

## FISHER AND MARTEN DEMOGRAPHY AND HABITAT USE IN MINNESOTA

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### SUMMARY OF FINDINGS

Following evaluation of field methods in 2007-08, we began full-scale fieldwork during winter 2008-09 on a study of fisher (*Martes pennanti*) and marten (*Martes americana*) ecology in northern Minnesota. Including the pilot year of the project, a total of 47 martens and 30 fishers have been radiocollared. Of the 47 marten radiocollared (25M, 22F), 7 individuals (2M, 5F) were able to subsequently slip their collars. In addition, 1 animal has not been relocated since shortly after capture. Of the remaining 39 animals, 26 are currently alive, 6 have died from predation (3 raptor kills, 3 mammalian carnivores), 3 were harvested during the legal trapping season, 2 died from capture or collar-related complications, and 2 are missing. Of the 30 fisher radiocollared (11M, 19F), 9 have shed their collars (2M, 6F), 8 due to insufficient collar design. Of the remaining 21 fisher, 13 are alive, 1 has not been relocated since release, 5 have died from predation (2 raptor kills, 3 mammalian carnivores), 1 additional collar (but no animal remains) was found under an active eagle nest, and 1 animal was trapped outside of the legal harvest season. Age information is not yet available, but 8 of the 13 female fishers monitored this spring produced litters, with an average litter size of 2.9. All natal dens were in tree cavities, primarily in large-diameter aspen trees or snags. Twelve kits from 4 females have subsequently died after the nursing females were killed by other predators. Most fisher kits appear to have been born during the first 10 days of April. Monitoring of marten reproductive success is ongoing, and it appears marten may give birth later than fisher. As of this writing, presence of kits has only been confirmed for 2 of 12 female marten (1 litter in a burrow within a rock/dirt berm, the other in a cedar tree cavity). We suspect other females have kits that we have yet to locate. Opportunistic sampling of rest sites used by marten suggests that during winter, most rest sites are underground, with increasing use of on- or above-ground locations in spring and summer. The majority of rest sites used by fisher, regardless of season, have been in tree cavities, with some use of on- or below-ground sites. Preliminary analysis of temperature sensor data from 4 fishers suggests that during winter they may spend over 75% of their time in den/rest sites. Data from 1 female fisher suggests that females with kits may spend only 25% of their time at the natal den during the first 10 days after kits are born. No temperature sensor data from radiocollared marten has yet been recovered. Visual examination of radiolocation data indicates that marten home ranges may be from 1 – 3 mi<sup>2</sup>, while fisher home ranges may vary between 3 – 5 mi<sup>2</sup>. The longest confirmed dispersal distance for one of our radiocollared fisher has been approximately 18 miles, while the longest confirmed dispersal distance for one of our radiocollared marten has been approximately 8 miles.

### INTRODUCTION

American marten and fisher are native to Minnesota, but reliable documentation of their historic distribution is limited. Undoubtedly, northeastern Minnesota was a stronghold for the marten population, though notable numbers likely occurred in the northern border counties as far west as Roseau county. Limited information suggests they occurred as far south and west as Crow Wing and Polk counties. As a result of over-harvest, marten were considered rare in Minnesota by 1900, and extensive logging and burning around the turn of the century further contributed to the near extirpation of marten from Minnesota by the 1930s (Swanson et al. 1945). Fishers in Minnesota appear to have historically occupied a larger geographic area than martens, extending further south and west into the hardwood dominated transition zone, and also into southeast Minnesota (Swanson et al. 1945, Balser and Longley 1966). The impacts of

over-harvest and habitat alteration were equally as detrimental to fisher, with populations substantially reduced by the 1930s.

Legally, fisher and marten were unprotected in Minnesota prior to 1917, after which harvest season length restrictions were implemented. These protections were removed in the mid-1920s, and remained so until all harvest was prohibited in 1929. Seasons remained closed until 1977 for fisher and 1985 for marten, when limited harvests were reinstated. Since then, trapping zones and quotas have periodically increased to the current combined quota of 5 fisher/marten per trapper. While harvest is legal in approximately the northern 50% of the state, most marten harvest occurs in counties bordering Canada, particularly in northeast and north-central Minnesota. Fisher harvest occurs in most of the northern 50% of the state, though harvest is comparatively low in extreme northeast Minnesota (Lake and Cook counties), and low, though perhaps increasing, in the Red River Valley (western Minnesota) and the highly fragmented transitional forests in central Minnesota. Recent harvest levels have been near 3,500 and 2,500 for marten and fisher, respectively. However, due to apparent multi-year population declines in both species, harvest seasons have been reduced from 16 days to 9 days for the past 2 seasons, reducing harvest by approximately 50%.

