PREY OF WOLVES IN THE GREAT LAKES REGION

Glenn D. DelGiudice, Keith R. McCaffery, Dean E. Beyer, and Michael E. Nelson

Wolves (*Canis lupus*) were abundant in the Great Lakes region just prior to early European settlement (early to mid-1800s). The subsequent extirpation of wolves and most of their large prey is just one of the many threats humans have posed to the existence of North American wildlife, both by exploitation and by indifference. To fully understand (and learn from) the recent ongoing recovery of wolves in the Great Lakes region (Minnesota, Wisconsin, and Michigan), and the historic declining trend that preceded it, requires consideration of their food or prey, both historically and today. Ungulate prey species are often at the center of the wolf-human conflict, where and when it occurs. What history, recent management, and research have taught us is that it was not wolf predation that diminished the diversity and richness of ungulate species in the Great Lakes region, but rather the human “appetite,” and unfortunately for wolves and their prey, the unprecedented drive to satisfy it.

On the other hand, humans have a great capacity for conservation when that is their true intention. But the success of such efforts also relies largely on species-specific biology, in this case, not just of wolves, but of their existing prey as well. Wolves are adaptable, opportunistic predators when it comes to their foraging behavior, but what animal species become their prey has depended largely on the potential prey’s size, abundance, and vulnerability. Consequently, the relative contributions of primary and secondary prey to the diets of Great Lakes wolves, to their individual health and welfare, and long-term population persistence, have changed historically, and today continue to vary seasonally, annually, and across the landscape.

This chapter begins with a brief description of the historic trends in distribution and relative abundance of the large ungulates that were likely most important in the multi-prey system of the Great Lakes wolf. Our major focus, however, is the more recent trend of white-tailed deer (Odocoileus virginianus), the wolves’ primary prey in a single ungulate prey system that has persisted throughout the 20th century and during their recent ongoing recovery. We concentrate our discussion on specific aspects of the deer’s ecology that have enabled its populations to thrive despite relatively heavy human exploitation, increasing numbers of wolves, and a concomitant expansion of their range. This discussion is based upon management efforts and an unparalleled amount of data generated from the study of coexisting white-tailed deer and wolves in the Great Lakes region. We devote similar, but more limited attention to moose, which are primary or secondary prey for wolves on Isle Royale and in various portions of northern Minnesota.

From Chapter 10 in “Recovery of Gray Wolves in the Great Lakes Region of the United States: an Endangered Species Success Story.” Submitted.

ASSESSING THE RELATIONSHIP OF CONIFER THERMAL COVER TO WINTER DISTRIBUTION, MOVEMENTS, AND SURVIVAL OF FEMALE WHITE-TAILED DEER IN NORTH CENTRAL MINNESOTA

Glenn D. DelGiudice, Barry A. Sampson, and David W. Kuehn

SUMMARY OF FINDINGS

During January to March 1991 to 2005, a total of 452 female white-tailed deer (*Odocoileus virginianus*), including 43 female newborns, were captured, radiocollared, and recruited into this study assessing the effects of varying winter severities and diminishing conifer cover on numerous aspects of white-tailed deer ecology. The wide-ranging severity of winter weather conditions (winter severity index of 38 in winter 2003-2004 to 195 in winter 1995-1996) during the past 15 years, and the diverse data we have collected, will continue to provide a more comprehensive understanding of white-tailed deer ecology in much of Minnesota's forest zone as we continue our data analyses. During the past year, we've been concentrating our efforts on several tasks, including: 1) organizing the diverse, 15-year data sets in preparation for analyses specifically related to the experimental design of this study; 2) completing data analyses and manuscript preparations; and 3) updating the habitat composition layers for the 4 study sites relative to vegetative succession, natural habitat destruction (e.g., by flooding), and timber harvests. We describe how we addressed vegetative succession and the types of changes we observed, particularly relative to conifer cover, and 4) continued monitoring of the survival of wolves (*Canis lupus*) radiocollared in the region of our study sites and reporting cause-specific mortality during 2006.

INTRODUCTION

The goal of this long-term investigation is to assess the value of conifer stands as winter thermal cover/snow shelter for white-tailed deer at the population level. Historically, conifer stands have declined markedly relative to the increasing number of deer in Minnesota and elsewhere in the Great Lakes region. The level of logging of all tree species collectively, and conifer stands specifically, has recently reached the estimated allowable harvest. Most land management agencies and commercial landowners typically restrict harvests of conifers (to varying degrees) compared to hardwoods, because of evidence, at least at the individual animal level, indicating the seasonal value of this vegetation type to white-tailed deer and other wildlife species. However, agencies have anticipated increased pressure to allow more liberal harvests of conifers in the future. Additional information is needed to assure future management responses and decisions are ecologically sound. Both white-tailed deer and the forests of the Great Lakes region have significant positive impacts on local and state economies, and they are highly regarded for their recreational value.

HYPOTHESES AND OBJECTIVES

The null hypotheses in this study are that conifer stands have no effect on the survival, movement, or distribution of female white-tailed deer during winters of varying severities. Relative to varying winter severities, the specific objectives of the comprehensive, quasi-experimental approach of this study have been to:

- monitor deer movements between seasonal ranges by aerial radio-telemetry, and more importantly, within winter ranges, for determination of home range size;
- determine habitat composition of winter home ranges and deer use of specific vegetation types;

- monitor winter food habits;
- monitor winter nutritional restriction and condition via serial examination of deer body mass and composition, blood and bladder-urine profiles, and urine specimens suspended in snow (snow-urine);
- monitor age-specific survival and cause-specific mortality of all study deer; and
- collect detailed weather data in conifer, hardwood, and open habitat types to determine the functional relationship between the severity of winter conditions, deer behavior (e.g., use of habitat) and their survival.

STUDY DESIGN AND PROGRESS

This study employed a replicated manipulative approach, which is a modification of the Before-After-Control-Environmental Impact design (BACI; Stewart-Oaten et al. 1986; see DelGiudice and Riggs 1996). The study involves 2 control sites (Willow and Dirty Nose Lakes) and 2 treatment sites (Inguadona and Shingle Mill Lakes), a 5-year pre-treatment (pre-impact) phase, a 4-year treatment phase (conifer harvest serves as the experimental treatment), and a 6-year post-treatment phase. The 4 study sites located in the Grand Rapids-Remer-Longville area of north-central Minnesota are 13.0-23.6 km^2 (5.0-9.1 mi^2) in area. The study began with the Willow and Inguadona Lakes sites during winter 1990-1991. The Shingle Mill and Dirty Nose Lakes sites were included beginning in winter 1992-1993. The objective of the experimental treatment (impact) was to reduce moderate (40-69% canopy closure) and optimum (≥70% canopy closure) conifer thermal cover/snow shelter to what is considered a poor cover class (< 40% canopy closure).

Data collected on all 4 study sites included the following: 1) descriptive quantification of deer habitat by color infrared air photo interpretation, digitizing, and application of a geographic information system (GIS); 2) monitoring of ambient temperature, wind velocity, snow depth, and snow penetration (index of density) in various habitat types (e.g., openings versus dense conifer cover) by automated weather data-collecting systems, minimum/maximum thermometers, and conventional hand-held measurements; 3) deer capture, chemical immobilization, and handling data; 4) age determination by last incisor extraction and cementum annuli analysis; 5) physiological samples collected during captures and recaptures of radiocollared female deer and data generated from laboratory analyses, including complete blood cell counts (CBCs), serum profiles of about 20 characteristics, (e.g., reproductive and metabolic hormones, chemistries), urine chemistry profiles, and partial and complete body composition determination by isotope-dilution and visual ultrasound; 6) morphological measurements; 7) assessment of winter nutritional restriction by chemical analysis of urine in snow; 8) seasonal migrations and other movements via very high frequency (VHF) and Global Positioning System (GPS) radiocollars; 9) habitat use; 10) annual and seasonal cause-specific mortality; 11) age-specific survival rates; 12) winter food habits; and 13) movements, territory size, survival, and cause-specific mortality of radiocollared wolves.

We completed the 15th and final year of data collection. Ultimately, we radiocollared and monitored a total of 452 female deer, including 43 female newborn fawns. During 1991 to 2006, in annual issues of the Minnesota Department of Natural Resources' "Summaries of Wildlife Research Findings" we've presented summary data describing the winter weather conditions (e.g., weekly snow depths, monthly mean daily minimum and maximum ambient temperatures, winter severity index); live-capture success; and age distribution, pregnancy, and fecundity (fetuses:doe) rates of the female cohort recruited for this study. Additionally, in those summaries we've addressed winter and annual mortality rates (and their relations to the varying severities of winter weather conditions), specific causes of mortality (e.g., hunting, wolf predation, "miscellaneous"), and how the underlying age-specific hazard function (instantaneous probability of death) drove age-specific, seasonal, and annual survival rates of these females from birth to old age (up to 17.5 years old). To varying degrees we've presented preliminary descriptions of seasonal migration patterns of the collared deer; margins of understanding of safe capture, chemical immobilization, and handling; food habits; assessments of winter nutritional restriction and condition; as well as the territory sizes, survival, and specific fates of wolves ranging over the study sites.

Additionally, during the past 16 years, we've published a number of scientific and popular articles that have delved into many of the aforementioned topics in much greater detail than appropriate for the annual research summaries. Importantly, often as a result of collaborations with our Research Unit's biometricians (M. Riggs, J. Fieberg), these scientific articles and their associated in-depth analyses have allowed us to explore new, more scientifically rigorous, and illuminating analytical approaches to viewing the diverse data sets we were accumulating during this long-term study. These large data sets, analyses, and articles facilitated not only an increased understanding of numerous aspects of white-tailed deer ecology that we've been able to share with the scientific and management communities, but ultimately served as preparation for our most important upcoming data analyses relative to the long-term study's BACI design, primary goals, and objectives (described above). The many popular articles and presentations also allowed us to share current, interesting information synthesized from the data with numerous, diverse special interest groups, academic (K-12 and college-level) audiences, and the general public over the years.

During the past year, we've been concentrating our efforts on several tasks, including 1) completing data analyses and manuscript preparations of the type discussed above;
2) organizing the diverse, 15-year data sets in preparation for analyses specifically related to the BACI experimental design, and 3) updating the habitat composition layers for the 4 study sites relative specifically to vegetation succession and any other changes over the 15-year period. We describe below how we addressed vegetative succession and the types of changes we observed.

HABITAT ANALYSES AND UPDATES

Detailed baseline habitat analyses using mirror stereoscope interpretation of color infrared air photos (1:15,840) and geographic information systems (GIS, Arc/Info, ArcView) were completed early in the study (Figures 1 to 4). Forest stand types were classified according to their dominant 2-3 tree species, height, and winter canopy closure classes. Open habitat types, water sources, and roads were also delineated. The classification system was developed with the specific intent that it would facilitate an examination of potential relations between use of habitat types by white-tailed deer and their winter biological requirements. Page and space constraints herein would not allow us to present near the level of detail in Figures 1 through 4 of the actual habitat analyses, but the coverages depicted in these figures provide a general representation of the vegetative mosaics (highlighting conifer canopy closure class) that comprise the winter range of white-tailed deer in north-central Minnesota.

During the 15-year study period there was potential for natural and human-induced changes of the vegetation/habitat to occur. Because we are examining habitat use by study deer (via radio-telemetry) during each year, it was important to update the classification of the habitat layers of the 4 study sites to account for vegetative succession, as well as habitat destruction (e.g., by flooding). This was particularly important for types that were openings when the study began, as well as for conifer types with canopies that may have succeeded from a less dense closure class (A [< 40%] or a B [40-69%]) to a more dense class (B or a C [≥70%]).

We had current air photo coverage taken and rectified by fall 2006 at a scale of 1:15,840. We then were able to compare specific habitat types from the initial interpretation with the current coverage and determine whether significant change, particularly in conifer canopy closure classes, had occurred. Overall, on the Willow Lake control site, conifers increased 22.6% due to succession, with increases specifically in canopy closure classes A, B, and C of 29.7, 26.9, and 16.5%, respectively, from 1991 to 2005 (Table 1). Conversely, on the Dirty Nose Lake control site, conifers declined 22.7%, with specific changes of 20.5, 30.8, and 23.7% in canopy closure classes A, B, and C, respectively, from 1993 to 2005 (Table 1). At the Inguadona Lake treatment site, conifers were reduced by 18.2%, primarily associated with the mid-study treatment harvests, with 19.0 and 65.5% decreases in the A and C classes, respectively. However, canopy closure class B showed an overall 39.7% net increase. Finally, at the Shingle Mill Lake site, decreases in all classes (A, 8.2%; B, 27.5%; and C, 7.5%) accounted for an overall decrease in area of conifers of 12.9% (Table 1). Net changes in conifer canopy closure classes were attributable primarily to a combination of natural and human-induced sources, including: 1) destruction of stands by natural seasonal flooding; 2) planned, mid-study, experimental treatment conifer harvests; 3) non-study, planned timber harvests committed to by cooperators (primarily U. S. Forest Service) prior to initiation of the study; and 4) gradual natural succession during the 13-15 years each site was part of the long-term study. The specific effects of all of these sources of change will be quantified using GIS technology, air photo coverage, and documentation available from the different landownership cooperators. Furthermore, a number of mitigating circumstances will be considered as part of the statistical analyses ultimately conducted relative to the *a priori* objectives and goals of the study. For example, many of the non-study timber harvests conducted at the Dirty Nose Lake site occurred at the periphery of the site or in areas of the site not being used regularly by radio-monitored study deer. Consequently, it is presumed their impact on deer behavior and habitat use will not be as significant as it potentially could have been. Also, a number of the cuts were made in the last year or 2 of the post-treatment phase; if examination of the data suggests that these cuts were a source of disturbance or bias relative to study deer distribution or habitat use, data of the potentially affected deer will be excluded from certain analyses.

Detailed spatial and temporal analyses of annual deer use of habitat types on the study sites relative to specific winter weather conditions and overall winter severity will begin during the current year. A preliminary analysis has shown that during phases of the study associated with mild to average winter conditions, deer distribution over the study sites was more dispersed and use of vegetative cover was more variable, whereas when influenced by severe winter conditions, deer locations were more concentrated in dense conifer cover. Location data sets from about 35 GPS-radiocollared deer (programmed to collect data at 1 to 6-hour intervals over 24-hour daily periods) during 2000-2006, will be used to augment analyses of data collected from VHF-radiocollared deer and to enhance our understanding of deer use of winter cover types relative to varying weather conditions.

MONITORING ACTIVITY AND CAUSE-SPECIFIC MORTALITY OF WOLVES

Wolves were extirpated from the region of our study sites during the 1950s-1960s, but as their population recovered, they naturally expanded their range and became re-established in this region just 5 years prior to the initiation of our study in 1991. With this, data from our long-term study show that wolves are the primary source of natural mortality for female deer at least 0.5 years old. Presently, our study sites are near the leading southern edge of wolf range expansion. Since spring 1993, we have captured and radiocollared 57 (31 females, 26 males) wolves from 7 to 9 packs that range over the 4 study sites (Table 2). We radio-located these wolves from fixed-wing, year-around, in order to monitor their survival and investigate causes of mortality. Fates of these wolves include being killed by a variety of human-related and natural causes. During 2006, 1 radiocollared wolf was shot, 1 was snared, and 1 had its collar chewed off by other wolves.

ACKNOWLEDGMENTS AND PROJECT COOPERATORS

We gratefully acknowledge the time and diligent efforts of volunteers Richard Nelles and Rod Schloesser during the many winter and spring field seasons of this study. Ken Kerr and Carolin Humpal provided excellent laboratory support to the study. Approximately 145 enthusiastic, competent, and dedicated interns have made collection of winter field data possible, and we thank them for their efforts. We also thank Mark Lenarz, Group Leader for the Forest Wildlife Populations and Research Group, for his continued support. The valuable support and contributions of Don Pierce, Gary Anderson, John Tornes, Dan Hertle, and Paul Lundgren (DNR); Larry Olson, Jerry Lamon, Ellisa Bredenburg, and Amy Rand (Cass County Land Department); Kelly Barrett, John Casson, and Jim Gallagher (U. S. Forest Service); John Hanson and Cheryl Adams (Blandin Paper Co.); Carl Larson and Michael Houser (Potlatch Corp.) have been essential to the success of this study.

Table 1. The change in area (hectares) of conifer canopy closure classes (A [< 40%], B [40-69%], and C [≥70%]) on the 4 study sites of the white-tailed deer/conifer winter cover study, Grand Rapids-Remer-Longville, Minnesota, 1991 to 2005.

Site	1991/1993[a]	2005	Change
Willow Lake			
A	273	354	81
B	108	137	29
C	399	465	66
Total	780	956	176
Dirty Nose Lake			
A	493	392	-101
B	120	83	-37
C	97	74	-23
Total	710	549	-161
Inguadona Lake			
A	788	638	-150
B	239	334	95
C	278	96	-182
Total	1,305	1,068	-237
Shingle Mill Lake			
A	389	357	-32
B	273	198	-75
C	398	368	-30
Total	1,060	923	-137

[a] The Willow Lake and Inguadona Lake sites entered the study in 1991 as a control and treatment site, respectively, whereas the Dirty Nose Lake and Shingle Mill Lake sites entered the study in 1993 as a replicate control and treatment site, respectively.

