stand maps to develop a new statewide model of bear habitat suitability following the methods of Clark et al. (1993a).

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Table II.1. Fifteen forest stand-types that were combined into 6 vegetation categories to perform compositional analysis to describe black bear habitat use in Ouachita National Forest, LeFlore County, Oklahoma, in 2001–2002.

Stand type	Vegetation category	Abbreviation	% of Area
Hardwood-pine immature sawtimber <sup>a</sup>	Pine-hardwood sawtimber	PHS	3.4
Hardwood-pine mature sawtimber			
Pine-hardwood immature sawtimber			
Pine-hardwood mature sawtimber			
Oak-hardwood immature sawtimber	Oak-hardwood sawtimber	OHS	0.4
Oak-hardwood mature sawtimber			
Oak-hardwood poletimber	Oak-hardwood poletimber	OHP	17.1
Scrub-oak poletimber			
Pine-hardwood poletimber	Pine-hardwood poletimber	PHP	9.2
Shortleaf pine immature sawtimber	Shortleaf-pine sawtimber	SLPS	55.6
Shortleaf pine mature sawtimber			
Shortleaf pine regeneration	Regeneration	REGEN	14.3
Shortleaf pine seedling & sapling			
Pine-hardwood seedling & sapling			
Scrub-oak regeneration			

<sup>a</sup> See text for description of age and size classes for stands.

Table II.2. Annual home ranges (km<sup>2</sup>) of female black bears based on minimum convex polygon (MCP) and adaptive kernel (AK) methods, Ouachita National Forest, LeFlore County, Oklahoma, 2001–2002.

Bear ID	Capture date	Age (yrs)	Radio-days Monitored	No. of Locations	Home-range size (km <sup>2</sup> )			
					MCP 100%	MCP 95%	AK 95%	AK 50%
2	22 May 2001	3	666	71	24.4	7.5	11.9	1.3
6	1 June 2001	4	363	20	17.1	17.1	24.1	3.5
8	4 June 2001	7	653	67	10.2	8.1	8.4	1.2
11	10 June 2001	7	418	30	30.6	19.3	30.0	4.9
12	11 June 2001	5	646	65	31.8	2.9	20.1	3.3
19	20 June 2001	3	333	30	35.2	35.2	50.9	5.3
21	7 July 2001	5	620	75	15.7	10.1	11.7	1.7
22	11 July 2001	3	616	76	9.7	9.4	9.6	0.8
23	15 July 2001	2	612	67	25.2	18.1	14.3	1.5
24	15 July 2001	2	612	61	41.7	13.5	12.9	1.9
28	29 July 2001	3	535	53	13.3	10.2	12.2	1.2
31	22 August 2001	11	573	47	62.3	32.2	56.1	4.2
38	21 May 2002	3	302	24	7.1	5.7	11.5	1.6

Table II.3. Seasonal home ranges (km<sup>2</sup>) of adult female black bears based on minimum convex polygon (MCP) and adaptive kernel (AK) methods, Ouachita National Forest, LeFlore County, Oklahoma, 2001–2002.

Bear ID	Season	Capture date	Age (yrs)	No. of Locations	Home-range size (km <sup>2</sup> )			
					MCP 100%	MCP 95%	AK 95%	AK 50%
2	Fall	22 May 2001	3	47	10.8	6.6	8.7	0.9
6	Fall	1 June 2001	4	11	10.1	10.1	16.1	3.1
8	Fall	4 June 2001	7	46	6.5	5.5	7.8	1.1
11	Fall	10 June 2001	7	21	12.2	9.8	16.2	2.2
12	Fall	11 June 2001	5	39	19.0	18.7	21.9	4.4
19	Fall	20 June 2001	3	20	29.9	26.5	43.2	4.6
21	Fall	7 July 2001	5	47	8.1	3.8	8.7	1.7
22	Fall	11 July 2001	3	53	9.5	9.2	8.5	0.9
23	Fall	15 July 2001	2	44	19.4	12.9	14.4	1.9
24	Fall	15 July 2001	2	45	24.8	6.3	11.2	1.9
28	Fall	29 July 2001	3	37	9.6	7.4	9.4	1.3
31	Fall	22 August 2001	11	34	53.9	24.2	32.5	3.3
38	Fall	21 May 2002	3	17	3.5	3.5	5.9	1.2
2	Summer	22 May 2001	3	23	19.4	16.5	25.6	5.2
8	Summer	4 June 2001	7	21	7.4	4.74	10.1	2.2
12	Summer	11 June 2001	5	25	20.8	19.7	18.3	2.2
19	Summer	20 June 2001	3	10	15.3	15.3	41.0	8.0
21	Summer	7 July 2001	5	16	11.2	11.2	25.2	6.0
22	Summer	11 July 2001	3	23	5.9	5.4	12.9	1.9
23	Summer	15 July 2001	2	23	9.1	6.0	11.4	1.6
24	Summer	15 July 2001	2	16	26.8	26.8	29.6	4.6
28	Summer	29 July 2001	3	15	8.7	8.7	17.2	2.8
31	Summer	22 August 2001	11	13	9.0	4.6	21.6	5.9