While both species appear to have naturally re-colonized a significant portion of their historic range, Minnesota-specific information on species biology and ecology is limited. Except for carcass data obtained from harvested fisher and marten, we are aware of only 1 published field study in Minnesota. Specifically, Mech and Rogers (1977) opportunistically radiocollared 4 marten and reported survival and home range information for those animals. This information is now nearly 30 years old, and based on a very limited sample size. The low reproductive potential, low density, and comparatively specialized habitat requirements of fisher and marten make them more susceptible to over-harvest and the negative effects of human development and habitat alteration.

The primary objectives of this study are to: (1) estimate survival rates and causes of mortality for fisher and marten in Minnesota; (2) describe and quantify features of natal den sites used by females; (3) directly estimate parturition rates and, if possible, litter size and kit survival; (4) evaluate variability in survival or reproduction as a function of forest attributes, prey abundance and weather conditions; and (5) to evaluate the design of winter track surveys.

After initial evaluation of field methods during the pilot year of the study (Erb et al. 2008), winter 2008-09 marked the beginning of full-scale research activities. Herein we present basic summaries of field methods and preliminary findings for key objectives, particularly radiocollaring activities and survival and reproductive monitoring. More detailed analyses and other objective-specific methods (e.g., stable isotope analysis, home range/habitat analysis, track survey evaluation) will be detailed in future years as results become available.

## STUDY AREA

Marten research is focused on 1 study area located in northeastern Minnesota (Figure 1; Area 1). The area (~ 700 km<sup>2</sup>) is composed of 69% mixed forest, 15% lowland conifer or bog, 5% upland coniferous forest, 4% gravel pits and open mines, 3% regenerating forest (deciduous and coniferous), 2% shrubby grassland, 1% marsh and fen, 1% open water, and 0.4% deciduous forest. The area is 90% public ownership, including portions of the Superior National Forest and state and county lands. Fishers are also present in this area at low to moderate density.

Fisher research will take place in 3 areas (Figure 1; Areas 1, 2, and 3). The work in Area 3 is a collaborative effort between Camp Ripley Military Reservation, Central Lakes Community College, and the Minnesota Department of Natural Resources. While we do include animals captured in that area in our summary of sample sizes and known-fates, we do not discuss other aspects of that project in this report. Area 2 (1075 km<sup>2</sup>), our primary fisher study area, is composed of 74% deciduous forest, 11% open water, 5% lowland conifer or bog, 5% marsh and fen, 2% regenerating forest (deciduous and coniferous), 1% coniferous forest, 1%

grassland, and 1% mixed forest. Area 2 is 67% public ownership, including portions of the Chippewa National Forest and State and county lands. Extremely few martens occupy Area 2.

## METHODS

We used cage traps to capture both fishers (Tomahawk Model 108) and martens (Tomahawk Model 106 or 108) during winter. Traps were baited with either deer (*Odocoileus virginianus*) or beaver (*Castor canadensis*) meat, and commercial lure was placed in or above the traps. We enclosed traps inside white plastic 'feed sacks' or burlap bags and further covered traps with natural vegetation. All traps were checked daily.

To immobilize animals, we used metal 'combs' to restrict the animal to a small portion of the trap, or restrained the animal against the side of the trap by pulling its tail through the cage mesh. Animals were injected with a hand-syringe using a 10:1 mixture of ketamine and xylazine (fisher: 30 mg/kg ketamine and 3 mg/kg xylazine; marten: 20 mg/kg ketamine, 2 mg/kg xylazine) (Kreeger et al. 2002). After processing, the xylazine was reversed with yohimbine at a dosage of 0.1 mg/kg (marten) or 0.15 mg/kg (fisher). Fisher were either ear-tagged with a monel # 3 tag in one ear (National Band and Tag Co., Newport, KY) and a 2-piece plastic mini-tag (Dalton I.D. Systems, UK) in the other ear, or with a monel # 3 tag in both ears. Marten were ear-tagged with a monel #1 tag (National Band and Tag Co., Newport, KY) in each ear.