Table 2. History of radiocollared gray wolves, north-central Minnesota, 1993 to 2006 (AD=adult, JUV=juvenile).

Wolf number	Pack	Capture date	Sex	Age class	Fate	Date
2093	WILLOW	MAY 1994	F	AD	SHOT	MAR 1996
2094	WILLOW	MAY 1994	M	AD	SHOT	NOV 1997
2056	WILLOW	MAY 1996	M	AD	NOT COLLARED	
2058	WILLOW	MAY 1996	F	AD	PROB. SHOT	AUG 1996
2052	NORTH INGY	MAY 1993	M	AD	UNKNOWN	DEC 1996
2087	SOUTH INGY	MAY 1993	F	AD	DIED FROM NATURAL CAUSES (EMACIATED, MANGEY)	AUG 2, 1998
2062	SOUTH INGY	AUG 1997	F	AD	SHOT	FEB 1998
2089	SHINGLE MILL	MAY 1993	F	AD	KILLED BY WOLVES	SEP 1994
2050	SHINGLE MILL	MAY 1993	M	AD	COLLAR CHEWED OFF	AUG 1993
2095	SHINGLE MILL	MAY 1995	F	AD	LOST SIGNAL	NOV 1995
2064	SHINGLE MILL	AUG 1996 MAY 2004	F	JUV	ON THE AIR	
2060	SHINGLE MILL	AUG 1996 JUL 1998 – RECAPTURED	F	JUV	LOST SIGNAL	FEB 1, 2000
2059	SHINGLE MILL	AUG 1996	M	JUV	LOST SIGNAL	OCT 1996
2085	DIRTY NOSE	MAY 1993	M	AD	DISPERSED	OCT 1993
2054	DIRTY NOSE	MAY 1993	M	AD	DISPERSED	SEP 1993
2091	DIRTY NOSE	APR 1994	F	AD	RADIO FAILED	MAY 27, 1998
2092	DIRTY NOSE	APR 1994	F	AD	RADIO FAILED	MAY 27, 1998
2096	MORRISON	MAY 1995	F	AD	DROPPED TRANSMITTER	NOV 22, 1996
2252	WILLOW	APR 1998	M	AD	ROAD-KILL	JUN 1998
2253	DIRTY NOSE	APR 1998	F	AD	UNKNOWN MORTALITY	AUG 3, 1998
2254	SHINGLE MILL	JUL 1998	M	AD	DROPPED TRANSMITTER	JUL 17, 2001
2066	MORRISON	JUL 1998	M	AD	KILLED BY WOLVES	JUN 4, 1999
2067	SHINGLE MILL	JUL 1998	M	JUV	COLLAR CHEWED OFF	JUL 1998
2068	HOLY WATER	JUL 1998	M	AD	LOST SIGNAL	AUG 27, 1999
2069	SOUTH INGY	JUL 1998	M	AD	LOST SIGNAL	DEC 4, 1998
2070	SOUTH INGY	JUL 1998	F	AD	LOST SIGNAL	JUL 3, 2002
2255	SOUTH INGY	JUL 1998	F	AD	DISPERSED	MAR 22, 1999
2256	DIRTY NOSE	AUG 1999	M	AD	DROPPED TRANSMITTER	JUL 6, 2001
2257	E. DIRTY NOSE	MAY 1999	M	AD	LOST SIGNAL	JAN 14, 2001
2258	WILLOW	AUG 1999	M	AD	DISPERSED	MAR 16, 2000
2259	DIRTY NOSE	JUL 2000	M	AD	DISPERSED	JUL 2001
2261	SHINGLE MILL	AUG 2000	M	AD	DROPPED TRANSMITTER	APR 10, 2002
2074	SOUTH INGY	AUG 2001	F	AD	SHOT BY FARMER	OCT 23, 2002
2073	SHINGLE MILL	AUG 2001	F	JUV	DROPPED TRANSMITTER	AUG 28, 2001
2071	SHINGLE MILL	SEP 2000	F	AD	SNARED	JAN 13, 2001
2139	SHINGLE MILL	AUG 2002 RECAPTURED JUN 2003	F	AD	SNARED	DEC 24, 2006
2141	INGUADONA	SEP 2002	F	JUV	DROPPED TRANSMITTER	SEP 22, 2002
2149	INGUADONA	MAY 2003	M	AD	SHOT	NOV 2003
2143	WILLOW	MAY 2003	M	AD	KILLED BY WOLVES	JUN 20, 2004
2144	MORRISON BROOK	JUN 2003	F	AD	SHOT	NOV 12, 2004
2145	INGUADONA	JUL 2003	F	AD	DIED, MANGE	JAN 3, 2004
2148	WILLOW	AUG 2003	F	AD	DISPERSED	DEC 2, 2003
2291	SMITH CREEK	AUG 2003	F	AD	LOST SIGNAL	MAR 28, 2005
2146	WILLOW	AUG 2003	F	JUV	DISPERSED	MAR 15, 2005
2262	DIRTY NOSE	SEP 2003	F	AD	SHOT	NOV 14, 2003
2263	SHINGLE MILL	MAY 2004	F	AD	SHOT	NOV 24, 2006
2264	DIRTY NOSE	MAY 2004	F	AD	ON THE AIR	
2266	WILLOW	MAY 2004	F	AD	ROAD-KILL	NOV 6, 2004
2267	INGUADONA	MAY 2004	M	AD	KILLED BY WOLVES	MAR 3, 2005
2268	INGUADONA	MAY 2004	M	AD	UNKNOWN MORTALITY	JAN 19, 2005
2269	WILLOW	MAY 2004	M	AD	DISPERSED	JUN 2004
2270	WILLOW	MAY 2005	M	AD	COLLAR CHEWED OFF	NOV 7, 2006
2271	SHINGLE MILL	MAY 2005	F	AD	ON THE AIR	
2272	UNAFFILIATED	MAY 2005	M	AD	ON THE AIR	
2273	INGUADONA	JUN 2005	F	AD	ROAD-KILL	FEB 8, 2006
2289	UNAFFILIATED	JUL 2005	M	AD	KILLED BY WOLVES	AUG 13, 2005
2290	SHINGLE MILL	AUG 2005	F	JUV	SLIPPED COLLAR	AUG 2005
2292	SHINGLE MILL	AUG 2005	M	JUV	SLIPPED COLLAR	AUG 2005

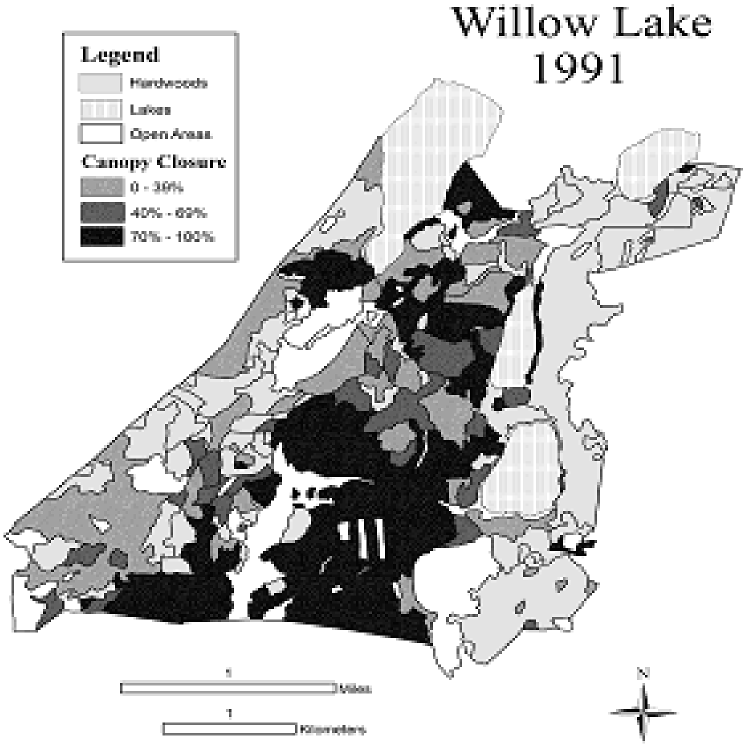

Figure 1. Habitat mosaic of Willow Lake control site, Grand Rapids-Remer-Longville, Minnesota, 1991–2005.

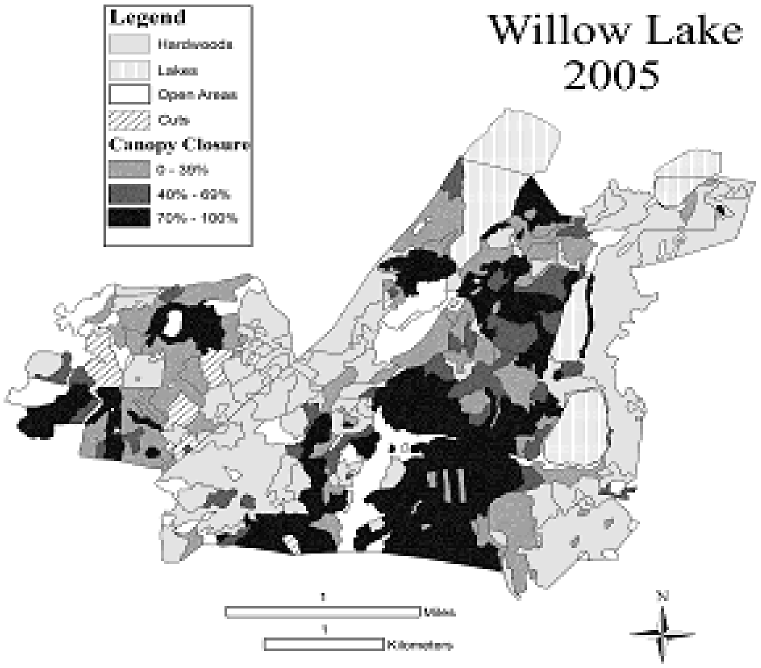

Figure 1. Continued.

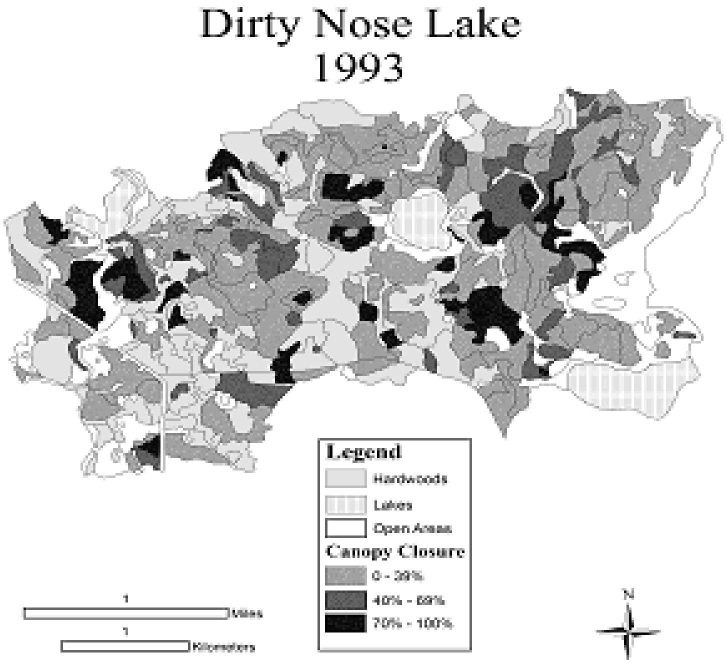

Figure 2. Habitat mosaic of Dirty Nose Lake control site, Grand Rapids-Remer-Longville, Minnesota, 1993–2005.

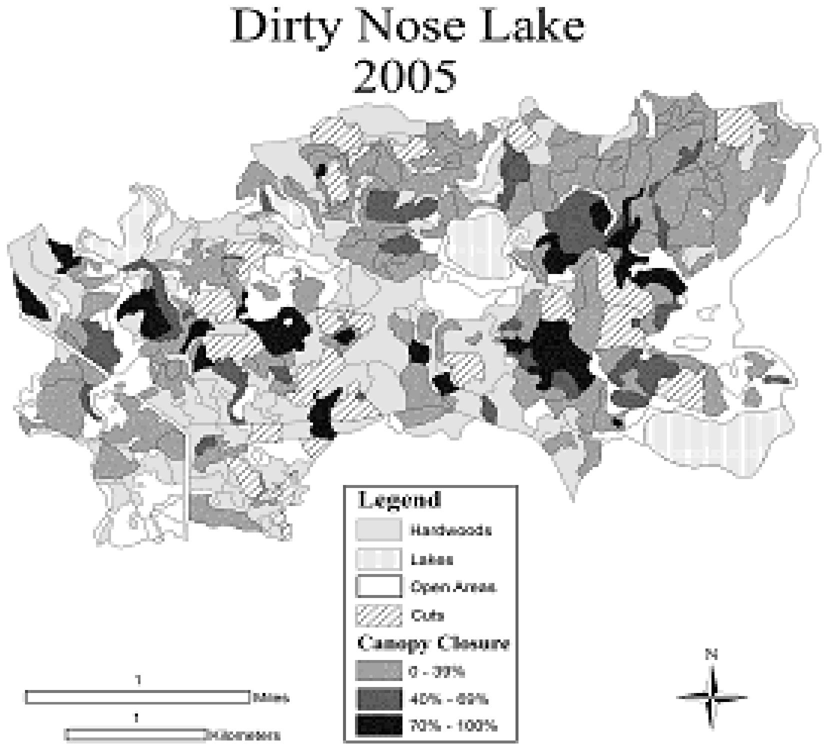

Figure 2. Continued.

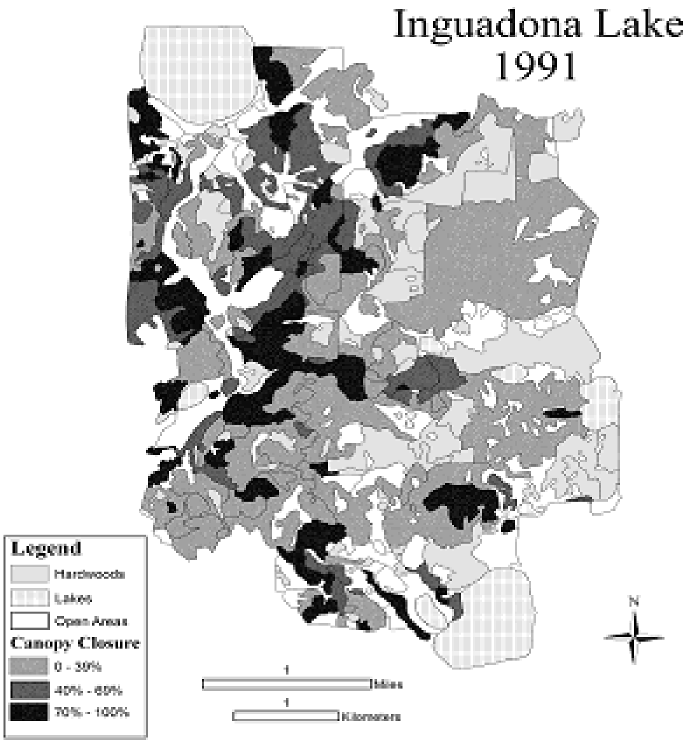

Figure 3. Habitat mosaic of Inguadona Lake treatment site, Grand Rapids-Remer-Longville, Minnesota, 1991–2005.

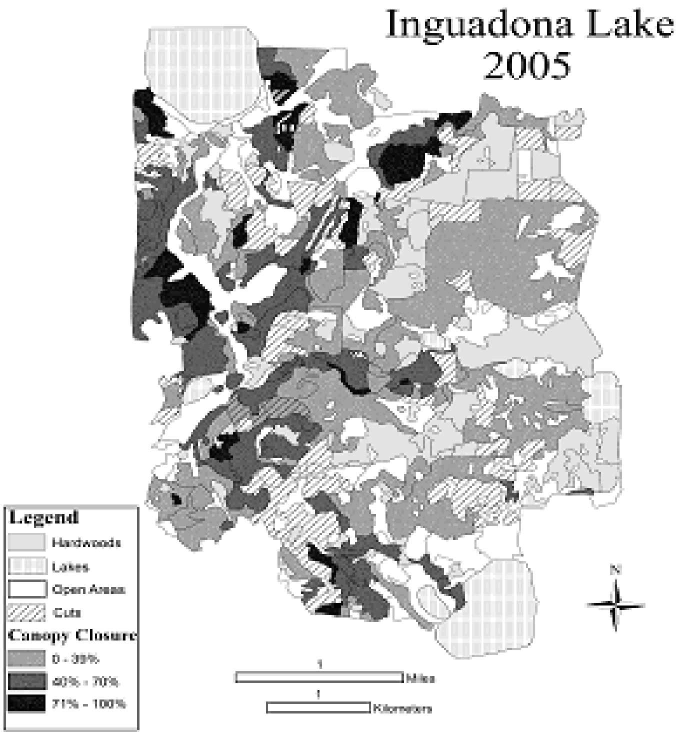

Figure 3. Continued.