Table II.4. Rank of habitat types for black bears in Ouachita National Forest, LeFlore County, Oklahoma, in 2001-2002 based on compositional analysis.

Order	Time	Ranking					
		Highest					Lowest
		1	2	3	4	5	6
2 <sup>nd</sup>	Annual	PHP	OHP	REGEN	SLPS	PHS	OHS
2 <sup>nd</sup>	Summer	PHP	OHP	REGEN	SLPS	PHS	OHS
2 <sup>nd</sup>	Autumn	REGEN	SLPS	PHP	OHP	PHS	OHS
3 <sup>rd</sup>	Annual	SLPS	REGEN	OHS	OHP	PHP	PHS
3 <sup>rd</sup>	Summer	SLPS	OHS	REGEN	PHP	PHS	OHP
3 <sup>rd</sup>	Autumn	SLPS	OHS	PHP	OHP	REGEN	PHS

PHS = pine-hardwood sawtimber, OHS = oak-hardwood sawtimber, OHP = oak-hardwood poletimber, PHP = pine-hardwood poletimber, SLPS = shortleaf pine sawtimber, REGEN = regeneration

Figure II.1. Multivariate GIS model of black bear habitat suitability developed by Hellgren et al. (1998) for the Ouachita National Forest, LeFlore County, Oklahoma.