During processing, animals were placed on either chemical hand warmers or heating pads connected to a power inverter and 12 volt battery. Portable propane heaters were also used to keep animals warm during processing. We monitored respiration, pulse, and rectal temperature during anesthesia. We weighed and sexed animals and typically removed a first pre-molar for aging. Morphological measurements taken included body length, tail length, hind foot length, and chest, neck, and head circumference. We removed guard hair samples for possible genotyping, and for evaluating the use of stable isotope analysis for deciphering food habits (Ben-David et al. 1997). To determine which females were pregnant in mid-winter, and eventually the percent of those that actually produce a litter in spring, we attempted to draw blood samples to measure serum progesterone levels (Frost et al. 1997). After gaining some experience the first year, we were usually successful at drawing blood from female fisher the second year, but have been largely unsuccessful at drawing blood from female marten. Antibiotics were administered subcutaneously to all animals prior to release. All blood samples were sent to the University of Minnesota Veterinary Diagnostics Lab for progesterone analysis.

During the pilot year, we deployed several radiocollar designs on fisher, including an ATS M1585 zip-tie collar (~ 43 g), an ATS M1930 collar (~ 38 g), and a Lotec SMRC-3 collar (~ 61 g; deployed on adult males only). During 2008-09 collaring efforts, we primarily deployed ATS M1940 collars (~ 43 g) or Sirtrack TVC-162 collars (~ 45 g) on fisher. The majority of martens in both years have been fitted with a Holohil MI-2 collar (~ 31 g).

In an effort to better understand winter activity patterns and frequency of winter rest site use, we evaluated the potential use of miniature temperature loggers (iButton model DS1922-L, Maxim Integrated Products, Sunnyvale, CA) attached to collars. iButtons were epoxied to a sample of 8 fisher collars and 8 marten collars. We programmed them to record temperature at 30 minute intervals. Recovering the data requires recapturing the animal, or recovering the collar if the animal dies or slips the collar. Temperature monitors were also placed in each of 6 cover types in both Study Area 1 and 2 to allow for analysis of the effects of ambient temperature on animal behavior, reproductive success, and survival. Snow depth readings were also recorded throughout winter along a transect in each of the 6 cover types where the temperature monitors were placed.

All radiolocations, except for some taken during the den-monitoring period, will be obtained from fixed-wing aircraft at approximately weekly intervals. During the pilot year, and periodically thereafter, we will test the accuracy of aerial radiolocations by placing transmitters in known locations of varying forest structure, and compute the mean distance between known and estimated locations. Detailed information on radiolocation methods and analysis will be presented in future years.

While data is absent for Minnesota, nearly all reported fisher natal dens have been in elevated tree cavities (Powell et al. 2003). Marten natal dens are also frequently in tree cavities (Gilbert et al. 1997), but may occur in more varied features (e.g., under-ground burrows, exposed root masses of trees, rock piles, large downed logs; Ruggiero et al. 1998). We primarily used ground and aerial radiolocations to locate natal den sites, but also deployed remotely-activated cameras (Reconyx PC-85 or RC-55, Reconyx, Inc, Holmen, WI) at suspected den sites to monitor female activity (Jones et al. 1997). If a suspected natal den was located in a tree cavity, we used an MVC2120-WP color video camera (Micro Video Products, Bobcaygeon, Ontario) attached to a fiberglass telescoping pole and connected to a laptop computer to confirm and count kits. Underground dens were examined when possible using the same video probe attached to a flexible rod. Dens were only examined when the radiomarked female was not present. After initial den and litter confirmation, we hope to relocate (if moved) kits at 30-day intervals (up to 120 days) to evaluate kit survival and determine which females recruit at least 1 offspring to the fall population.

Prey sampling transects have been established in both study areas. Prey sampling is being conducted primarily to understand within-area trends in prey abundance, and whether trends are correlated with any observed changes in fisher or marten survival and reproductive success. Transects ( $n \approx 150$  in each study area) consist of 10 sampling locations spaced 20m apart, distributed in various cover types throughout the study area. Transects will generally be oriented perpendicular to roads or trails, with the first plot 30m off the trail. In spring, we will count snowshoe hare (*Lepus americanus*) pellets in a 1-m<sup>2</sup> plot at each sampling station (McCann et al. 2008). During fall, small mammal snap-trapping will occur for 2 consecutive days at the same sampling stations, similar to protocol used on an existing small mammal survey in Minnesota (Olson 2006). During both spring (hare pellet sampling) and fall (small mammal trapping), we will also count the number of red squirrels (*Tamiasciurus hudsonicus*) observed or heard along each transect. Rather than using 10-min point counts (Mattson and Reinhart 1996, Bayne and Hobson 2000), with our small mammal/hare pellet stations as the sampling points, we will simply record the number of unique squirrels observed/heard along each transect while checking pellet plots and small mammal traps. Information on white-tailed deer and ruffed grouse (*Bonasa umbellus*) populations may be available from existing surveys or population models.