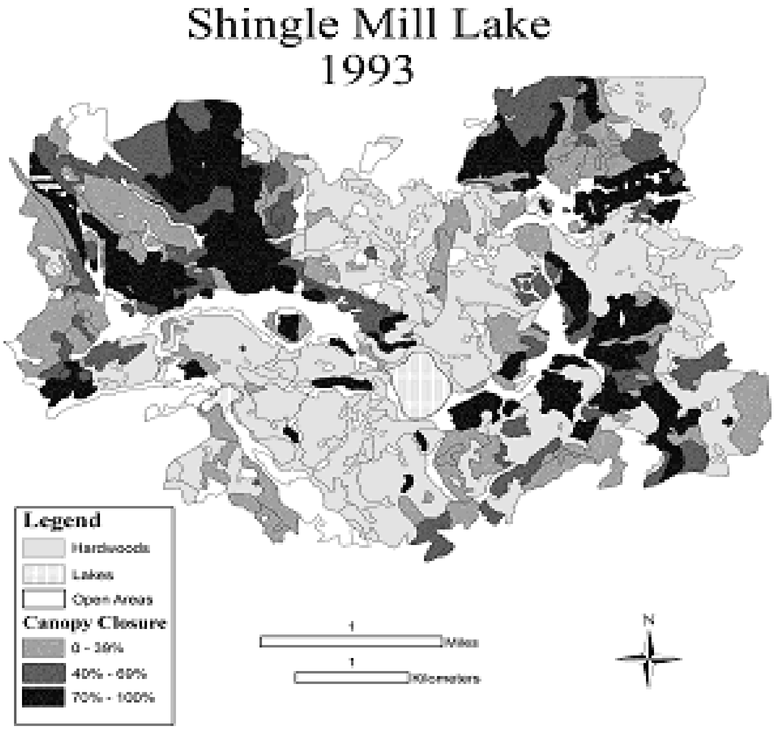

Figure 4. Habitat mosaic of Shingle Mill Lake treatment site, Grand Rapids-Remer-Longville, Minnesota, 1993–2005.

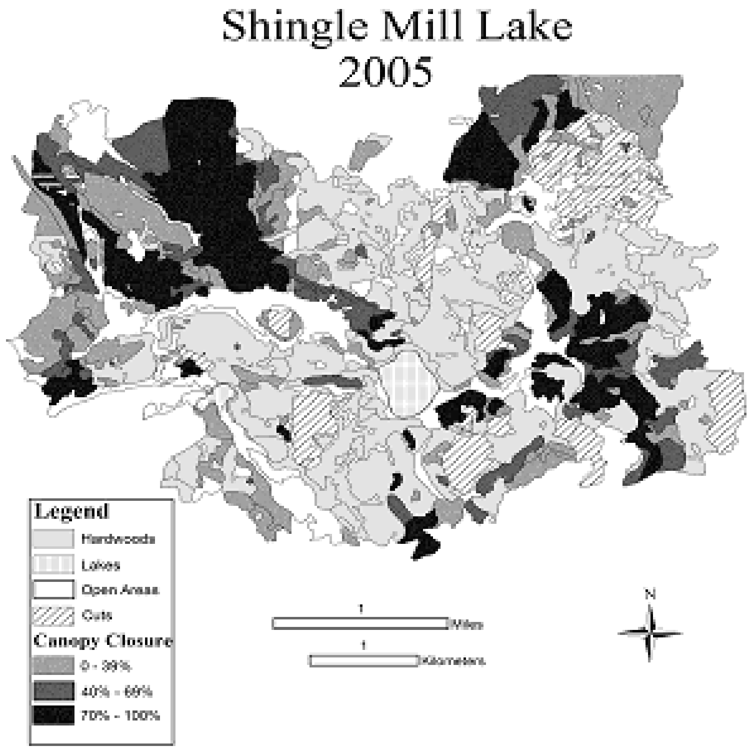

Figure 4. Continued.

MOOSE POPULATION DYNAMICS IN NORTHEASTERN MINNESOTA

Mark S. Lenarz, Michael W. Schrage[1], Andrew J. Edwards[2], and Michael E. Nelson[3]

SUMMARY OF FINDINGS

A total of 114 moose (54 bulls and 60 cows) have been captured and collared since beginning the study in 2002. As of 31 March 2007, 77 collared moose (41 bulls and 38 cows) have died. Annual mortality rates varied between sexes and among years, and generally were higher than found elsewhere in North America. Pregnancy rates of captured cows were variable, but higher than found in northwestern Minnesota. Radio collared moose were used to develop a "sightability model" to correct observations during the annual aerial moose survey. This model will likely improve the accuracy and precision of the aerial survey.

INTRODUCTION

Moose (*Alces alces)* formerly occurred throughout much of the forested zone of northern Minnesota, but today, most occur within 2 disjunct ranges in the northeastern and northwestern portions of the state. The present day northeastern moose range includes all of Lake and Cook counties, and most of northern St. Louis County. In recent years, population estimates based on aerial surveys suggest that moose numbers are relatively stable. That moose numbers in northeast Minnesota have not increased in recent years is an enigma. Research in Alaska and Canada has indicated that adult non-hunting mortality in moose populations is relatively low. When these rates are used in computer models to simulate change in Minnesota's northeastern moose population, moose numbers increase dramatically, counter to the trend indicated by aerial surveys. Several non-exclusive hypotheses can be proposed to explain this result: 1) average non-hunting mortality rate for moose in northeastern Minnesota is considerably higher and/or more variable than measured in previous studies; 2) recruitment rates estimated from the aerial surveys and used in the model are biased high; and/or 3) moose numbers estimated by the aerial survey are biased low.

OBJECTIVES

- Determine annual rates of non-hunting mortality for adult and calf moose in northeastern Minnesota;
- Determine annual rates of reproduction in northeastern moose; and
- Determine the proportion of moose observed during aerial surveys and the factors that influence observability.

METHODS

Moose were captured in southern Lake County and southwestern Cook County, an area within the Laurentian Upland and North Shore Highland subsections of Minnesota's Ecological Classification System.

---

[1] Fond du Lac Resource Management Division, 1720 Big Lake Road, Cloquet, Minnesota, 55720, USA

[2] 1854 Authority, 4428 Haines Road, Duluth, Minnesota, 55811, USA

[3] United States Geological Survey, U.S. Geological Survey, Northern Prairie Wildlife Research Center, Jamestown, North Dakota , 58401, USA

In 2002, moose were captured by netgunning from a helicopter. We found this to be an inefficient method in our chosen study area. Thus in 2003–2005, moose were immobilized with a combination of carfentanil and xylazine delivered by a dart gun from a helicopter. A radio-collar was attached, and blood, hair, and fecal samples were collected from each moose. Beginning in 2003, a canine tooth also was extracted for aging.

Mortality was determined by monitoring a sample of up to 78 radiocollared moose. The transmitter in each radio-collar contained a mortality sensor that increased the pulse rate (mortality mode) if it remained stationary for more than 6 hours. When a transmitter was detected in mortality mode, we located the moose and conducted a necropsy to determine, if possible, the cause of death. Mortality rates were calculated using Kaplan-Meier survival functions (Pollock et al. 1989). During the first year of the study, the GPS location of each moose was determined weekly from the air. Beginning in March 2003, GPS locations were determined for one-half of the moose each week, and a mortality check was conducted on the remaining moose. After moose were located on 30 or more occasions, only mortality checks were conducted.

Pregnancy was determined from serum and fecal progesterone levels (Haigh et al. 1981, Monfort et al. 1993). Beginning in 2004, all collared cows were located in late May to determine the number of calves born, and the following April to determine calf survival. In addition, the presence/absence of a calf with a collared cow was determined, when possible during the telemetry flights.

A sightability model (Anderson and Lindzey 1996, Quayle et al. 2001) was developed using observations of the radiocollared moose during the 2004-2007 aerial moose surveys. During the survey, test plots were identified that contained 1 or more radiocollared moose. Each test plot was surveyed using procedures identical to those used in the operational survey. If the collared moose was observed within the plot, a suite of covariates including environmental conditions, group size, and visual obstruction were recorded. If the collared moose were not observed, they were located using telemetry, and the same set of covariates were recorded. Logistic regression was used to determine which covariates should be included in the sightability model.

RESULTS

No additional moose were captured in 2007. A total of 114 moose (60 cows and 54 bulls) have been captured and radiocollared in northeastern Minnesota between February 2002 and February 2005 (Figure 1).

As of 31 March 2007, 79 collared moose (41 bulls and 38 cows) have died. The cause of death in 33 cases could be identified (15 hunter kill, 2 poached, 7 train/ car/truck collision, 7 wolf predation, 1 natural accident, and 1 bacterial meningitis). Three additional deaths were censored from the study because they occurred within 2 weeks of their capture (1 wolf predation and 2 unknown). We were unable to examine the remains of 5 moose. Two died within the Boundary Waters Canoe Area Wilderness and in 3 cases, we only found the radio-collar. Thirty-eight collared moose appear to have died from unknown, non-traumatic causes. In 16 cases, scavengers had consumed the carcasses, but evidence suggested that predators might not have killed them. In the remaining 22 cases, most had little or no body fat (rump, kidney, abdominal, or heart), and were often emaciated. Moose dying of unknown causes died throughout year (Figure 2). To date, samples from unknown cases have tested negative for Chronic Wasting Disease, Rabies, Eastern Equine Encephalitis, and West Nile Virus. Sera from captured moose were tested for Bovine Viral Diarrhea, Borreliosis (Lyme's disease), Leptospirosis, Malignant Catarrhal Fever, Respiratory Syncytial Virus, Parainfluenza 3, Infectious Bovine Rhinotracheitis, Epizootic Hemorrhagic Disease, and Blue Tongue. All test results were negative except for Borreliosis (21 of 64 serum samples had positive titers 1:320 or greater). Follow up tests on tissues of moose harvested by hunters did not reveal any evidence that moose were infected with Lyme's disease.

Annual non-hunting and total mortality varied considerably among years and between sexes (Table 1). It should be noted that only 7 bulls were collared during 2002. In both sexes, non-hunting mortality was substantially higher than documented for populations outside of Minnesota (generally 8 to 12%) (Ballard, 1991, Bangs 1989, Bertram and Vivion 2002, Kufeld and Bowden 1996, Larsen et al. 1989, Mytton and Keith 1981, Peterson 1977).

Serum samples from 30 additional collared moose were tested for the presence of P. tenuis-specific antibodies using an enzyme-linked immunosorbent assay procedure (ELISA) (Ogunremi et al. 1999). Eighteen (15 cows and 3 bulls) of the 109 collared moose tested were sero-positive for antibodies against P. tenuis. Subsequently, 5 died of unknown causes, 3 were likely killed by wolves, 1 was killed by a hunter, and 1 is listed as capture related because it died within 2 weeks of capture. Only 3 skulls were examined for the presence of P. tenuis; results were positive in one case, negative in the other 2.

Pregnancy rate between 2002 and 2005 was 84% (*n*=56). In 3 of the 4 years, pregnancy rate ranged between 92 and 100%; in 2003, pregnancy rate was only 57%. This contrasts with a pregnancy rate of only 48% between 1996 and 1999 in northwestern Minnesota (Murray et al. 2006).

Survival of calves born to radiocollared cows remained constant since 2004 when calf surveys were initiated. In late April 2005, 12 of the 26 calves born to radiocollared cows in May 2004 were still alive (46% survival). In late April 2006, 41% of calves born the previous year were still alive. As of late January 2007, 56% of calves born in May 2006 were still alive. Annual calf survival in northwestern Minnesota averaged 66% (53-81%, Murray et al. 2006).

In January 2007, radio collared moose were located 49 times on test plots in the process of developing a sightability model. In 19 cases, the collared moose was observed using the standard survey protocol. In 30 cases, the collared moose was not observed, and telemetry had to be used to locate the collared moose. Since 2004, telemetry was required to locate 50% of the moose in test plots which suggests that, on average, only half of the moose are observed from the transects during the operational survey. Analyses of the data are ongoing and a final sightability model will be available for use during the 2008 moose survey.

ACKNOWLEDGMENTS

We thank the collaborators in this study including Glenn DelGiudice and Barry Sampson from the Department of Natural Resource's Forest Wildlife Populations and Research Group, and Jim Rasmussen from the Minnesota Zoo for their assistance in capturing moose and collecting biological samples. We also thank Al Buchert, Mike Trenholm, Brad Maas, and John Heineman from the DNR's Enforcement Division for their piloting skills throughout the project. John Fieberg and John Giudice were essential to the development of the sightability model.

LITERATURE CITED

Anderson, C. R., and F. G. Lindzey. 1996. Moose sightability model developed from helicopter surveys. Wildlife Society Bulletin 24:247-259.

Ballard, W. B., J. S. Whitman, and D. J. Reed. 1991. Population dynamics of moose in south-central Alaska. Wildlife Monograph 114.

Bangs, E. E., T. N. Bailey, and M. F. Portner. 1989. Survival rates of adult female moose on the Kenai Peninsula, Alaska. Journal of Wildlife Management 53:557-563.

Bertram, M. R., and M. T. Vivion. 2002. Moose mortality in eastern interior Alaska. Journal of Wildlife Management 66:747-756.

Murray, D. L., e. W. Cox, W. B. Ballard, H. A. Witlaw, M. S. Lenarz, T. W. Custer, T. Barnett, and T. K. Fuller. 2006. Pathogens, nutritional deficiency, and climate influences on a declining moose population. Wildlife Monographs 166.

Drew, M. L., and W. M. Samuel. 1986. Reproduction of the winter tick, Dermacentor albipictus, under field conditions in Alberta, Canada. Canadian Journal of Zoology 64:714-721.

Gasaway, W. C., S. D. DuBois, D. J. Reed, and S. J. Harbo. 1986. Estimating moose population parameters from aerial surveys. Biological Papers University of Alaska, Fairbanks. Number 22, Fairbanks, Alaska, USA.

Haigh, J. C., E. W. Kowal, W. Runge, and G. Wobeser. 1982. Pregnancy diagnosis as a management tool for moose. Alces 18: 45-53.

Kufeld, R. C., and D. C. Bowden. 1996. Survival rates of Shiras moose (Alces alces shirasi) in Colorado. Alces 32: 9-13.

Larsen, D. G., D. A. Gauthier, and R. L. Markel. 1989. Cause and rate of moose mortality in the southwest Yukon. Journal of Wildlife Management 53:548-557.

Mech, L. D. 1986. Wolf population in the central Superior National Forest, 1967-1985. U.S. Forest Service Research Paper. NC-270. Duluth, Minnesota, USA.

Monfort, S. L., C. C. Schwartz, and S. K. Wasser. 1993. Monitoring reproduction in moose using urinary and fecal steroid metabolites. Journal of Wildlife Management. 57:400-407.

Mytton, W. R., and L. B. Keith. 1981. Dynamics of moose populations near Rochester, Alberta, 1975-1978. Canadian Field-Naturalist 95:39-49.

Ogunremi, O., M. Lankester, J. Kendall, and A. Gajadhar. 1999. Serological diagnosis of Parelaphostrongylus tenuis infection in white-tailed deer and identification of a potentially unique parasite antigen. Journal of Parasitology 85(1): 122-127.

Peterson, R. O. Wolf ecology and prey relationships on Isle Royale. National Park Service Scientific Monograph. 88. Washington, D.C., USA.

Peek, J. M., D. L. Urich, and R. J. Mackie. 1976. Moose habitat selection and relationships to forest management in northeastern Minnesota. Wildlife Monograph 48.

Pollock, K. H., C. T. Moore, W. R. Davidson, F. E. Kellogg, and G. L. Doster. 1989. Survival rates of bobwhite quail based on band recovery analyses. Journal of Wildlife Management 53:1-6.

Quayle, J. F., A. G. MacHutchon, and D. N. Jury. 2001. Modeling moose sightability in south central British Columbia. Alces 37:43-54.

Samuel, W. M. and D. A. Welch. 1991. Winter ticks on moose and other ungulates: factors influencing the population size. Alces 27:169-182.

Table 1. Annual non-hunting and total mortality of collared moose. Number of collared moose in sample at beginning of calendar year is listed in parentheses.