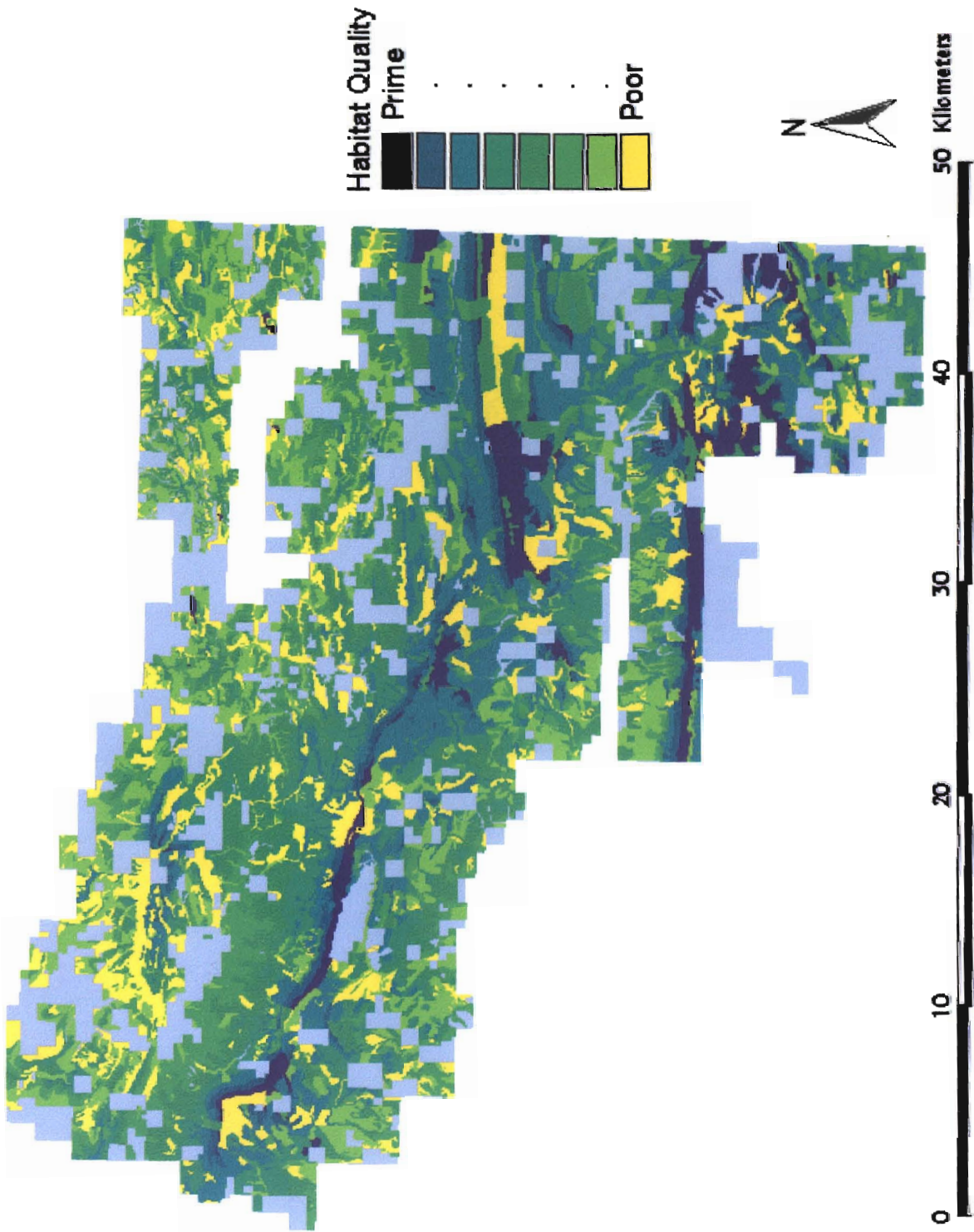


Figure II.2. 95% MCP home range estimates of 13 female black bears in Ouachita National Forest, LeFlore County, Oklahoma in 2001-2002.