## RESULTS AND DISCUSSION

Including the pilot year of the study, a total of 47 martens (25M, 22F) and 30 fishers (11M, 19F) have been radiocollared. Tooth aging has not yet been completed, and herein we do not report any formal survival estimates. Instead, we provide a simple overview of the fate of collared animals.

Of the 47 martens collared, 7 individuals (2M, 5F) were able to subsequently slip their collars, 6 within the first 3 months after release, and 1 more than a year after release. In addition, 1 animal has not been relocated since shortly after capture. Of the remaining 39 animals whose status is known, 26 are currently alive, 6 have died from predation (3 raptor kills, 3 mammalian carnivores), 3 were harvested during the legal trapping season, 2 died from capture or collar-related complications, and 2 are missing.

Nine of the 30 fisher have also shed their collars, 8 due to insufficient collar design, and 1 possibly slipped by the animal (with some evidence of other animal or human involvement). Most of these came off within the first 2 months after release. Of the remaining 21 fisher, 14 are alive, 1 has not been relocated since release, 5 have died from predation (2 raptor kills, 3 mammalian carnivores), 1 additional collar (but no animal remains) was found under an active eagle nest, and 1 animal was trapped outside of the harvest season.

This spring represented the first concerted effort to locate natal dens and confirm kit presence. Based on several 'measures', we obtained reproductive information for 13 female fishers (Table 1). Of these, 8 produced litters and 5 were not pregnant. Because age information is not yet available for these animals, we cannot determine whether the females that

did not produce litters were of pre-reproductive age, or reproductive-age animals that failed to produce a litter. Mean litter size for the 8 females with confirmed litters was 2.9 (Table 1). All natal dens were located in tree cavities, most commonly large-diameter aspen trees or snags (Table 1). We were unable to determine exact dates of birth, but most kits appear to have been born in the first 10 days of April. To date, 4 nursing females have been killed by other predators, indirectly leading to the death of 12 kits.

Reproductive information for female marten is not yet available. We were unsuccessful at drawing blood from marten, so we do not have pregnancy status data (mid-winter progesterone level) at the time females were captured. Examination of the reproductive tract from a dead female marten indicated she may have been 1-5 days post-implantation in early April, and that birth may occur in early to mid-May. We have confirmed that 2 of the 12 have given birth, while the status of the others is unknown at the time of this writing. It appears that the den attendance patterns of marten make it more challenging to confirm natal dens, at least compared to fisher. Intensive monitoring will continue over the next few weeks.

As part of collar/mortality retrieval efforts and ground checks on potentially denned females, we have had opportunity to document and examine various rest/den sites for both species. Throughout winter, most resting sites we have located for marten are on or below ground (or below snow), most commonly in tunnels through the mossy substrate of lowland conifer stands, but also in tunnels in boulder-laden soil, near the base of trees, or near exposed tree roots. During spring and summer, marten rest sites appear more varied, including those used during winter as well as increased use of above-ground features (tree cavities, hollow logs, and 'witches brooms'). While this sparse and opportunistic sample of resting sites is inadequate to draw any strong conclusions, it appears that martens may primarily use on- or below-ground dens in winter, with increasing use of above-ground sites in other seasons.

The majority of fisher rest sites, regardless of season, have been in tree cavities or hollow logs on the ground. However, we have documented use of a slash pile, an abandoned beaver dam, an abandoned muskrat (*Ondatra zibethicus*) or beaver bank burrow, a tree-squirrel leaf nest, and an underground burrow in an upland area. During winter, we confirmed that 1 female rested in a hollow log, without leaving, for 12 days.

Temperature sensor data has been recovered from iButtons attached to 4 fisher collars. While data has not been formally analyzed, herein we illustrate the type of information that may be obtained from the sensors by presenting data obtained from 1 female fisher. Based on preliminary examination, we believe that temperature readings for an animal in a den during winter are typically above 80°F (see Figure 2 (top) as an example), noting that the temperature in this case is a result of the thermal influence of both the den and the animal's body. As an animal leaves a den, ambient temperature has an increasing influence on temperature readings, while the animal's body temperature has a decreasing influence (though always has some influence). For purposes of discussion here, we assumed the animal was using a rest site if temperature readings were above 60°F. Using this criterion, female fisher F09-370 appeared to quickly return to a den site after capture/release, and remained there, with 1 brief exception, for 3 days (Figure 2 - middle). From Jan. 2 – March 31, we estimate that she spent 72% of her time at a den/rest site (81%, 69%, and 66% during January, February, and March, respectively).