Non-Hunting Mortality				Total Mortality			
Year	Bulls	Cows	Combined	Year	Bulls	Cows	Combined
2002	0% (7)	29% (17)	21% (24)	2002	14% (7)	29% (17)	25% (24)
2003	27% (27)	23% (33)	24% (60)	2003	33% (27)	23% (33)	28% (60)
2004	14% (23)	6% (35)	9% (59)	2004	35% (23)	6% (35)	17% (59)
2005	16%(35)	19%(43)	17%(78)	2005	24%(35)	19%(43)	23%(78)
2006	30%(25)	37%(35)	34%(60)	2006	41%(25)	37%(35)	39%(60)

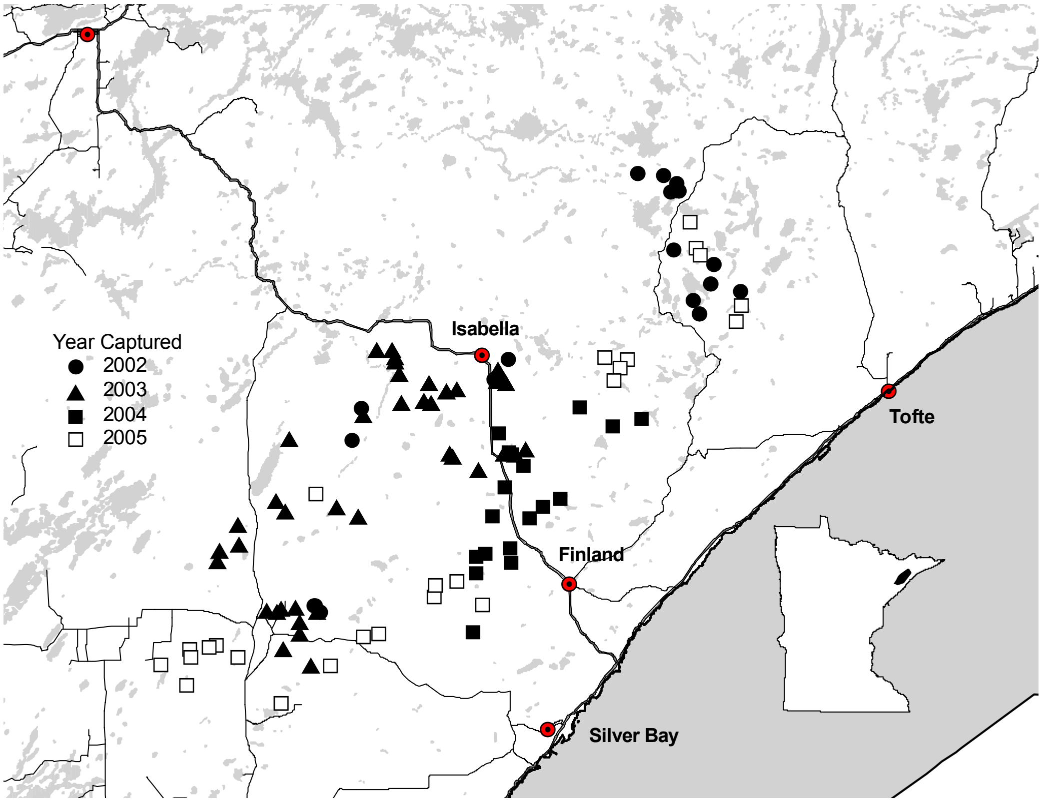

Figure 1. Capture locations of moose radio collared, 2002-2005.

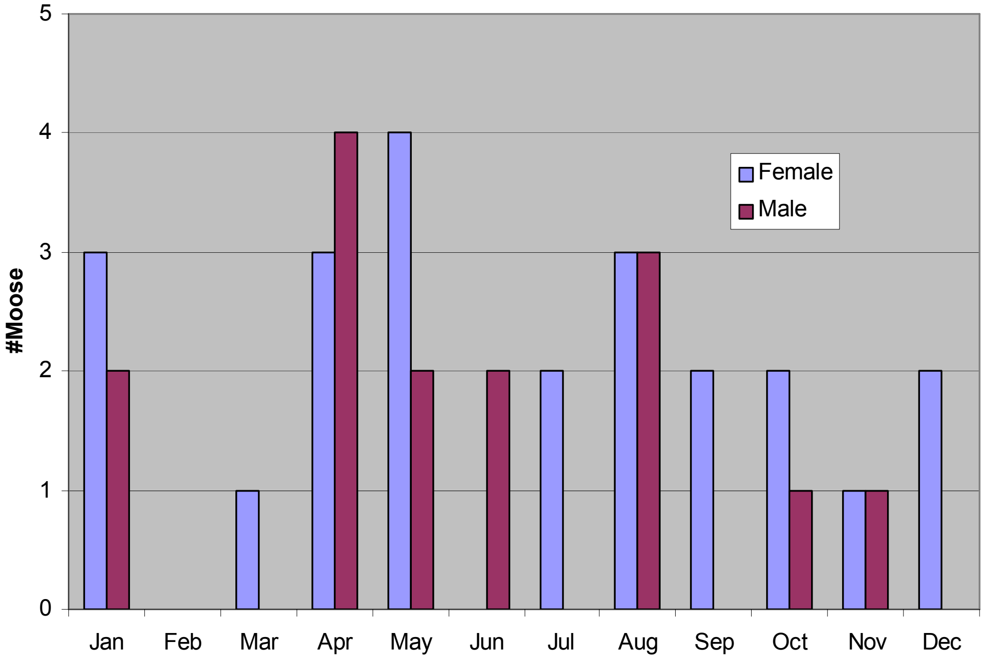

Figure 2. Timing of unknown mortality by sex in radiocollared moose in northeastern Minnesota study area.

ECOLOGY AND POPULATION DYNAMICS OF BLACK BEARS IN MINNESOTA

David L. Garshelis, Karen V. Noyce, and Pamela L. Coy

SUMMARY OF FINDINGS

During April 2006–March 2007, we monitored 35 radiocollared black bears (*Ursus americanus*) at 3 study sites spanning the black bear's geographic range, north to south in Minnesota: Voyageurs National Park (VNP, northern), Chippewa National Forest (CNF; central), Camp Ripley (southern). Mortality data were obtained through collars turned in by hunters or collars tracked to carcasses. Hunting continues to be the largest source of mortality of collared bears, even though hunters were asked not to shoot bears with bright orange radiocollars. In fact, the hunting mortality rate of collared bears was higher this year (62% of collared bears killed in the CNF) than in any year since our study began in 1981. Reproductive output was highest in the southern study site and declined northward in response to diminishing food. All sites have exhibited largely synchronous reproduction, with high cub production occurring in odd-numbered years. This pattern continued in 2007.

INTRODUCTION

A paucity of knowledge about bear ecology and effects of harvest on bear populations spurred the initiation of a long-term telemetry-based bear research project by the Minnesota Department of Natural Resources (MN DNR) in the early 1980s. For the first 10 years, the study was limited to the Chippewa National Forest (CNF), near the center of the Minnesota bear range. After becoming aware of significant geographic differences within the state in sizes, growth rates, and productivity of bears, apparently related to varying food supplies, we started other satellite bear projects in different study sites. Each of these began as graduate student projects, supported in part by the MN DNR. After completion of these student projects, we continued studies of bears at Camp Ripley Military Reserve, near the southern fringe of the Minnesota bear range, and in Voyageurs National Park (VNP), on the Canadian border.

These study sites differ enormously. The CNF is one of the most heavily hunted areas of the state, with large public (national, state, and county), heavily-roaded forests dominated by aspen of varying ages. Camp Ripley is unhunted, but bears may be hunted when they range outside, which they often do in the fall, as the reserve is only 6–10 km wide. Oaks are far more plentiful here than in the 2 study sites further north. VNP, being a national park, is also unhunted, but again bears may be hunted when they range outside. Soils are shallow and rocky in the park, and foods are generally least plentiful of the 3 sites.

OBJECTIVES

- Monitor temporal and spatial variation in cub production and survival;
- Monitor rates and sources of mortality; and
- Compare body condition indices across sites and years (not covered in this report).

METHODS

Radiocollars with breakaway and/or expandable devices were attached to bears either when they were captured in barrel traps during the summer or when they were handled as yearlings in the den of their radiocollared mother. Limited trapping has been conducted in recent years. However, during December–March, all radio-instrumented bears were visited once or twice a year at their den site. Bears in dens were immobilized with an intramuscular injection of Telazol, administered with a jab stick or Dan-Inject dart gun. Bears were then removed from the den for processing, which included changing or refitting the collar, or attaching a first collar on yearlings, measuring, weighing, and obtaining blood and hair samples. We also measured biolelectrical impedance (to calculate percent body fat) and vital rates of all immobilized bears. Additionally, with the cooperation of investigators from the University of Minnesota (Dr. Paul Iaizzo) and Medtronic (Dr. Tim Laske), heart condition was measured with a 12-lead EKG and ultrasound on a select sample of bears. Bears were returned to their den after processing. Reproduction was assessed by observing cubs in dens of radiocollared mothers. Cubs were not immobilized, but were removed from the den after the mother was drugged, then sexed, and weighed. We evaluated cub mortality by examining dens of these same mothers the following year: Cubs that were not present as yearlings with their mother were presumed to have died.

During the non-denning period we monitored mortality of radio-instrumented bears from an airplane approximately once each month. We listened to their radio signals, and if a pulse rate was in mortality mode (no movement of the collar in >4 hours), we tracked the collar on the ground to locate the dead animal or the shed radiocollar. If a carcass was located, we attempted to discern the cause of death.

RESULTS AND DISCUSSION

Since 1981 we have handled >800 individual bears and collared nearly 500. As of April 2006, the start of the current year's work, we were monitoring 41 collared bears: 17 in the CNF, 16 at Camp Ripley, and 8 in VNP. However, 6 bears dropped their collars so were lost from the sample.

Mortality

Legal hunting has been the predominant cause of mortality among radiocollared bears from all 3 study sites (Table 1). In previous years, hunters were encouraged to treat collared bears as they would any other bear so that the mortality rate of collared bears would be representative of the population at large. With fewer collared bears left in the study, and the focus now primarily on reproduction rather than mortality, we sought to protect the remaining sample of bears. We asked hunters not to shoot radiocollared bears, and we fitted these bears with bright orange collars so hunters could more easily see them in dim light conditions. Nevertheless, 8 of 13 bears (62%) with functional collars in the CNF were killed during this year's hunt (Sep-Oct, 2006). This includes 1 bear that was wounded and lost. All were females, aged 1–7 years. This is the highest rate of hunter-caused mortality observed in this study over the past 26 years, leaving only 5 collared bears at this long-term study site. Additionally 1 of 8 collared bears was killed by a hunter outside Camp Ripley, and 3 other Camp Ripley bears could not be found after the hunt; either these bears were killed and not reported or their collar failed (wide-ranging unsuccessful aerial searches for them seemed to exclude the possibility that they simply moved far out of the area). No VNP bears were killed by hunters this year.

Other human-related mortalities included 1 bear whose cut-off collar was found in a river, and 2 male bears from Camp Ripley that were hit by vehicles on roads outside the reserve. A much higher proportion of the deaths at Camp Ripley (30% of those with known cause) were a result of collisions with vehicles than at the other 2 sites (4–6%).

Only 1 natural mortality was observed, a yearling in VNP. We do not know why this bear died (starvation seemed unlikely). However, it is interesting that of only 12 bears tracked in VNP over the past 2 years, 3 died of natural mortality, whereas no natural mortalities occurred at the other 2 study sites during this time (and few natural mortalities occurred earlier in the study; Table 1).

Reproduction

For the past decade, collared bears on all of our study sites had strong reproductive synchrony, with low cub production in even-numbered years and high production in odd-numbered years (Figure 1). This synchrony matches that exhibited in the age structure of the statewide bear harvest. It appears to have stemmed from a very poor food year in 1995, causing low cub production in 1996, followed by a good food year in 1996, yielding high cub production in 1997. Since then, all years have had average or above-average summer and fall foods, so the synchronous reproduction has persisted because nearly all bears have maintained a 2-year reproductive cycle. Of 13 mature bears checked in dens in March, 2007, 10 (77%) had cubs, 2 had yearlings, and 1 that was due to have cubs (because she had cubs in 2005) failed to produce. Reproductive synchrony appears to be strongest in VNP and least in Camp Ripley (Figure 1). In part this is because a large proportion of Camp Ripley bears produced their first cubs at 3 years old, which is out of synch with their mother.

Bears at Camp Ripley, where hard mast (especially oak) is abundant, grow faster and thus have an earlier age of first reproduction than at the other 2 study sites. This is reflected in the reproductive rates (cubs born/female) of 4–6 year-old females, which was twice as high at Camp Ripley as at VNP (where no bears produced cubs at 4 years old), and intermediate at CNF (Table 2). This north-south gradient was also apparent in the reproductive rates of older bears, due to fewer missed reproductive opportunities in Camp Ripley (the first bear that did not produce cubs on a 2-year cycle was observed this year) and more whole-litter losses and skipped litters at VNP (Table 2). If no bears skipped litters, all would be on a 2-year reproductive cycle, and thus 50% of females would have cubs, on average, per year. The proportion of females with cubs was lowest in VNP and highest in Camp Ripley (where it exceeded 50% as an artifact of sampling; Table 2).

Mean litter size was somewhat higher in the central CNF site (2.6 cubs/litter; Table 3) than at the other sites (2.3 cubs/litter; Tables 4–5). However, counting only litters where at least 1 cub survived 1 year, litter sizes were remarkably similar across areas for 7+ year-old bears (mainly multiparous mothers; Table 2). In all areas, litter size was smaller for younger females, nearly all of which were first-time mothers (Table 2). Notably, 2 collared bears produced litters of 5 cubs this year; of the 222 litters that we examined previously, only 2 other 5-cub litters were observed, both by the same female (in 1982 and 1984).

Only 1 bear was monitored through its age of senescence. She had her last cubs at age 25 (in 1999). This bear is still being monitored, now aged 33. This year, 1 bear in CNF had cubs at 22 years of age and 1 at VNP had cubs at 21.

Average sex ratio of cubs shortly after birth was slightly male-biased (52%) and virtually identical among all sites. Observed year-to-year variation in cub sex ratios (Tables 3–5) was likely attributable to sampling error. In all areas, the mortality rate of male cubs was higher than (1.5-2x) that of females. Overall, cub mortality appeared to be lower in CNF (18%; Table 3) than in the other 2 sites (26–28%; Tables 4–5). The difference, though, was not statistically significant (CNF vs. VNP and Camp Ripley combined, $P = 0.08$).

Cub production and cub mortality did not show an upward or downward trend during our 26 years of monitoring at CNF (or since 1999 at the other 2 sites). However, statewide bear harvests have shown an increasing proportion of yearlings, suggesting a changing statewide age structure, or possibly changing selectivity by hunters (with varying numbers of hunters).

FUTURE DIRECTION

We plan to continue monitoring bears on these 3 study sites, although sample sizes have been greatly diminished by the exceedingly high harvest of collared bears in the CNF this year. We are also initiating a new study site at the edge of the range in northwestern

Minnesota. This study will be led by a PhD student from the University of Minnesota. Our goal there is to assess the factors that may limit range expansion, including highly fragmented forested habitat, lack of agricultural crops that bears can eat, and human-related mortality. Bears will be outfitted with Global Positioning System (GPS) collars to document their fine-scale habitat use. Three bears whose dens were found by local people were collared this March. Comparisons will be made between these GPS-collared bears at the northwestern edge of the range and bears with GPS collars that have been monitored for the past several years at Camp Ripley, along the southern edge of the range.

ACKNOWLEDGMENTS

We thank the collaborators in this study: Brian Dirks and Julie DeJong at Camp Ripley, Steve Windels and Jen Fox at Voyageurs National Park, Paul Iaizzo at the University of Minnesota, and Tim Laske at Medtronic, Inc. We also thank the staff at Camp Ripley for trapping bears. Numerous volunteers assisted with den work.

Table 1. Causes of mortality of radiocollared black bears ≥1 year old from the Chippewa National Forest (CNF), Camp Ripley, and Voyageurs National Park (VNP), Minnesota, 1981–2006. Bears did not necessarily die in the area where they usually lived (e.g., hunting was not permitted within Camp Ripley or VNP, but bears were killed by hunters when they traveled outside these areas).

	CNF	Camp Ripley	VNP
Shot by hunter	219	10	10
Likely shot by hunter[a]	8	1	0
Shot as nuisance	22	2	1
Vehicle collision	12	7	1
Other human-caused death	9	0	0
Natural mortality	7	3	4
Died from unknown causes	3	1	0
Total deaths	280	24	16

[a] Lost track of during the hunting season. Does not include 3 bears lost at Camp Ripley in 2006 (see text).

Table 2. Reproductive rates (cubs/female), mean litter size, and proportion of females with cubs (for all measures, counting only litters in which at least 1 cub survived 1 year) in winter dens (March) in VNP (1997–2007), CNF (1981–2007) and Camp Ripley (1991–2007) (*n* = 4+ year-old female-years of observation). Reproduction increased from north (VNP) to south (Camp Ripley).

	VNP (*n* = 56)			**CNF** (*n* = 402)			**Camp Ripley** (*n* = 39)		
Age of female	Repro rate	Litter size	Prop w/ cubs	Repro rate	Litter size	Prop w/ cubs	Repro rate	Litter size	Prop w/ cubs
4–6 yrs	0.59	2.0	29	0.84	2.3	37	1.28	2.3	56
7–25 yrs	1.15	2.7	44	1.33	2.8	48	1.52	2.7	57
4–25 yrs	0.98	2.6	39	1.15	2.6	44	1.41	2.5	56

Table 3. Black bear cubs examined in dens of radiocollared mothers in or near the Chippewa National Forest during March, 1982–2007.