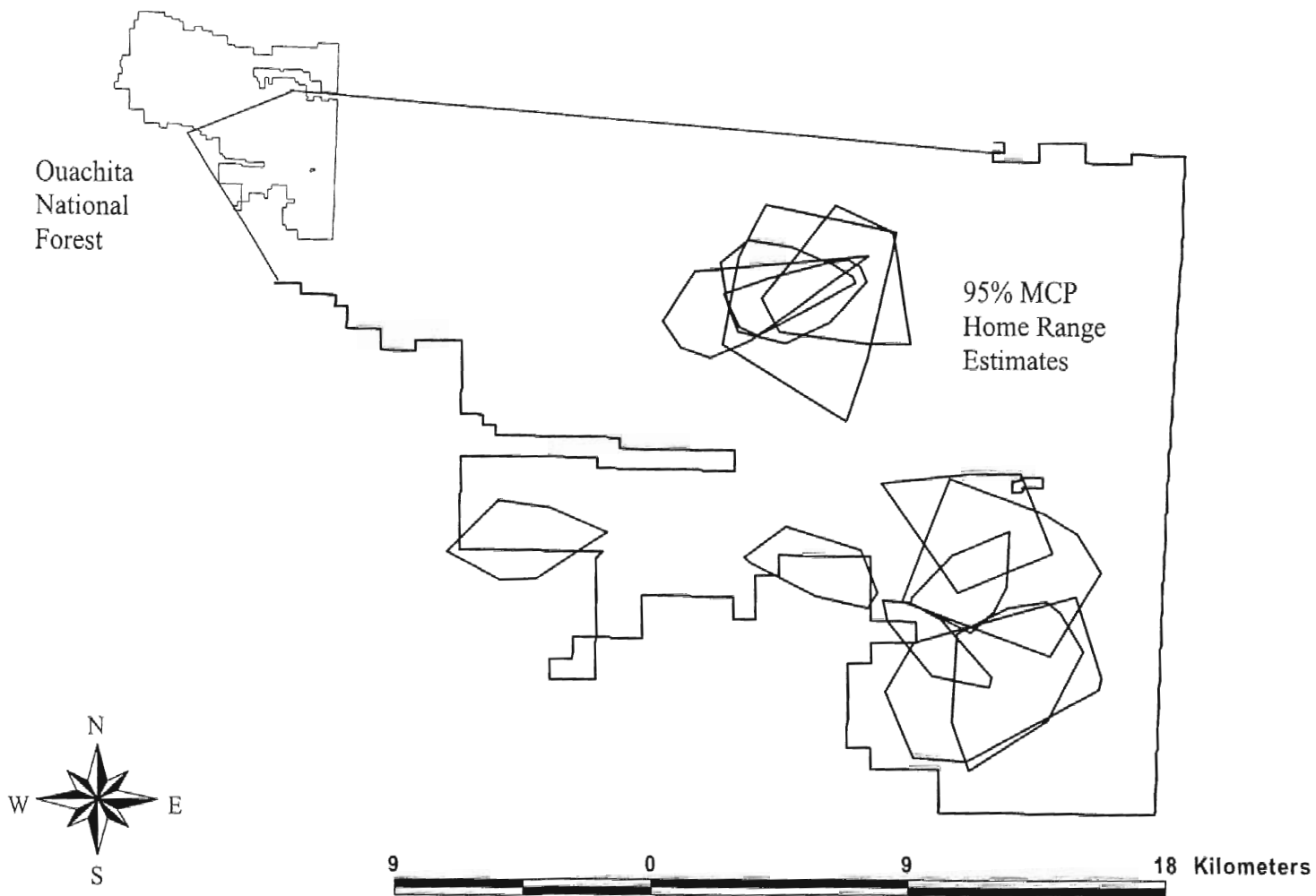


Figure II.3. Distribution of distances to paved roads from black bear and random locations in Ouachita National Forest, LeFlore County, Oklahoma in 2001-2002.

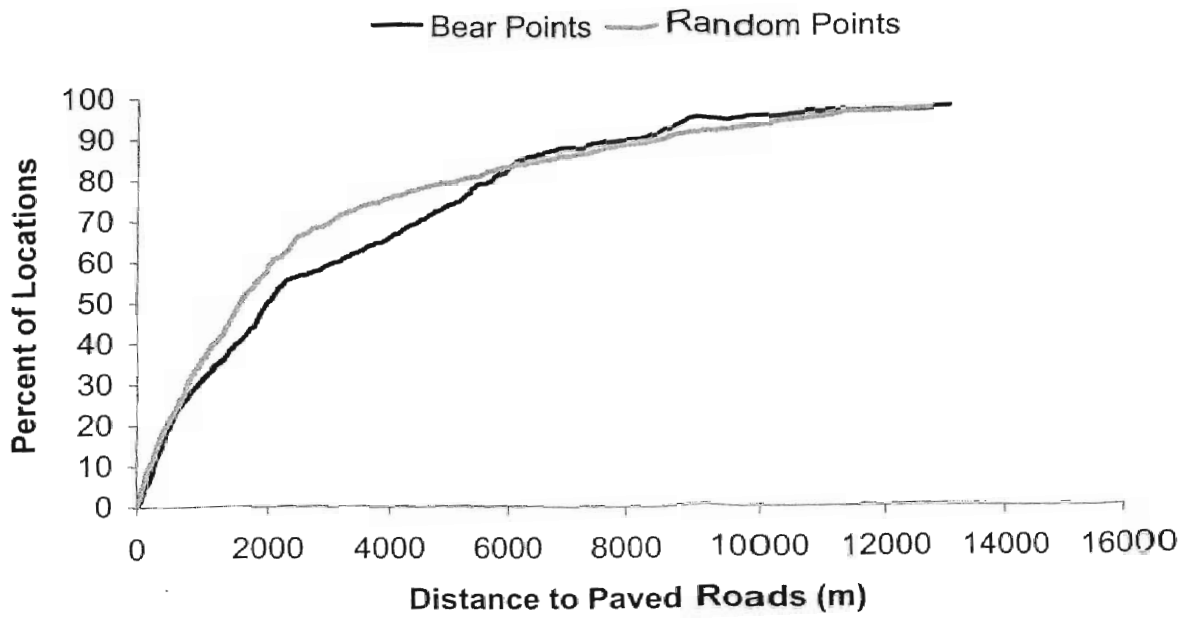


Figure II.4. Distribution of distances to roads (unpaved and paved) from black bear and random locations in Ouachita National Forest, LeFlore County, Oklahoma in 2001-2002.