While the temperature threshold for distinguishing den/rest site use versus activity will change, or perhaps not be distinguishable, as ambient temperature increases, we believe our preliminary thresholds are still applicable for much of April in northern Minnesota. Examination of data for female fisher F09-370 during the first 18 days of April suggests that her behavior changed significantly around April 9 (Figure 2- bottom). We first confirmed the presence of kits in her natal den on May 4, and suspected them to be approximately 3 weeks old. We believe the temperature sensor data suggests that she gave birth on or around April 10 when her behavior appears to have changed significantly (Figure 2 – bottom). If our above interpretation and preliminary temperature threshold is correct, she spent substantially more time away from a den post-birth than during winter (72% of her time in a den/rest site during winter, only 27% of the time in a den/rest site in the 8 days following birth). Data suggests that she returned to a den for brief intervals, and consistently around 3:30 am each morning, presumably to nurse.

Her extended time away from the den during this period may reflect the energetic demands of lactation and the need to hunt for food more often. Fisher are also known to breed in the first week or two following birth. Her increased activity outside a den/rest site may explain the apparent increase in predation mortality of nursing females following birth (no females died during winter, 4 were killed in the first 6 weeks post-birth).

Home range analysis has yet to be conducted. However, visual examination of the data suggests that marten home ranges may be from 1–3 mi<sup>2</sup>, while fisher home ranges may be from 3–5 mi<sup>2</sup>. The longest distance a fisher has dispersed from its capture location is approximately 18 miles, and several fishers have moved around 8 miles. One marten moved approximately 8 miles from its capture location, spent nearly a year in a new area, and recently returned close to its original location before being killed by a mammalian carnivore. Another marten dispersed approximately 5 miles before its signal was lost.

Because only 1 year of prey survey data has been collected, limited inference can be drawn. Based on the first year (2008) of surveys, hare abundance in both study areas was highest in regenerating forest, followed by lowland conifer and upland conifer. Hares were least abundant in mature upland deciduous forest. Pooling cover types, average hare pellet counts were 2 times higher in the fisher study area (Area 2, Figure 1) than the marten study area (Area 1, Figure 1), though confidence intervals overlap. Conversely, the small mammal capture rate (cover types pooled), dominated in both areas by red-backed voles (*Myodes gapperi*) and *Peromyscus* spp., was nearly 3 times higher in the marten study area (Area 1, Figure 1) than in the fisher study area (Area 2, Figure 1), solely a result of lower red-backed vole abundance in Area 2. More details on prey sampling methods and results will be presented in future years.

## FUTURE PLANS

Throughout early summer 2009, we will be continuing to confirm natal or maternal dens of marten, and will attempt to monitor survival of any confirmed marten litters, as well as for litters of fisher kits already confirmed. We will also begin collection of vegetative information from individual fisher and marten home ranges. Sampling will occur in a pre-determined number of 0.04-acre circular plots within each home range. We will collect quantitative data on: (1) tree DBH and height, and ultimately basal area and volume of trees, by species; (2) % canopy cover (deciduous and coniferous); (3) sapling density; (4) understory cover density; (5) density and volume of snags; (6) density, volume, and other characteristics of coarse woody debris; and (7) density and volume of exposed root masses.

Tissue samples from prey species, as well as hair samples from fisher and marten, will be sent to a lab for stable isotope analysis. If species-specific chemical signatures of prey are sufficiently distinct, we will further assess the potential for describing late-summer/fall food habits of fisher and marten based on chemical analysis of their guard hair.

## ACKNOWLEDGEMENTS

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Table 1. Reproductive data for radiocollared female fisher in Minnesota, 2009.