Year	Litters checked	No. of cubs	Mean cubs/litter	% Male cubs	Mortality after 1 yr[a]
1982	4	12	3.0	67%	25%
1983	7	17	2.4	65%	15%
1984	6	16	2.7	80%	0%
1985	9	22	2.4	38%	31%
1986	11	27	2.5	48%	17%
1987	5	15	3.0	40%	8%
1988	15	37	2.5	65%	10%
1989	9	22	2.4	59%	0%
1990	10	23	2.3	52%	20%
1991	8	20	2.5	45%	25%
1992	10	25	2.5	48%	25%
1993	9	23	2.6	57%	19%
1994	7	17	2.4	41%	29%
1995	13	38	2.9	47%	14%
1996	5	12	2.4	25%	25%
1997	9	27	3.0	48%	23%[b]
1998	2	6	3.0	67%	0%
1999	7	15	2.1	47%	9%
2000	2	6	3.0	50%	17%
2001	5	17	3.4	76%	15%
2002	0	0	—	—	—
2003	4	9	2.3	22%	0%
2004	5	13	2.6	46%	33%
2005	6	18	3.0	33%	28%
2006	2	6	3.0	83%	33%
2007	2	6	3.0	67%	—
Overall	172	449	2.6	52%	18%

[a] Cubs that were absent from their mother's den as yearlings were considered dead. Blanks indicate no cubs were born to collared females.

[b] Excluding 1 cub that was killed by a hunter after being translocated away from its mother.

Table 4. Black bear cubs examined in dens of radiocollared mothers in Camp Ripley Military Reserve during March, 1992–2007.

Year	Litters checked	No. of cubs	Mean cubs/litter	% Male cubs	Mortality after 1 yr[a]
1992	1	3	3.0	67%	0%
1993	3	7	2.3	57%	43%
1994	1	1	1.0	100%	—
1995	1	2	2.0	50%	0%
1996	0	0	—	—	—
1997	1	3	3.0	100%	33%
1998	0	0	—	—	—
1999	2	5	2.5	60%	20%
2000	1	2	2.0	0%	0%
2001	1	3	3.0	0%	33%
2002	0	0	—	—	—
2003	3	8	2.7	63%	33%
2004	1	2	2.0	50%	—
2005	3	6	2.0	33%	33%
2006	2	5	2.5	60%	—
2007	3	7	2.3	43%	—
Overall	23	54	2.3	52%	26%

[a] Cubs that were absent from their mother's den as yearlings were considered dead. Blanks indicate no cubs were born to collared females or collared mothers with cubs died before the subsequent den visit. Presumed deaths of orphaned cubs are not counted here as cub mortality.

Table 5. Black bear cubs examined in dens of radiocollared mothers in Voyageurs National Park during March, 1999–2007.

Year	Litters checked	No. of cubs	Mean cubs/litter	% Male cubs	Mortality after 1 yr[a]
1999	5	8	1.6	63%	20%
2000	2	5	2.5	60%	80%
2001	3	4	1.3	50%	75%
2002	0	0	—	—	—
2003	5	13	2.6	54%	8%
2004	0	0	—	—	—
2005	5	13	2.6	46%	20%
2006	1	2	2.0	50%	0%
2007	3	9	3.0	44%	—
Overall	24	54	2.3	52%	28%

[a] Cubs that were absent from their mother's den as yearlings were considered dead. Blanks indicate no cub mortality data because no cubs were born to collared females.

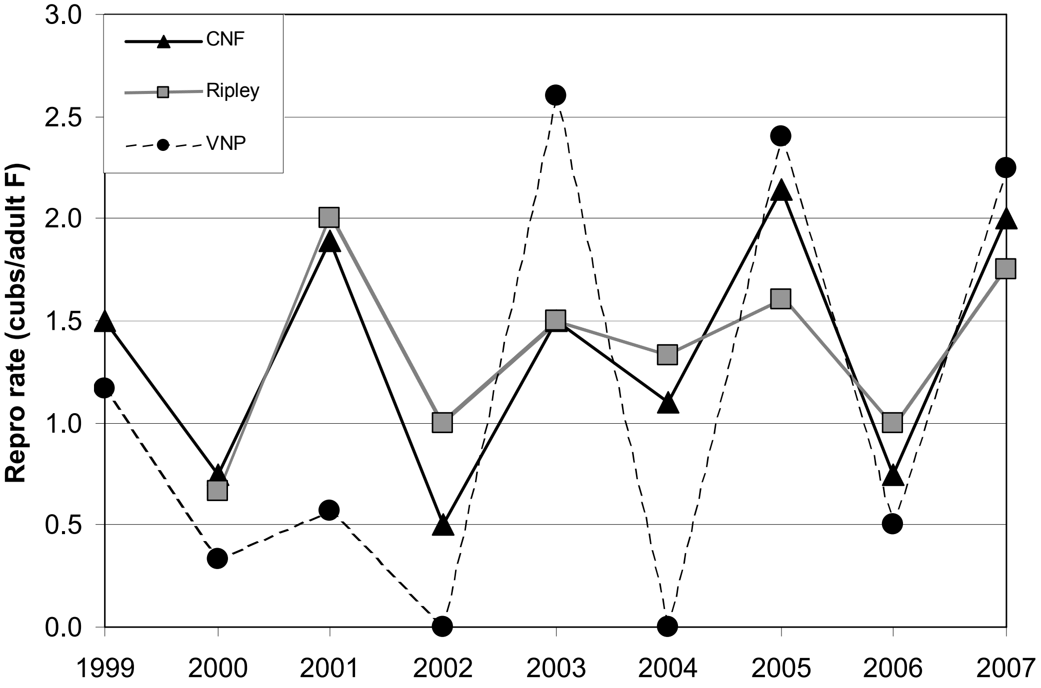

Figure 1. Reproductive rates (cubs per 4+ year-old female; counting only litters where at least 1 cub survived 1 year) of bears on the Chippewa National Forest (CNF), Camp Ripley, and Voyageurs National Park (VNP). All areas exhibited the same reproductive synchrony, although the pattern was most dramatic in VNP, at the northern extreme of the bear range, and weakest at Camp Ripley, at the southern edge of the range.

IDENTIFYING PLOTS FOR SURVEYS OF PRAIRIE-CHICKENS IN MINNESOTA

Michael A. Larson

SUMMARY OF FINDINGS

To explore potential improvements in surveys of prairie-chickens in Minnesota, I developed this study to determine landscape-scale characteristics associated with plots of land occupied by prairie-chicken leks and to evaluate potential within-year sources of variation in the probability of detecting a prairie-chicken lek, if one is present. The study area consisted of nearly the entire range of prairie-chickens in northwest Minnesota. Observers visited randomly selected Public Land Survey (PLS) sections (~259 ha) 3 times during April and early May of 2005 to detect leks. Confirmatory analyses indicated that wind speed and cloud cover were negatively correlated with the probability of detecting a lek. Road density was positively correlated with the probability of detection, but it was negatively correlated with the probability of a section being occupied by a lek. Exploratory analyses also revealed positive correlations between occupancy and both grass cover as a proportion of area and the area of all cover types considered as habitat and a negative correlation between occupancy and distance to the nearest known lek from the previous year. Approximately 13% of sections in the study area were occupied by a lek, but the precision of the estimated abundance of occupied sections was low ($\hat{Y}$ = 420, SD = 270).

INTRODUCTION

Nearly all methods for monitoring populations of greater prairie-chickens (*Tympanuchus cupido pinnatus*), including those currently employed by the Minnesota Department of Natural Resources (MN DNR), depend upon locating leks, or concentrations of the birds at their arenas for breeding displays (i.e., booming grounds), during spring. Surveying a statistically valid sample of leks requires identifying all areas where leks may occur and then sampling to find a number of plots occupied by active leks. The range of prairie-chickens in Minnesota covers approximately 10,000 km^2, so a major limitation to monitoring leks of prairie-chickens is determining where to survey within that range.

The availability of Geographic Information System (GIS) technology and databases of spatially explicit land cover have made it feasible to use landscape-scale habitat criteria to identify areas where leks may occur. Although land cover associated with prairie-chicken leks in Minnesota and Wisconsin have been quantified during previous studies (Merrill et al. 1999, Niemuth 2000, 2003), interpretation and application of those data are problematic. In particular, the previous studies were based on a case–control sampling design, which does not allow inferences about relative probabilities of occurrence (Keating and Cherry 2004). In addition, they did not select active leks randomly or verify nonuse at the randomly selected control locations.

Inferences about trends in the abundance of grouse throughout the state require statistically valid samples of survey locations from defined areas in which the species may occur. This study builds upon existing knowledge of landscape-scale habitat criteria that may be useful for identifying plots where prairie-chicken leks may occur, thereby dramatically reducing the area needed to be included in monitoring programs. It also serves as a pilot project for a new survey design that may prove to be more efficient than current survey methods for detecting changes in the abundance of prairie-chickens. Results of this study may benefit management programs for prairie-chickens by improving the quality of inferences drawn from spring surveys and developing resource selection functions for using landscape characteristics to estimate the relative probability of an area being occupied by a lek.

OBJECTIVES

- To determine landscape-scale characteristics associated with plots of land occupied by prairie-chicken leks in Minnesota; and
- To evaluate potential within-year sources of variation in the probability of detecting prairie-chicken leks in Minnesota.

METHODS

Study Area

Prairie-chickens occur in 3 distinct ranges in Minnesota. A study area was established in the northwest prairie-chicken range because the northwest range contained the largest population of prairie-chickens, was where the hunting permit areas were, and was the focus of all recent prairie-chicken monitoring efforts by the MN DNR. The study area included the northern 96% of the northwest range as defined by Giudice (2004) based upon land type associations of the Ecological Classification System (Figure 1). The size of the study area was limited only by a maximum distance of 90 km to the southeast of Moorhead, where the southernmost field technicians resided.

Notation

Methods for this study were based on analytical techniques for estimating the probability of site occupancy (MacKenzie et al. 2002). Throughout this report notation follows that of MacKenzie et al. (2002): ψ, probability that a sample plot is occupied by a lek; p, probability of detecting a lek within a sample plot, given that the plot is occupied; N, number of sample plots in a study area; T, number of surveys, or distinct sampling intervals during which all plots are visited once; the "hat" character (e.g., $\hat{\psi}$) denotes the estimated value of a quantity; and c, the probability of detecting a lek during visits that occur after a lek already has been detected within a plot (i.e., recapture).

Sampling Design

A sampling unit, or plot, was defined as a PLS section, most of which were 1.6- × 1.6-km squares (i.e., 259 ha = 1 mi^2). In portions of the prairie-chicken range in Minnesota some PLS sections were rectangular and much smaller than 259 ha. Variability in the size of plots was accounted for by the possible inclusion of habitat area within a plot as a covariate for ψ. The size of plots roughly corresponded to home range sizes of prairie-chickens during spring (<400 ha; Robel et al. 1970).

I applied a dual frame sampling design in which samples were drawn from a list frame consisting of plots known to have been occupied by a lek during 2004, and a much larger area frame consisting of the statistical population of plots to which the estimate of occupancy can be inferred (Haines and Pollock 1998). The area frame completely overlapped the list frame, so inferences were based upon the mutually exclusive overlap and nonoverlap domains. Dual frame sampling was appropriate for this study because an area frame was necessary for sample plots to be representative of other plots in the population, and the list frame was useful for focusing adequate sampling effort in plots where leks were known to have occurred recently. The locations of leks, especially those attended by more than a few males, are relatively consistent among years (Schroeder and Braun 1992), which makes them amenable to the use of a list frame.

An observer visited each sample plot once during each of T=3 consecutive biweekly periods from 4 April 2005 until 15 May 2005 (Svedarsky 1983). A visit consisted of a 20-minute interval between 0.5 hours before and 2 hours after sunrise (Cartwright 2000) during which a plot was surveyed with the purpose of detecting the presence of a lek (i.e., ≥2 male prairie-chickens) by sight or sound. The value of some time-dependent covariates of p were recorded during each visit, whereas the value of other covariates that vary only spatially were recorded only once for each plot. Observers also compared maps of land cover from the GAP level 4 database with actual land cover in sample plots and marked corrections on the maps. Most of the covariates of ψ were measured using a GIS, but some were verified by observers in the field.

Occupancy models often require an assumption that p is homogeneous (i.e., does not vary among plots). Using covariates of p in the model may ameliorate the negative effects of potential heterogeneity in p, but to prevent the sampling design from introducing heterogeneity, each observer visited a different set of plots during each biweekly survey period. Differences among observers in their ability to detect leks, therefore, would not be correlated with specific plots.

Data Analysis

I transformed the value of the covariates of ψ and p so they were within the interval [-9.9, 9.9], which precluded problems with numerical optimization that occur occasionally when using a logit link function. I developed sets of 8 and 14 *a priori* models to represent hypotheses about which covariates contributed to variation in p and ψ , respectively. Included in the set of models for ψ were 2 supported by previous studies (Table 1; Merrill et al. 1999, Niemuth 2003). I used Program MARK to fit occupancy models to the detection-nondetection survey data (MacKenzie et al. 2002). I used Akaike's Information Criterion adjusted for sample size (AIC_c) to calculate the Akaike weight (w), which is a relative weight of evidence for a model, given the data. I based all inferences on parameter estimates averaged over the best models that accounted for ≥95% of the Akaike weights (Burnham and Anderson 2002:150, 162). To estimate uncertainty in $\hat{p}$ and $\hat{\psi}$ given specific values of covariates I calculated limits of 95% confidence intervals on the logit scale then transformed them to the real scale (Neter et al. 1996:603). I combined estimates of $\hat{\psi}$ across sampling domains to estimate the number of plots occupied by prairie-chicken leks in the northwest range of Minnesota (Haines and Pollock 1998). Finally, I conducted an exploratory analysis by fitting models that were not specified *a priori*.

RESULTS AND DISCUSSION

I randomly selected n_{Area}=135 plots from the area frame (N_{Area}=3,137 plots), but 2 were excluded because they were not accessible by passable public roads and were not visited by observers (Figure 1). Inferences, therefore, were limited to portions of the study area that were accessible by public roads during spring. I randomly selected n_{List}=135 plots from the list frame (N_{List}=181 plots), 1 of which was excluded due to inaccessibility. Six of the plots selected from the area frame were also on the list frame, so $n_{nonoverlap}$=127 plots were in the nonoverlap domain (i.e., 127=135–2–6), and $n_{overlap}$=140 plots were in the overlap domain (i.e., 140=135–1+6).

The AIC-best *a priori* model for p was the "global" model, which contained all 16 covariates (i.e., 5 for observers, recapture, day of the study, time of day, temperature, wind speed, presence of precipitation, proportion of the sky obscured by clouds, road density, density of interior roads, proportion of suitable land cover types that were visible from roads, and proportion of suitable land cover types that were under snow or temporary water). It accounted for 97% of the AIC weight in the model set. The second-best model for p, labeled the "weather-1" model, had an AIC weight of 3% and contained 5 covariates (i.e., time of day, temperature, wind speed, precipitation, and cloud cover).

The 4 best occupancy models, which accounted for 98% of the AIC weight, included the global model for p (Table 2). Although they contained 21–25 parameters, only 6 model-averaged parameter estimates had confidence intervals that did not include 0 (Table 3). Wind speed, cloud cover, road density, and an observer effect were correlated with p (Figure 2; $\hat{\bar{p}}$ = 0.45, 95% CI=0.34–0.56). Road density was also correlated with occupancy (Figure 3). No land cover covariates, however, were correlated with occupancy within each sampling frame.

The probability of occupancy was 0.83 (95% CI=0.31–0.98) for plots in the overlap domain (i.e., from the list frame) and 0.09 (95% CI=0.01–0.46) for plots in the nonoverlap domain (i.e., from the area frame but not the list frame). Therefore, $\hat{\psi}$ = 420 (SD=270) plots in the study area were occupied by a lek. The lack of precision of $\hat{\psi}$ was acceptable, given the objectives of the study. The results, however, will be useful for evaluating the level of sampling effort necessary to estimate $\hat{\psi}$ with adequate precision at range-wide scales in the future.

I started the exploratory analysis by simplifying the model for p to include only the dominant 4 covariates rather than all 16 and by using combinations of covariates for ψ that may not have been included in the *a priori* set of models. The AIC-best occupancy model then included domain, habitat area, density of all roads, and density of paved roads as covariates for ψ. There was still much model-selection uncertainty, and the combined-1 and disturbance-1 models for ψ were only 2.0 and 3.1 AIC-units away from the best model.

I further refined the exploratory analysis by removing the domain covariate because it appeared to be an excellent discriminator between occupied and unoccupied plots and therefore potentially masking relationships between ψ and more informative landscape characteristics. Using a reduced model for p (K=5) and no domain covariate for ψ resulted in 3 models that accounted for >99% of the AIC-weight in the new model set. The model-averaged parameter estimates whose confidence intervals did not include 0 were those for the proportion of the plot covered in grass, distance to the next nearest lek observed the previous year, area of habitat in the plot, and density of roads (Figure 4).