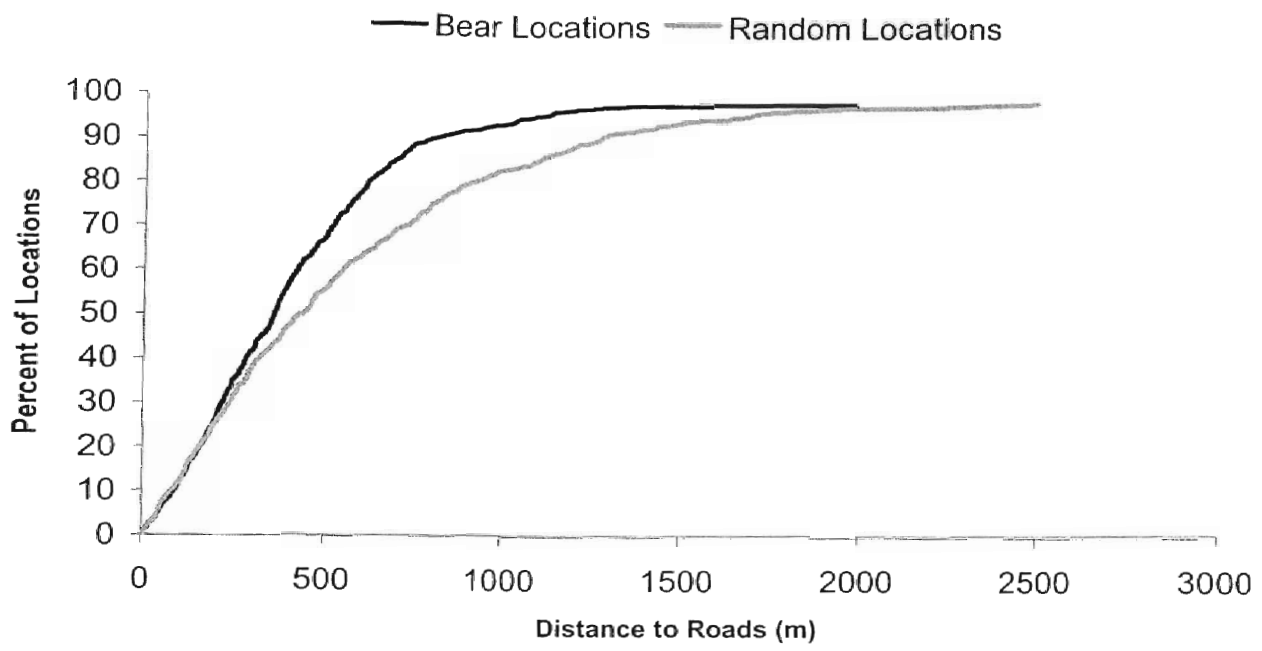


Figure II.5. Winter den locations of radiocollared female black bears in Ouachita National Forest, LeFlore County, Oklahoma in 2002-2003. Triangles represent dens located in 2002 and squares represent dens located in 2003.

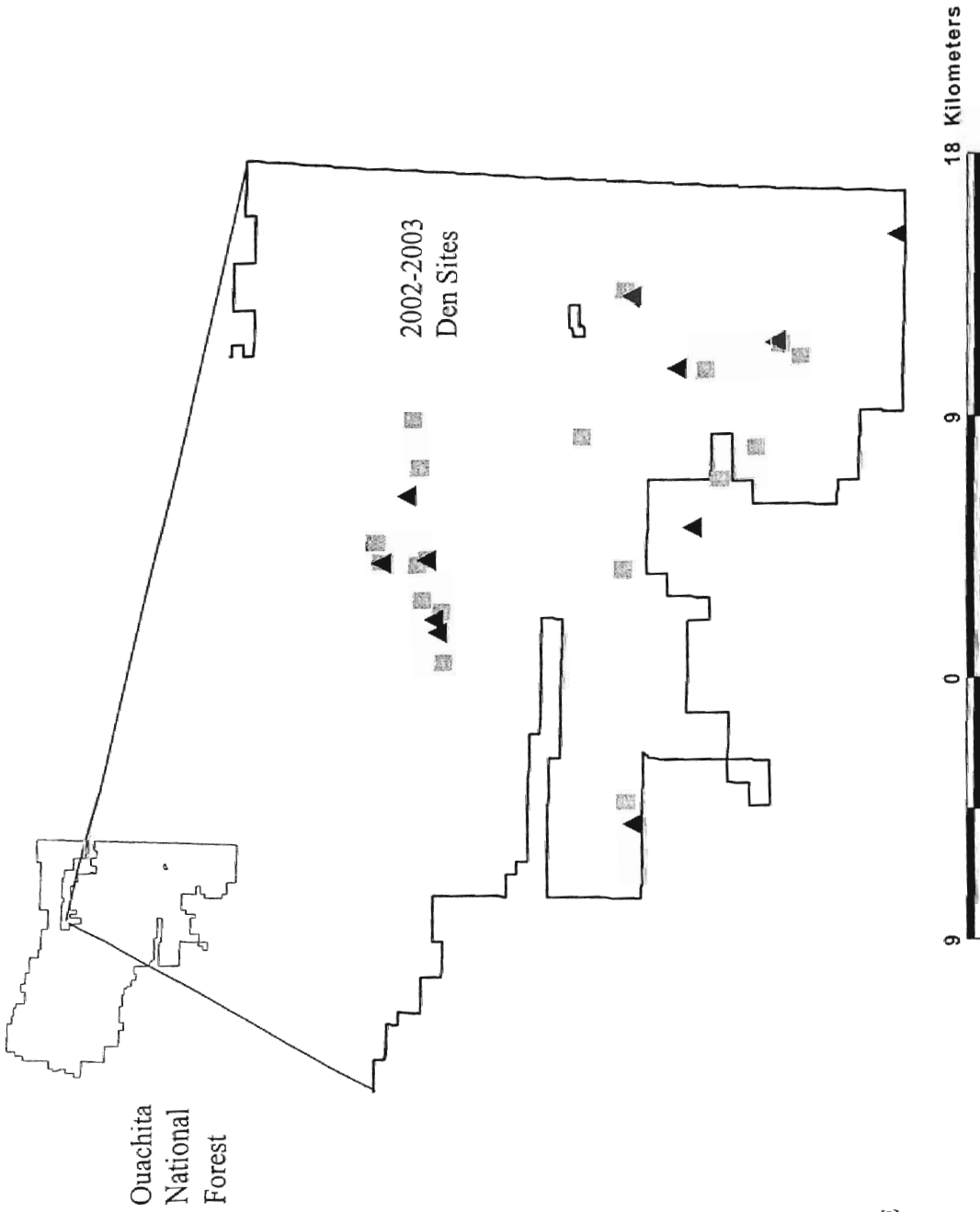


Figure II.6. Cumulative frequency distributions of Mahalanobis distance values for 350 sets of random-buffered locations (gray), bear locations (red), study area (blue), and entire National Forest (black) for the Ouachita National Forest, LeFlore County, Oklahoma. Mahalanobis distance is a unitless multivariate measurement that represents the standard squared distance between a set of sample variates and a multivariate mean vector for these variates.

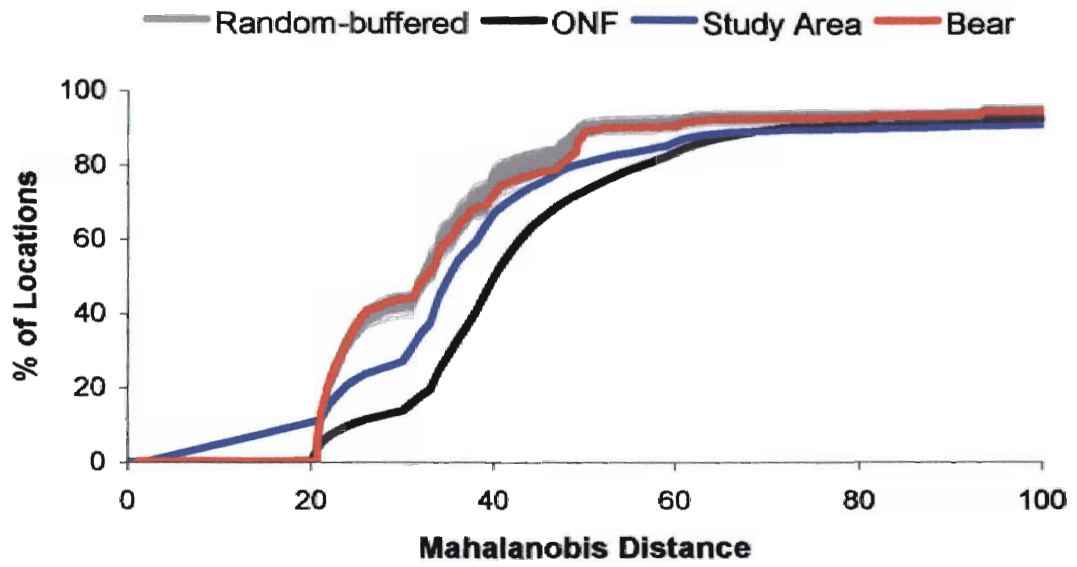
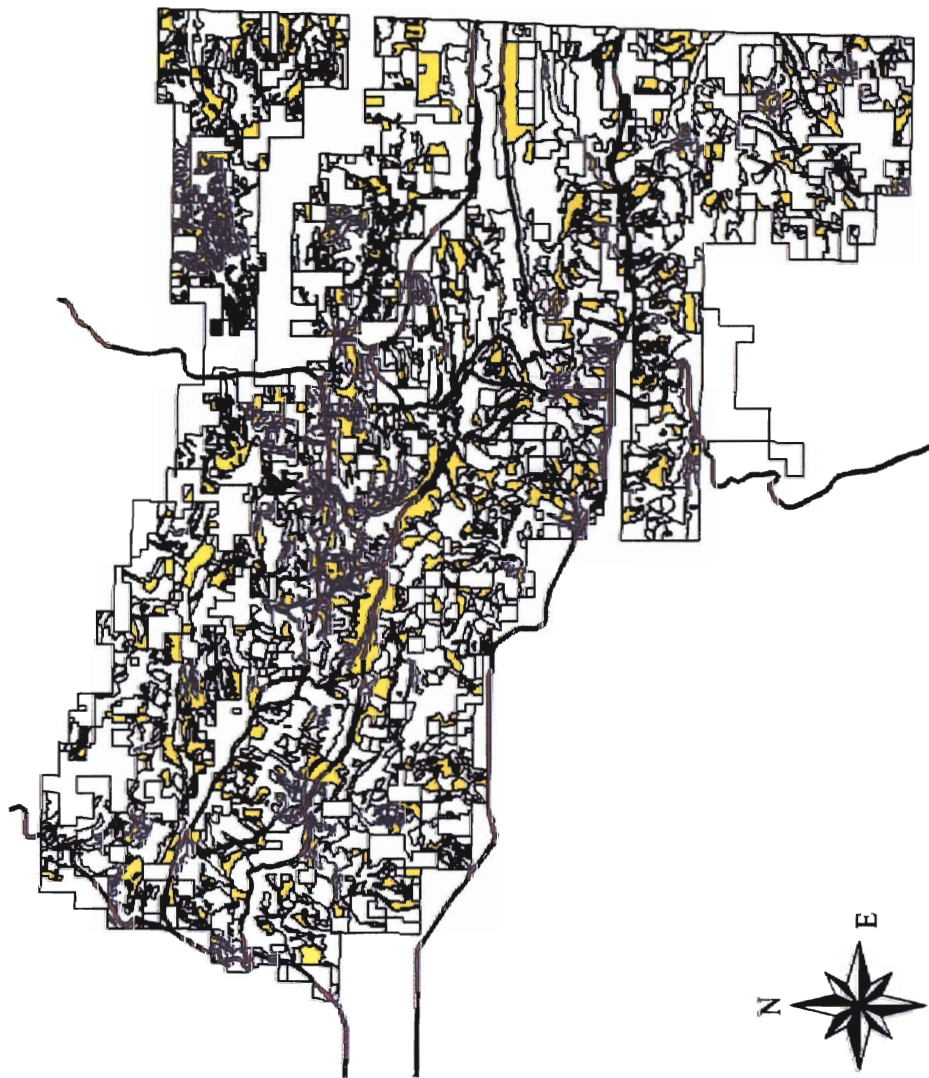