ACKNOWLEDGMENTS

I thank N. G. Emery, T. E. Flatt, N. W. Gruber, R. J. Kleinschmidt, E. D. Peterson, and J. B. Winter for collecting data in the field; R. Wright and B. Sampson for GIS assistance; J. Fieberg and D. Pierce for helping me improve the project proposal; and L. Gilbert for map interpretation and data entry.

LITERATURE CITED

Burnham, K. P., and D. R. Anderson. 2002. Model selection and multimodel inference: a practical information-theoretic approach. Second edition. Springer, New York.

Cartwright, K. S. 2000. Influence of landscape, land use, and lek attendance on greater prairie-chicken lek surveys. Thesis, Kansas State University, Manhattan, Kansas, USA.

Giudice, J. H. 2004. Minnesota prairie-chicken survey: 2004 annual report. Minnesota Department of Natural Resources, Madelia, Minnesota, USA.

Haines, D. E., and K. H. Pollock. 1998. Estimating the number of active and successful bald eagle nests: an application of the dual frame method. Environmental and Ecological Statistics 5:245-256.

Keating, K. A., and S. Cherry. 2004. Use and interpretation of logistic regression in habitat-selection studies. Journal of Wildlife Management 68:774-789.

MacKenzie, D. I., J. D. Nichols, G. B. Lachman, S. Droege, J. A. Royle, and C. A. Langtimm. 2002. Estimating site occupancy rates when detection probabilities are less than one. Ecology 83:2248-2255.

Merrill, M. D., K. A. Chapman, K. A. Poiani, and B. Winter. 1999. Land-use patterns surrounding greater prairie-chicken leks in northwestern Minnesota. Journal of Wildlife Management 63:189-198.

Neter, J., M. H. Kutner, C. J. Nachtsheim, and W. Wasserman. 1996. Applied linear statistical models. Fourth edition. Irwin, Chicago, Illinois.

Niemuth, N. D. 2000. Land use and vegetation associated with greater prairie-chicken leks in an agricultural landscape. Journal of Wildlife Management 64:278-286.

Niemuth, N. D. 2003. Identifying landscapes for greater prairie chicken translocation using habitat models and GIS: a case study. Wildlife Society Bulletin 31:145-155.

Robel, R. J., J. N. Briggs, J. J. Cebula, N. J. Silva, C. E. Viers, and P. G. Watt. 1970. Greater prairie chicken ranges, movements, and habitat usage in Kansas. Journal of Wildlife Management 34:286-306.

Schroeder, M. A., and C. E. Braun. 1992. Greater prairie-chicken attendance at leks and stability of leks in Colorado. Wilson Bulletin 104:273-284.

Svedarsky, W. D. 1983. Reproductive chronology of greater prairie chickens in Minnesota and recommendations for censusing and nest searching. Prairie Naturalist 15:120-124.

Table 1. *A priori* models for explaining variation in the probability (ψ) of a sample plot being occupied by a prairie-chicken lek in Minnesota during spring of 2005.

Name	Covariates included
Habitat-1	Grass[a], Prairie[a], Sedge[a], Forest[a], Crop[a], Edge[b], Tree[c], Lek distance[d]
Habitat-2	Grass, Prairie, Forest, Edge, Lek distance
Habitat-3	Grass, Forest, Lek distance
Habitat-4	Grass
Disturbance-1	Homes[e], Road density, Density of interior roads, Density of paved roads
Disturbance-2	Homes, Road density
Combined-1	Grass, Forest, Lek distance, Habitat area, Homes, Road density
Combined-2	Grass, Forest, Lek distance, Homes, Road density
Combined-3	Grass, Forest, Lek distance, Habitat area
Lek distance	Lek distance
Forest	Forest
Habitat area	Habitat area
Niemuth	Grass, Sedge, Forest, Lek distance
Merrill	Forest, Homes

[a] Proportion of area of a plot in this cover type.
[b] Edge between forest and nonforest cover types.
[c] Presence of trees within suitable cover types.
[d] Distance from the nearest known lek during the 2004.
[e] Number of occupied human residences within the plot.

Table 2. Ranking of *a priori* models of occupancy of PLS sections by leks of greater prairie-chickens in northwest Minnesota during spring of 2005 (models with AIC-weight <0.001 not included).

Model[a]	K[b]	AIC_c	AIC-weight
p(global) ψ (disturbance-1)	22	608.9	0.677
p(global) ψ (combined-1)	25	612.0	0.143
p(global) ψ (disturbance-2)	21	612.6	0.107
p(global) ψ (combined-2)	24	613.9	0.056
p(weather-1) ψ (combined-1)	14	619.1	0.004
p(global) ψ (combined-3)	23	619.2	0.004
p(global) ψ (habitat-2)	24	619.7	0.003
p(global) ψ (lek distance)	20	620.4	0.002
p(weather-1) ψ (disturbance-1)	11	621.9	0.001
p(global) ψ (habitat-1)	27	622.5	0.001
p(global) ψ (habitat-4)	20	622.7	0.001
p(global) ψ (habitat-3)	22	622.8	0.001
p(global) ψ (domain)	19	622.9	0.001

[a] Models for p, the probability of detection, are described in the text; models for ψ, the probability of occupancy, are explained in Table 1.

[b] K = number of parameters, which includes 2 intercept terms—1 for the p portion of the model and 1 for the ψ portion.

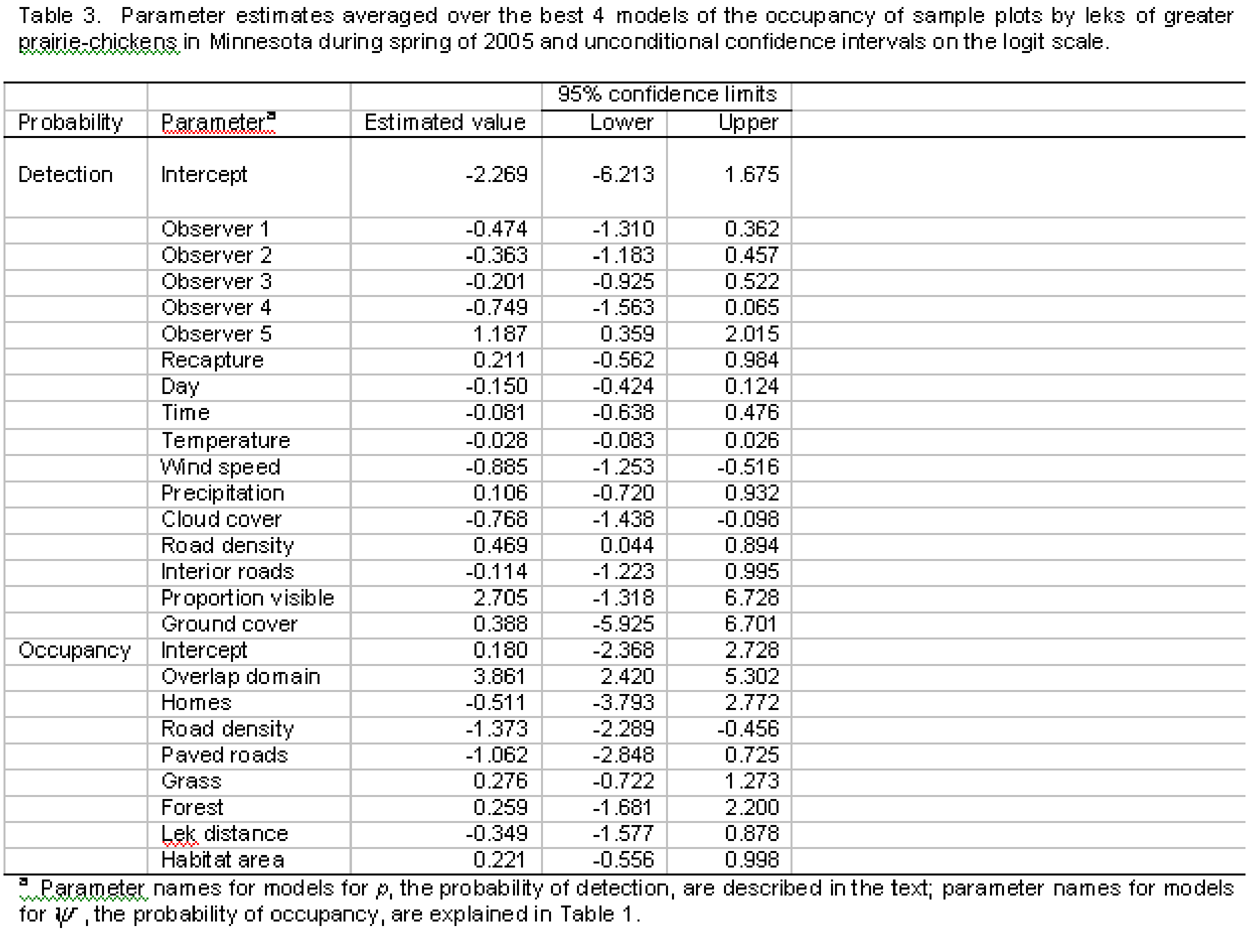

Table 3. Parameter estimates averaged over the best 4 models of the occupancy of sample plots by leks of greater prairie-chickens in Minnesota during spring of 2005 and unconditional confidence intervals on the logit scale.

			95% confidence limits	
Probability	Parameter[a]	Estimated value	Lower	Upper
Detection	Intercept	-2.269	-6.213	1.675
	Observer 1	-0.474	-1.310	0.362
	Observer 2	-0.363	-1.183	0.457
	Observer 3	-0.201	-0.925	0.522
	Observer 4	-0.749	-1.563	0.065
	Observer 5	1.187	0.359	2.015
	Recapture	0.211	-0.562	0.984
	Day	-0.150	-0.424	0.124
	Time	-0.081	-0.638	0.476
	Temperature	-0.028	-0.083	0.026
	Wind speed	-0.885	-1.253	-0.516
	Precipitation	0.106	-0.720	0.932
	Cloud cover	-0.768	-1.438	-0.098
	Road density	0.469	0.044	0.894
	Interior roads	-0.114	-1.223	0.995
	Proportion visible	2.705	-1.318	6.728
	Ground cover	0.388	-5.925	6.701
Occupancy	Intercept	0.180	-2.368	2.728
	Overlap domain	3.861	2.420	5.302
	Homes	-0.511	-3.793	2.772
	Road density	-1.373	-2.289	-0.456
	Paved roads	-1.062	-2.848	0.725
	Grass	0.276	-0.722	1.273
	Forest	0.259	-1.681	2.200
	Lek distance	-0.349	-1.577	0.878
	Habitat area	0.221	-0.556	0.998

[a] Parameter names for models for p, the probability of detection, are described in the text; parameter names for models for ψ, the probability of occupancy, are explained in Table 1.

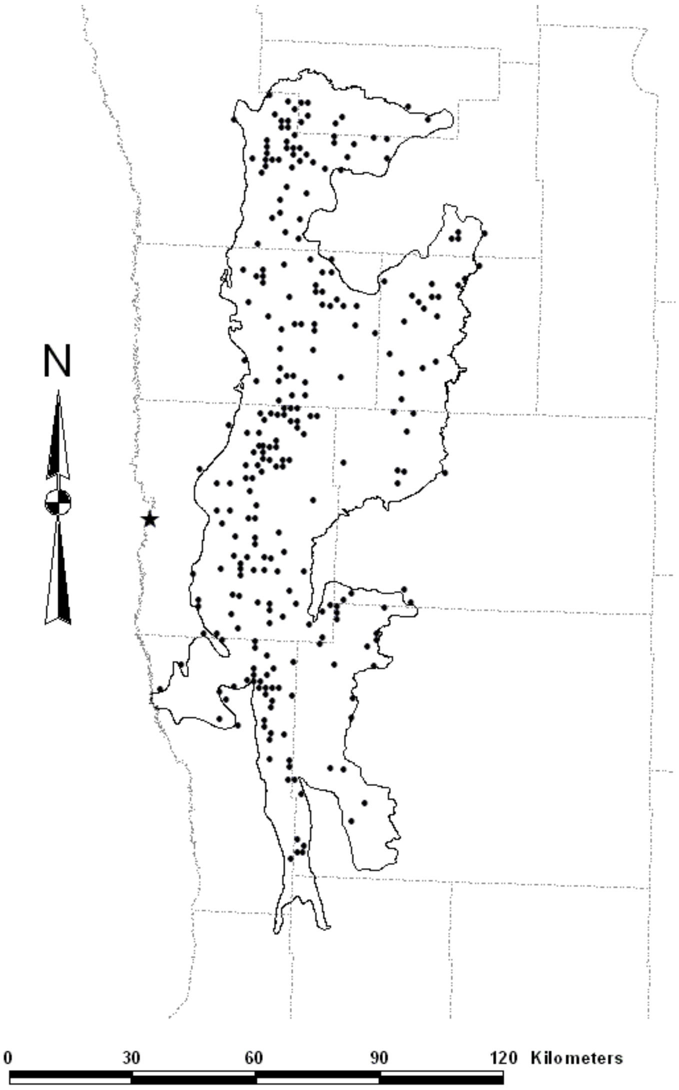

Figure 1. The northwest prairie-chicken range based on land type associations of the Ecological Classification System (solid line) relative to county boundaries (dashed lines) in western Minnesota. Sample plots (dots) were not selected from areas >90 km southeast of Moorhead (star).

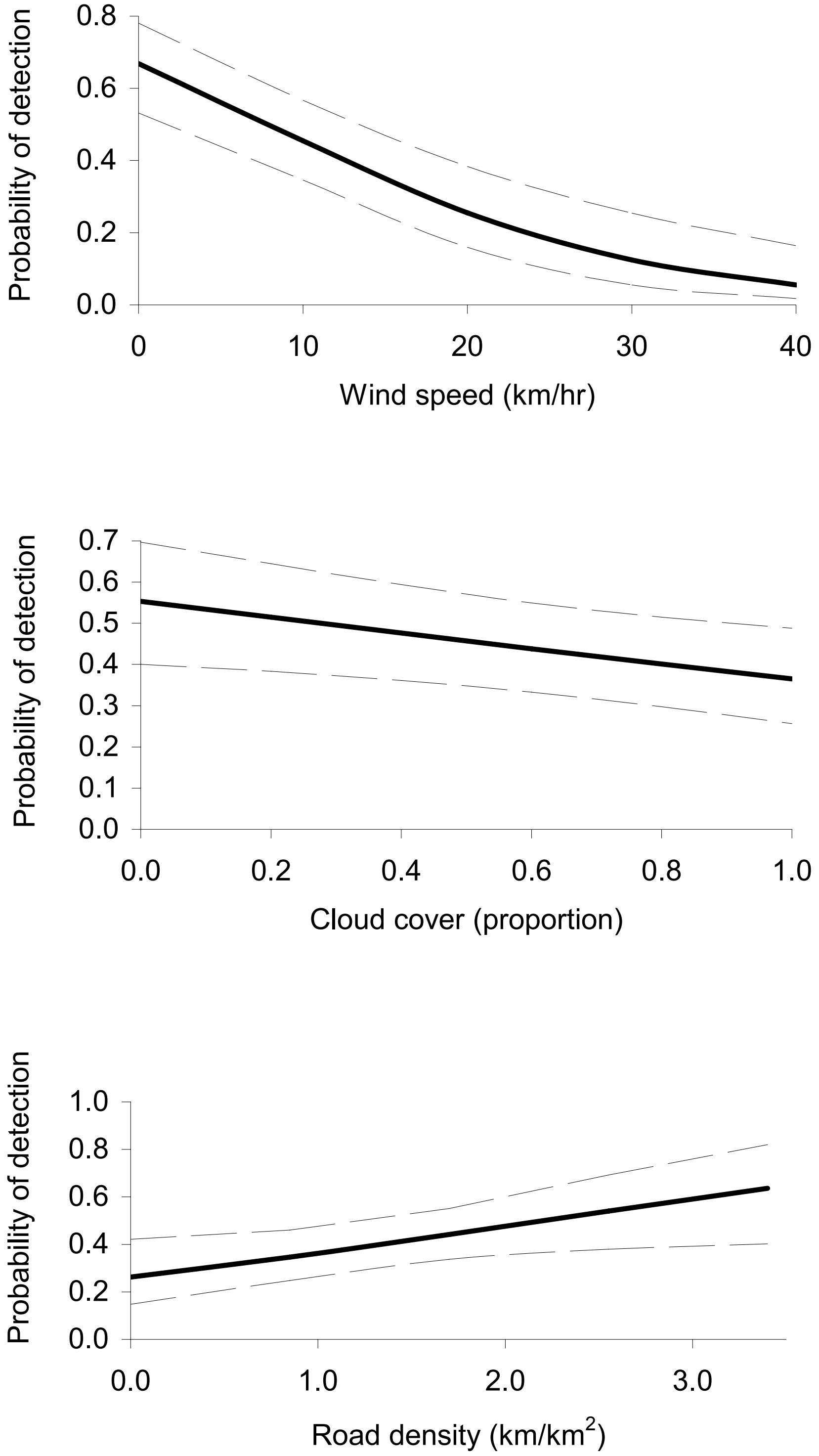

Figure 2. Model-averaged probabilities (and 95% confidence intervals) of detecting a prairie-chicken lek in sample plots in Minnesota during spring of 2005 over the range of observed values of 3 selected model parameters based on *a priori* models.