Figure II.7. Distribution of oak-hardwood stands in the Ouachita National Forest, LeFlore County, Oklahoma. Oak-hardwood stands (yellow) composed 17.5 % of the National Forest. Paved roads shown in brown.



30

0

30 Kilometers

## SUMMARY

The distribution and abundance of black bears in Oklahoma has increased dramatically in the last 15 years, with populations in Arkansas serving as the source for the recolonization. I studied habitat ecology and demographic characteristics of black bears on a study area in southeastern Oklahoma to provide background information necessary for effective management of the species and to provide insight into characteristics of colonizing populations of large carnivores. Demographic characteristics supported the view of the Oklahoma population as recolonizing. The sex ratios of adults and cubs were female-biased and the age structure was very young. Survival of adult females was estimated to be  $0.9 \pm 0.1$  and fecundity was estimated to be 0.77 female young/female/year (based on an average litter size of 2.2, 69% female cubs, and a 2-year interbirth interval). Density on the study area was estimated to be 0.21 bears/km<sup>2</sup>. The growth rate of the study population was estimated to be 1.14, or a 14% annual growth. Managers should interpret these data on population dynamics conservatively due to the short duration and limited spatial extent of the study. Population density estimates cannot be extrapolated to areas west of the study site in the remaining 60% of the Ouachita National Forest.

It is necessary to understand wildlife-habitat relationships to effectively manage wildlife populations. I radiotracked adult female black bears to estimate home range areas, determine habitat use, and test a model of habitat suitability. Relatively small home-range estimates indicated suitable habitat in the southeastern portion of the



Ouachita National Forest. Bears were using mixed pine-hardwood and shortleaf-pine stand types more than expected in the Ouachita National Forest, likely due to the soft mast available in those habitats. Currently, the Ouachita National Forest is heavily managed for pine timber; however, hardwood stands are not converted to pine. The combination of pine, mixed pine-hardwood, and hardwood stands on the Ouachita National Forest provides a variety of food and living resources for the black bear population in Oklahoma. However, the heterogeneity of the western portion of the Ouachita National Forest landscape may require larger home ranges for bears to gather necessary resources.

I tested a model of habitat suitability developed by Hellgren et al (1998) from the original model of Clark et al. (1993a). This model accurately predicted habitat suitability for bears in the Ouachita National Forest, Oklahoma. As predicted for expanding populations, bears in this study were distributed closer to the mean habitat vector than expected. As this population grows, bears will be forced to expand and to occupy habitats less suitable to avoid competition for resources. Again, this may require larger home ranges for bears to gather critical resources. This was a unique opportunity to test a published model with independent data. In this case, the model was not invalidated. Future research should focus on developing consistent methods of model creation so that managers may apply models to different areas of similar habitats to aid in conservation.

Appendix Table 1. Data for female black bears captured in Ouachita National Forest, LeFlore County, Oklahoma in 2001 and 2002. Individuals that dropped collars before personnel could relocate them were monitored for 0 radio-days.

ID	Capture date	Body mass		Radio-days monitored	Status
		(kg)	Age (yrs)		
OBB2	22 May 2001	44	3	666	<sup>1</sup> C
OBB3	23 May 2001	49	3	131	<sup>2</sup> DC
OBB5	29 May 2001	25	1		<sup>3</sup> NC
OBB6	1 June 2001	62	4	363	<sup>1</sup> C
OBB8	4 June 2001	71	7	653	<sup>1</sup> C
OBB9	9 June 2001	21	1		<sup>3</sup> NC
OBB10	10 June 2001	69	10	319	<sup>4</sup> PC
OBB11	10 June 2001	68	7	418	<sup>1</sup> C
OBB12	11 June 2001	55	5	646	<sup>1</sup> C
OBB15	16 June 2001	70	5	29	<sup>2</sup> DC
OBB16	16 June 2001	65	6	67	<sup>2</sup> DC
OBB19	20 June 2001	48	3	333	<sup>2</sup> DC
OBB21	7 July 2001	50	5	620	<sup>1</sup> C
OBB22	11 July 2001	39	3	616	<sup>1</sup> C
OBB23	15 July 2001	41	2	612	<sup>1</sup> C
OBB24	15 July 2001	48	2	612	<sup>1</sup> C
OBB28	29 July 2001	63	3	535	<sup>2</sup> DC
OBB30	4 August 2001	77	6	20	<sup>2</sup> DC
OBB31	22 August 2001	82	11	573	<sup>1</sup> C
OBB38	21 May 2002	53	3	302	<sup>1</sup> C
OBB39	25 May 2002	74	8	298	<sup>1</sup> C
OBB41	29 May 2002	44	2	0	<sup>2</sup> DC
OBB42	31 May 2002	76	7	0	<sup>2</sup> DC
OBB43	22 June 2002	64	6	0	<sup>2</sup> DC
OBB44	22 June 2002	60	3	271	<sup>1</sup> C
OBB47	8 July 2002	49	2	38	<sup>2</sup> DC
OBB48	9 July 2002	23	1	253	<sup>5</sup> C
OBB50	12 July 2002	28	1	163	<sup>1</sup> C
OBB51	13 July 2002	60	4	249	<sup>1</sup> C
OBB55	8 October 2002	113	unknown	162	<sup>1</sup> C

<sup>1</sup>C = collared, <sup>2</sup>DC = dropped collar, <sup>3</sup>NC = not collared, <sup>4</sup>PC = pulled

collar, <sup>5</sup>C = yearling collar

Appendix Table 2. Data for male black bears captured in Ouachita National Forest, LeFlore County, Oklahoma in 2001 and 2002.

ID #	Capture date	Body mass (kg)	Age
OBB4	27 May 2001	82	3
OBB7	4 June 2001	154	6
OBB14	11 June 2001	25	1
OBB17	17 June 2001	109	4
OBB18	18 June 2001	27	1
OBB20	25 June 2001	25	1
OBB25	20 July 2001	66	2
OBB26	20 July 2001	86	4
OBB27	27 July 2001	89	3
OBB1	28 July 2001	84	3
OBB29	3 August 2001	39	1
OBB32	14 November 2001	57	1
OBB35	17 May 2002	26	1
OBB36	17 May 2002	159	10
OBB37	18 May 2002	45	2
OBB40	28 May 2002	74	2
OBB45	25 June 2002	79	3
OBB46	28 June 2002	77	3
OBB52	29 July 2002	46	1
OBB53	8 October 2002	45	unknown
OBB54	8 October 2002	58	unknown
OBB56	9 October 2002	43	unknown

VITA



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