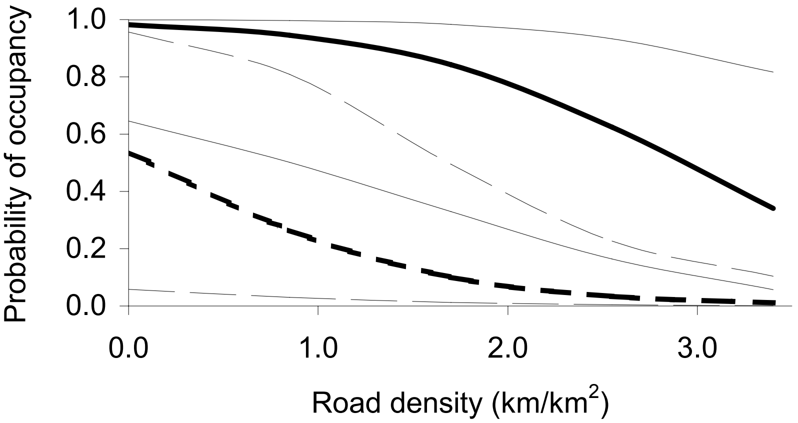

Figure 3. Model-averaged probabilities (heavy lines) and 95% confidence intervals (light lines) of a sample plot in Minnesota being occupied by a prairie-chicken lek during spring of 2005 over the observed range of road densities in the overlap domain (i.e., plots known to have contained a lek during 2004; solid lines) and nonoverlap domain (i.e., all other plots in the study area; dashed lines) based on *a priori* models.

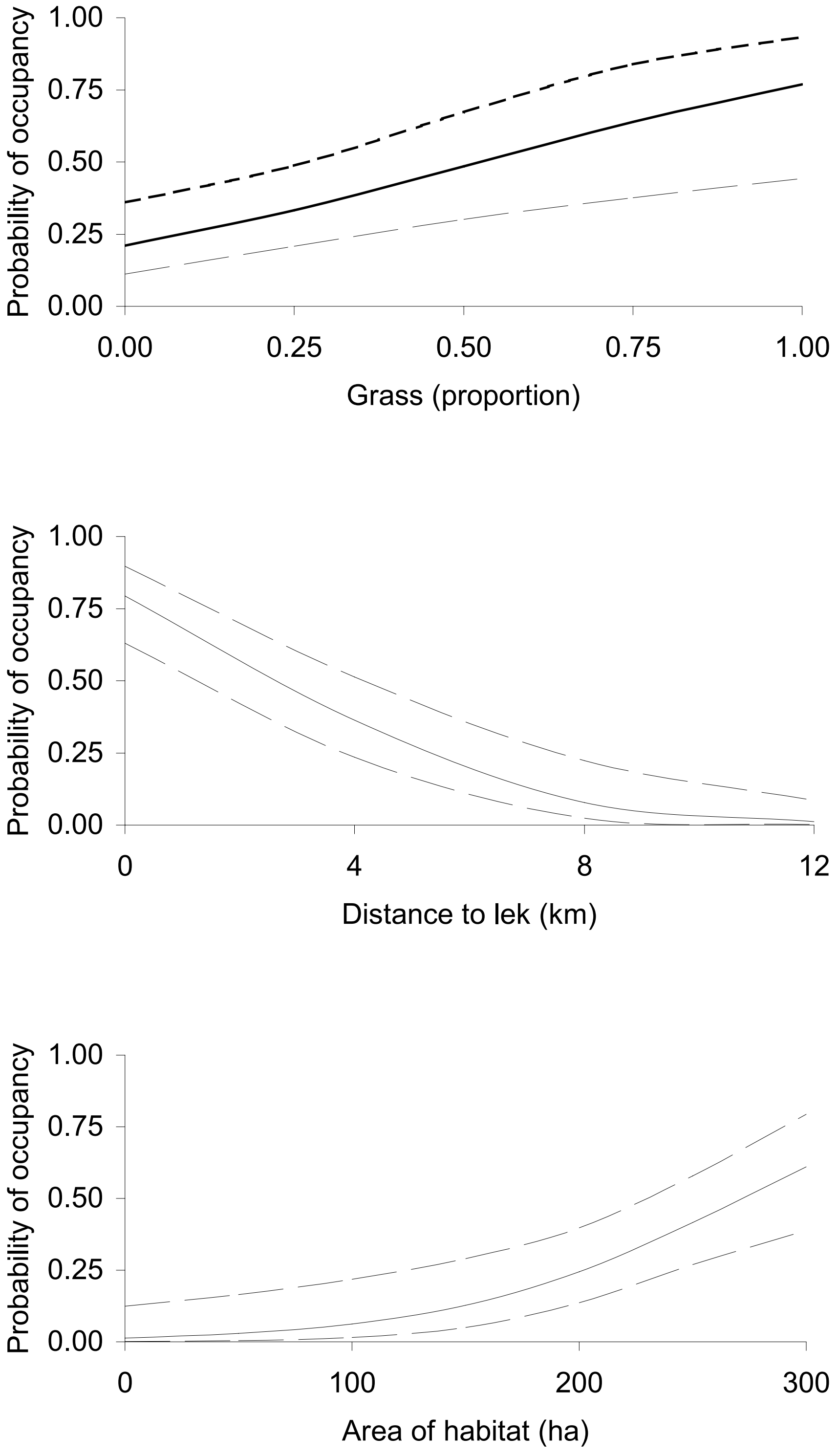

Figure 4. Model-averaged probabilities (and 95% confidence intervals) of detecting a prairie-chicken lek in sample plots in Minnesota during spring of 2005 over the range of observed values of 3 selected model parameters based on an exploratory analysis.

IDENTIFYING PLOTS FOR SURVEYS OF SHARP-TAILED GROUSE IN MINNESOTA

Michael A. Larson

SUMMARY OF FINDINGS

The justification, objectives, and methods for this project are identical to those for the prairie-chicken project, which is summarized separately. Collection of data for sharp-tailed grouse (Tympanuchus phasianellus campestris) occurred during spring 2006 in the eastern portion of their range in Minnesota, and is occurring during spring 2007 in the northwestern portion of their range in Minnesota (Figure 1). Data from both years will be analyzed together. Therefore, results for sharp-tailed grouse are not available at this time.

ACKNOWLEDGMENTS

I thank J. Bartz, P. Coy, C. Humpal, J. Feine, A. Norton, K. Noyce, P. Ojanen, M. Thompson, and G. Wright for collecting field data. Housing for field staff was provided by M. Anderson (Agassiz National Wildlife Refuge), G. Mehmel (Red Lake WMA), R. Prachar (Thief Lake WMA), R. Tuszynski (Mille Lacs WMA), and S. Wood (Roseau River WMA). I also thank R. Wright and B. Sampson for GIS support, R. Danielson for consultations on data management, L. Gilbert for data entry, and J. Fieberg and D. Pierce for reviewing drafts of the proposal for this project.

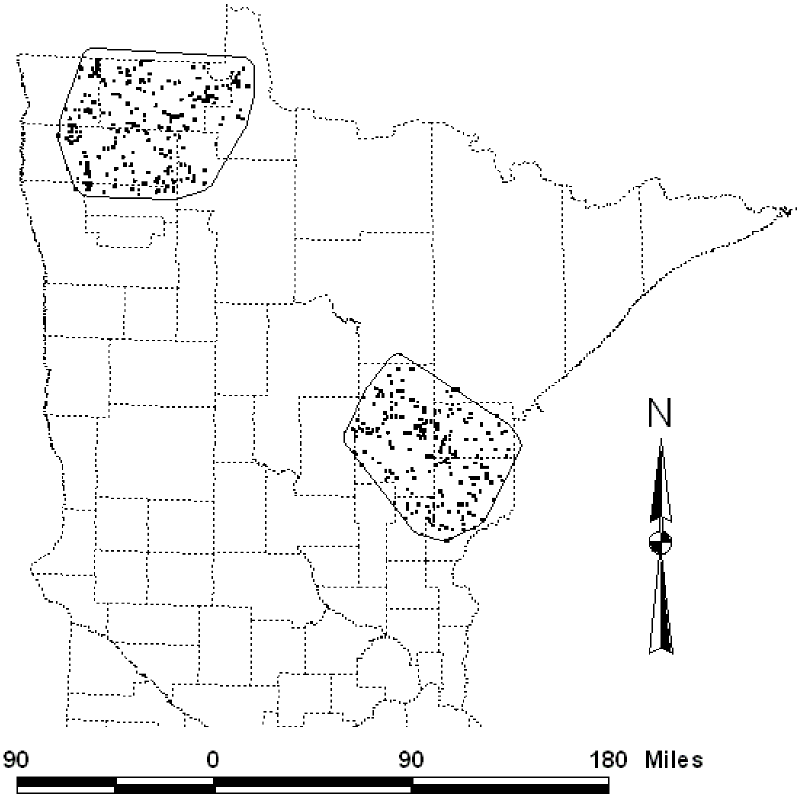

Figure 1. Sharp-tailed grouse study areas (solid lines) relative to county boundaries (dashed lines) in northern Minnesota. Sample plots (dots) were selected randomly after screening for road access and minimum habitat availability.

USE OF STABLE ISOTOPES OF CARBON, NITROGEN, AND OXYGEN IN STUDIES OF DIET AND NUTRITION OF MINNESOTA BLACK BEARS

Karen V. Noyce

SUMMARY OF FINDINGS

Hair from Minnesota black bears (*Ursus americanus*), as well as samples of common plant and animal bear foods, were analyzed for stable isotopes of carbon, nitrogen, and oxygen, to explore their potential utility in studies of bear diet and nutrition. Values for both $\delta^{13}C$ and $\delta^{15}N$ in plant foods were distinct from those in bear hair; values in animal foods (ants, fawns) were intermediate, overlapping somewhat with both bear hair and plant foods. Carbon isotope ratios in bear hair exhibited an increasing north-south gradient (moving from core to peripheral bear range) that likely reflected an increasing proportion of C4 plants (corn, cane sugar) in the diet. Bears known to have been feeding on corn (one denned in a corn field) had the highest $\delta^{13}C$ values. Hair collected from 2 yearling bears and cut into 4 equal lengths representing sequential 2-month periods during their cub year, showed abrupt increases in $\delta^{13}C$ values between mid- and late-summer, likely signaling a shift in diet to corn or other anthropogenic foods. There was no difference in $\delta^{15}N$ in bears from study areas with very different nutritional resources, but $\delta^{15}N$ was typically higher in cubs than in their mothers, consistent with their consumption of their mothers' milk. In contrast, $\delta^{18}O$ did not show the expected elevation in offspring relative to their mothers in 5 paired samples, nor did it provide clear indication of nursing/weaning history in cubs. However, $\delta^{18}O$ values in adults (n=6) all fell between 16.9 and 17.8, whereas values from whole hair or portions of hair collected from yearling bears spanned a much wider range (14.4–19.3). Further work to improve our understanding of $\delta^{18}O$ in bear tissues might yet reveal ways that it can help decipher infant nutritional patterns in bears.

INTRODUCTION

The use of stable isotopes in wildlife studies has expanded rapidly in recent years as new applications for these methods have been developed. Interpreting stable isotope ratios involves comparing the ratio of 2 isotopes of an element (e.g. ^{13}C and ^{12}C) in the tissues of an animal to their constant ratio in the inorganic environment in order to deduce information about the animal's feeding behavior (the comparison is expressed as $\delta^{13}C$). Although isotopes of the same element behave in the same manner in metabolic reactions in animal tissues, because of their different masses, heavy and light isotopes react at different rates and are thus assimilated or released into the environment at different rates. For example, plants assimilate ^{13}C and ^{12}C in different proportion to their abundance in the environment, and 2 major groups of plants (C3 and C4), using different photosynthetic pathways, assimilate ^{13}C and ^{12}C in characteristically different ratios from each other. Plants in the C3 group include temperate forest species, such as Minnesota's native forest vegetation, whereas C4 plants are mostly of tropical origin and include the common agricultural crops, corn and sugar cane. Carbon isotope ratios in animal tissues can indicate the relative proportion of C3 and C4 plants in the assimilated diet. This application of stable isotope analysis has recently been used to compare the feeding histories of back-country versus nuisance bears in Japan (Mizukami et al. 2005).

Because of differential uptake of nitrogen isotopes in animal tissues, the $\delta^{15}N$ signature changes by a relatively constant factor with each successive step in the trophic chain; i.e., $\delta^{15}N$ is higher in plants than in the environment, higher in herbivores than in plants, and higher in carnivores than in herbivores. This characteristic has been used to compare the relative importance of plant versus animal protein in the diets of omnivores, such as black bears and grizzly bears in British Columbia (Hobson et al. 2000) and grizzlies in different parts of North America (Mowat and Heard 2006).

Stable isotopes of carbon and nitrogen have also been used to investigate weaning and fasting in mammals (polar bears: Polischuk et al. 2001). Though not previously used in wildlife studies, anthropologists have found $\delta^{18}O$ in tooth enamel and dentine to be helpful in deducing weaning practices in ancient cultures (Wright et al. 1999). Oxygen isotopes show a small trophic effect similar to nitrogen, such that ^{18}O in body fluids is more enriched than in environmental (meteoric) water. Thus, if an animal obtains most of its water from body fluids, such as milk, $\delta^{18}O$ will be higher than if its water is imbibed from the environment. In combination with $\delta^{13}C$ and $\delta^{15}N$, this can be used to surmise the age at which supplemental foods enter the diet of infants and the age of weaning.

I undertook exploratory work to examine the feasibility of using hair samples from denning black bears to investigate several aspects of their diet and nutritional ecology. The composition of hair reflects an animal's body chemistry at the time it is growing, thus a shaft of hair sequentially sectioned, can provide a record through time of changes in isotopic composition. The hair of yearling bears in the den reflects their diet from time of birth the previous winter until the cessation of hair growth in late summer or early fall. Adult hair collected in the den represents a shorter period of time, probably about May–September of the previous year (Mizukami et al. 2005), as hair growth for the year's molt does not start until mid-spring. I collected bear hair from a variety of bears and locations, as well as samples of common plant and animal foods of bears. For this work, sample sizes were small, but the scope of samples broad, in keeping with the intent of the study as preparatory to planning future, more comprehensive investigations.

OBJECTIVES

My objectives were to investigate the feasibility of the following:

- determining the importance of anthropogenic foods in the diet of bears from different regions of Minnesota;
- correlating nutritional condition of individual bears with particular temporal dietary (isotopic) patterns;
- determining dietary composition, using mixing models, based on isotopic signatures of bear hair and major bear foods; and
- comparing weaning histories of cubs exhibiting widely different nutritional status distinguishing hair from cubs from that of older juveniles and adults.

METHODS

I sampled male bears residing at the southern fringe of the bear range and females from 3 study areas in other parts of Minnesota including: Camp Ripley, located in central Minnesota in the southern transition area between forest and agriculture; the Chippewa National Forest (CNF), located 150 km to the north in the center of the bear range; and Voyageurs National Park (VNP), located along the Canadian border, another 130 km to the north (Table 1). I collected 2 mother-daughter pairs of samples from females denned with their yearlings in each study area. Hair from 3 yearlings (2 from CNF, 1 from Camp Ripley) was subsampled by dividing it into 4 equal lengths, each portion representing a 2-month time period between February–September during the bear's previous year. Samples of bear foods were collected from either the CNF or Camp Ripley areas.

Samples of bear foods were air-dried on low heat in a convection drying oven, then ground to a fine powder using mortar and pestle. Vegetation samples comprised a mixture of equal parts clover (*Trifolium* spp), wild calla (*Calla palustris*), jewelweed (*Impatiens biflora*), and new leaves of quaking aspen (*Populus tremuloides*). Fruit samples included strawberries (*Fragaria* spp) and raspberries (*Rubus idaeus*), mixed in equal parts. Samples of bear hair were rinsed 3 times in a 2:1 chloroform-methanol solution to remove oils, and then air-dried, cut into small segments and mixed. All samples were sent to the Colorado Plateau Stable Isotope Laboratory at Northern Arizona University, where they were weighed and analyzed, using a Thermo-Finnigan Delta^{plus} Advantage gas isotope-ratio mass spectrometer, interfaced with a Costech Analytical ECS4010 elemental analyzer. All samples were analyzed for $\delta^{13}C$ and $\delta^{15}N$; mother-daughter pairs were analyzed for $\delta^{18}O$ as well.

RESULTS AND DISCUSSION

There was distinct separation in both $\delta^{13}C$ and $\delta^{15}N$ values between bear hair and plant foods, except for 1 of 2 mixed-berry samples, in which $\delta^{15}N$ was anomalously high (Figure 1). In other plant samples, $\delta^{13}C$ and $\delta^{15}N$ were lower than and non-overlapping with values in bear hair. The 2 samples of mixed green vegetation yielded the lowest values of $\delta^{13}C$ (mean of -28.8 $\delta^{13}C$, versus mean –26.6 $\delta^{13}C$ for berries and nuts). All plant samples were below –25.9. Among plant foods, $\delta^{15}N$ was lowest in 2 hazelnut samples. Overall, $\delta^{15}N$ ranged from –2.67 to 0.88 in plant material, except in 1 berry sample where $\delta^{15}N$ was 3.14, in sharp contrast to the other berry sample.

Bear hair samples spanned a broader range of values; however, in all cases, $\delta^{13}C$ was >–25 and $\delta^{15}N$ was >3. Animal foods of bears were intermediate: ants were lower in both $\delta^{13}C$ and $\delta^{15}N$ than bear hair and higher in $\delta^{13}C$ and $\delta^{15}N$ than most plant foods. Samples from 2 white-tailed deer (*Odocoileus virginianus*) fawns of different ages were very different in $\delta^{15}N$; values in hair and muscle from a 20-day-old fawn (Figure 1, points 1, 2) were considerably higher than in a 12-day-old fawn (Fig 1, points 3,4). For $\delta^{13}C$, hair samples from the 2 fawns were similar, but muscle samples differed. In both fawns, $\delta^{13}C$ was higher in muscle than in hair.

Bear hair showed distinct differences by location (Figure 2); $\delta^{13}C$ in bears living in or south of Camp Ripley was higher than in any bears from the CNF or VNP. The highest values were in 2 adult males living in the southern fringe of the bear range. One was known to have fed on corn extensively before denning (in the cornfield). A third male living at the edge of the bear range, along with Camp Ripley bears, yielded somewhat lower $\delta^{13}C$ values, but all were above –21; one Camp Ripley female was also known to have fed on corn in the late summer. Bears from the CNP all had $\delta^{13}C$ below –21 and VNP bears ≤-23.

Results for mother-daughter pairs were varied (Table 2). There were no consistent relationships between adults and their nursing offspring in any stable isotope ratios, though whole-hair $\delta^{15}N$ tended to be higher in offspring than in their mothers (4 of 5 cases). There were no consistent temporal trends in subsampled yearling hair that appeared indicative of time of weaning. Moreover, we did not see the expected increase in $\delta^{18}O$ in hair of daughters relative to that of mothers. In fact, particularly in one case (Table 2), $\delta^{18}O$ in the daughter (bear #21) was lower than that of her mother (bear #13), even early in the season, when she should have been nursing.

Nevertheless, in 2 of 3 cases where we subsampled yearling hair (1 Camp Ripley and 1 CNF yearling), $\delta^{13}C$ values showed a similar marked and sudden change between the third and fourth time periods, representing approximately June-July and August-September (Table 2, Figure 3). The first 3 sampling periods (the distal portions of the hairs) were similar in $\delta^{13}C$ (-22 to -23), but the proximal section (the last to grow before denning) jumped to –20.16 in the CNF cub and –16.81 in the Camp Ripley cub. A second CNF cub did not show this pattern; this yearling was extremely undernourished, and it may be significant that its $\delta^{18}O$ values throughout the season were lower than those of the other CNF yearling, who was of average body condition and further below those of the Camp Ripley yearling, who was the largest and fattest of the three.

DISCUSSION AND FUTURE DIRECTION

These preliminary results seem to indicate that δ^{13}C may be a sensitive and useful index to regional and temporal differences in the reliance of Minnesota bears (and perhaps other species) on some types of anthropogenic foods. In this case, they appear to indicate that corn and/or other human-related foods (e.g. corn-based dog food, processed foods containing cane or corn sugar) increasingly enter the food chain of bears from relatively unpopulated parts of the northern bear range (VNP) to the populated and agricultural south. Though consistent with what we know of bear feeding behavior across the state, the consistency of the trend in all the samples was somewhat surprising — every one of the Camp Ripley and more southerly bears appeared to include significantly more such foods in their diet than any of the CNF and VNP bears. It would be prudent to rule out, through further sampling, any other potential sources of this consistent geographic gradation in δ^{13}C.

Neither δ^{18}O nor δ^{15}N provided an easy or clear-cut way to infer nutritional condition of bears, interpret the weaning history of cubs, or to distinguish between cubs and adults based on hair samples. There were, nevertheless, some intriguing suggestions in this small exploratory data that may be revealed more fully with further investigation and larger sample sizes from bears with known histories. For example, if δ^{18}O is consistently more labile in nursing cubs than in adults, this could perhaps provide a first screen for distinguishing between cubs and adults based on hair samples. The counterintuitive occurrence of higher δ^{18}O values in mothers than in their nursing cubs raises questions about whether the metabolism of body fat, which produces water, in post-denning mothers may influence values in females of different body condition. Though current data are insufficient to attempt diet reconstruction using mixing models due to the variation in stable isotope signatures of similar types of food, more comprehensive sampling and analysis of food items should make this possible. Finally, further sampling of mother-offspring pairs and studies using captive bears can help to better document hair growth patterns, enabling clearer interpretations of isotope data.

ACKNOWLEDGMENTS

Many people helped with the handling of bears and collection of hair samples, especially D. Garshelis and P. Coy, of the Minnesota Department of Natural Resources Forest Wildlife Populations and Research Group. I thank C. Humpal for her help preparing samples for analysis and R. Doucett, of the Colorado Plateau Stable Isotope Laboratory, for advice and performing analyses.

LITERATURE CITED

HOBSON, K. A., B. N. MCLELLAN, AND J. G. WOODS. 2000. Using stable carbon (δ^{13}C) and nitrogen (δ^{15}N) isotopes to infer trophic relationships among black and grizzly bears in the upper Columbia River basin, British Columbia. 2000. Canadian Journal of Zoology 78: 1332 – 1339.

MIZUKAMI, R. N, M. GOTO, S. IZUMIYAMA, H. HAYASHI, AND M. YOH. 2005. Estimation of feeding history by measuring carbon and nitrogen stable isotope ratios in hair of Asiatic black bears. Ursus 16:93 – 101.

MOWAT, G., AND D. HEARD. 2006. Major components of grizzly bear diet across North America. Canadian Journal of Zoology 84:473 – 489.

POLISCHUK, S. C., K. A. HOBSON, AND M. A. RAMSAY. 2001. Use of stable-carbon and –nitrogen isotopes to assess weaning and fasting in female polar bears and their cubs. Canadian Journal of Zoology 79:499 – 511.

WRIGHT, L. E., AND H. P. SCHWARCZ. 1999. Correspondence between stable carbon, oxygen and nitrogen isotopes in human tooth enamel and dentine: infant diets at Kaminaljoyu. Journal of Archaeological Science 26: 1159 – 1170.

Table 1. Description of samples collected in Minnesota for isotopic analysis.

Sample ID	Sample material	Location collected	Analyses performed
1a,b	Green vegetation mix	CNF	C,N
2a,b	Acorns	CNF, Garrison	C,N
3a,b	Hazelnuts	CNF	C,N
4a1,4b1	Deer fawns (hair)	12 and 20 days old	C,N,O,
4a2,4b2	Deer fawn (meat)	“ “ “	C,N,O
5a,b	Mix of berries	Misc.	C,N
6a,b	Ants	Camp Ripley	C,N
7	Bear - ad. M (denned in cornfield)		C,N
8	Bear – ad. M	Long Prairie	C,N
9	Bear – ad. M	Buckman	C,N
10	Bear – ad. F (fed in corn)	Camp Ripley	C,N,O
11	Bear – ad. F	Camp Ripley	C,N,O
12	Bear – ad. F	CNF	C,N,O
13	Bear – ad. F	CNF	C,N,O
14	Bear – ad. F	VNP	C,N,O
15	Bear – ad. F	VNP	C,N,O
16	Bear – ad. F	CNF	C,N
17	Bear – ad. F	CNF	C,N
18	Bear – yrl. F, daughter of #10	Camp Ripley	C,N,O
19a,b,c,d	Bear – yrl. F, daughter of #11	Camp Ripley	4 subsamples C,N,O
20a,b,c,d	Bear – yrl. F, daughter of #12	CNF	“ “ “ ” “
21a,b,c,d	Bear – yrl. F, daughter of #13	CNF	“ “ “ “ “
22	Bear – yrl. F, daughter of #14	VNP	C,N,O
23	Bear – yrl. F, daughter of #15	VNP	C,N,O

Table 2. Ratios of stable carbon, nitrogen, and oxygen in mother-daughter pairs of black bears in Minnesota.

	$\delta^{13}C$		$\delta^{15}N$		$\delta^{18}O$	
	Mother	Daughter	Mother	Daughter	Mother	Daughter
Bears 11,19						
Feb – March		-21.96		4.90		18
April – May		-22.18		4.73		19.3
June – July		-22.5		3.70		19.1
Aug – Sept		-16.81		3.03		16.8
Whole hair	-20.34	-20.86	3.44	4.09	17.8	18.3
Bears 12,20						
Feb – March		-22.95		5.08		16
April – May		-22.14		5.25		16.8
June – July		-22.02		4.95		16.6
Aug – Sept		-20.16		5.57		17.3
Whole hair	-21.75	-22.32	4.38	5.21	17.3	16.7
Bears 13,21						
Feb – March		-22.67		4.89		14.8
April – May		-23.1		3.02		14.4
June – July		-23.63		3.45		15.4
Aug – Sept		-24.37		5.15		
Whole hair	-22.84	-23.44	4.34	4.12	17.1	
Bears 14,22						
Whole hair	-22.90	-24.44	6.86	7.67	17.2	15.4
Bears 15,23						
Whole hair	-22.67	-22.95	5.11	5.81	16.9	17.3

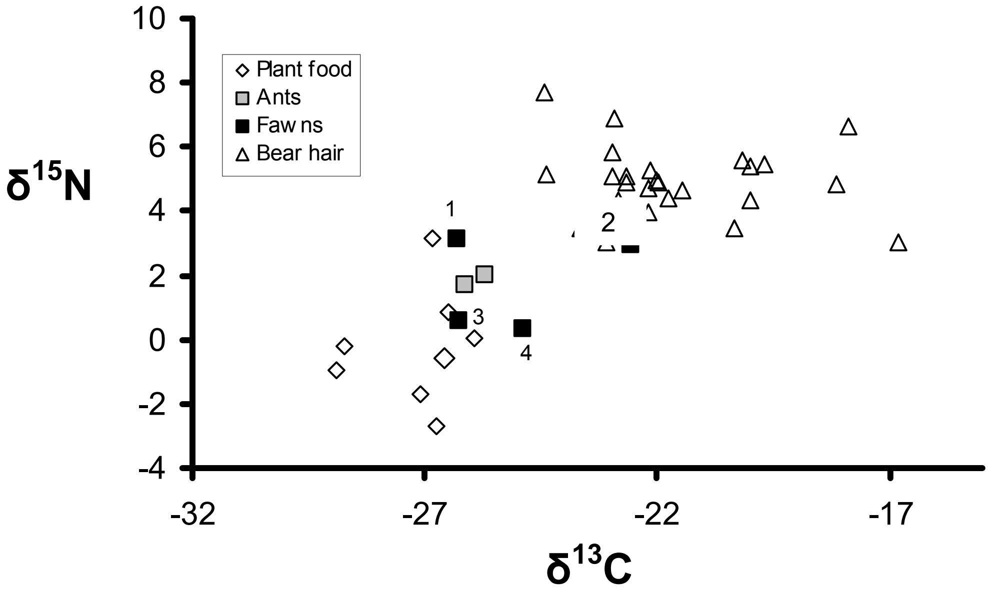

Figure 1. Stable isotope signatures of δ^{13}C and δ^{15}N in samples of common wild bear foods and all hair samples from Minnesota black bears. (Numbered samples correspond to: 1,2 – hair and muscle, respectively, from 20-day-old white-tailed deer fawn; 3,4 – hair and muscle from 12-day-old fawn).

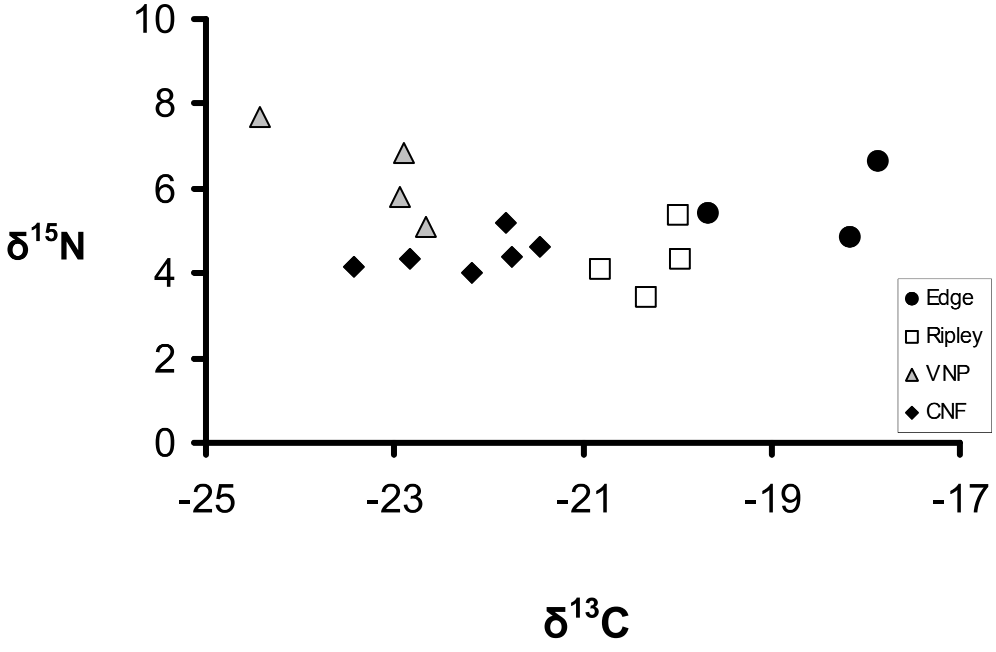

Figure 2. Stable isotope signatures of δ^{13}C and δ^{15}N in hairs collected from denning bears that resided in different parts of Minnesota's bear range. (In cases where hairs were subsampled to obtain chronologic results, values on this graph represent the mean of the subsamples.)

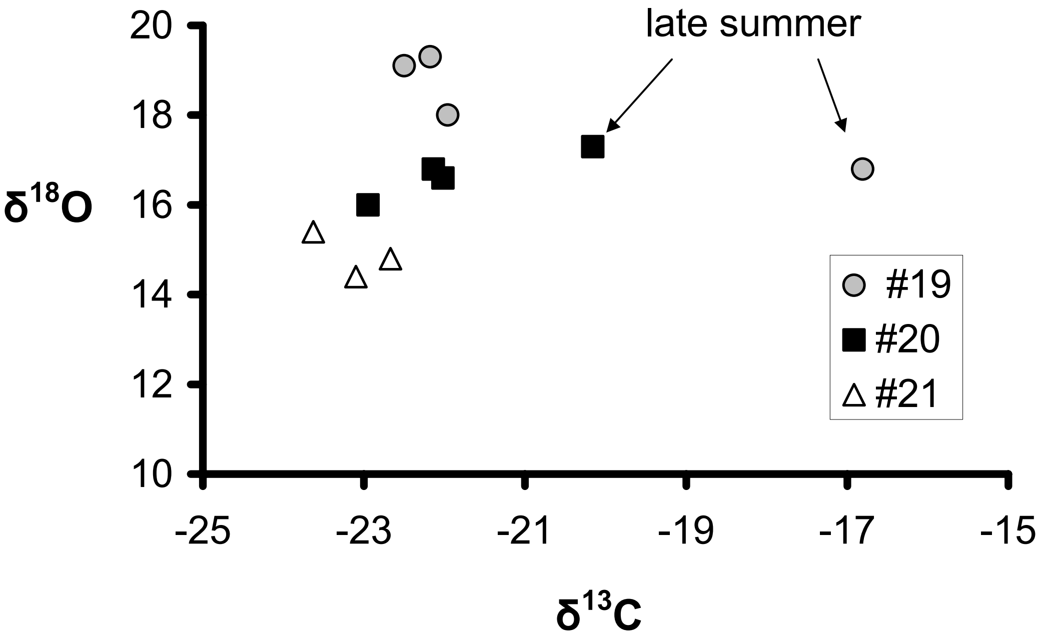

Figure 3. Stable isotope signatures of δ^{13}C and δ^{18}O in hairs collected from yearling bears. Hairs were divided into 4 equal portions, representing different 2-month periods in the bear's life from approximately 0 – 8 months old. Due to thinning of the hair shaft at the proximal end, there was insufficient sample to analyze for late summer stable isotope signatures for bear #21.