ID	Pregnancy		# kits	Litter info	
	Status	Method		Method	Natal den
F08-353	Y	Kits	3	Video	Tree cavity; 23" dbh live aspen
F08-375	Y	Kits	3	Video / uterine scars	Tree cavity; 22" dbh w.pine snag
F09-360	Y	Progesterone/Kits	2	Video	Tree cavity; 15" dbh aspen snag
F09-362	N	Behavior	0	--	--
F09-364	N	Progesterone	0	--	--
F09-376	N	Progesterone	0	--	--
F09-380	Y	Progesterone/Kits	3	Video	Tree cavity; 24" dbh aspen snag
F09-390	N	Progesterone	0	--	--
F09-394	Y	Kits	3	Video	Tree cavity; 22" dbh live aspen
F08-077	Y	Nursing	4	Uterine swellings / scars	Died before den located
F08-304	Y	Nursing	2	Uterine swellings / scars	Died before den located
F09-354	N	Behavior	0	--	--
F09-370	Y	Kits	3	Video	Tree cavity; 16" dbh aspen snag



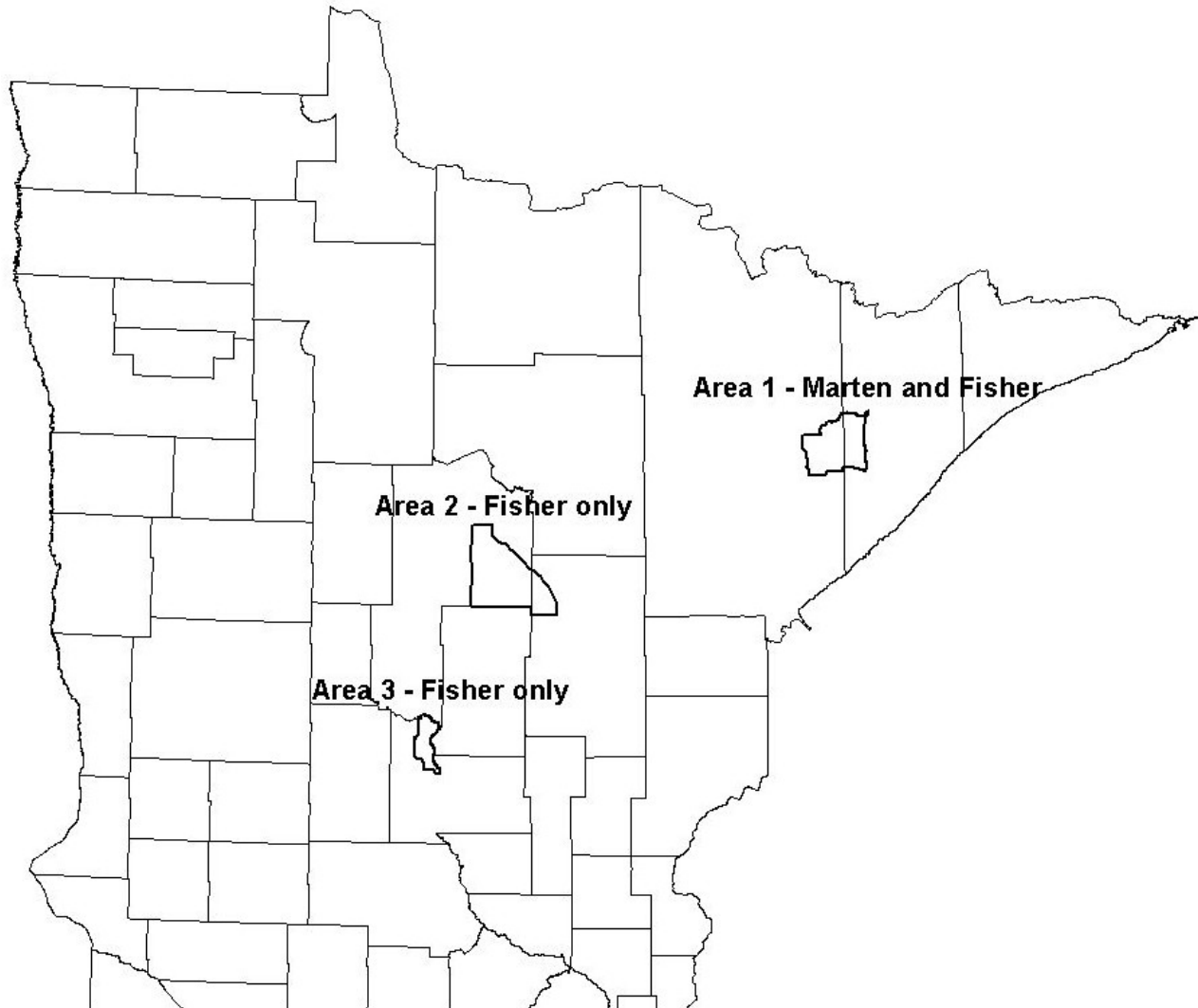


Figure 1. Fisher and marten study areas in Minnesota, 2008-09.

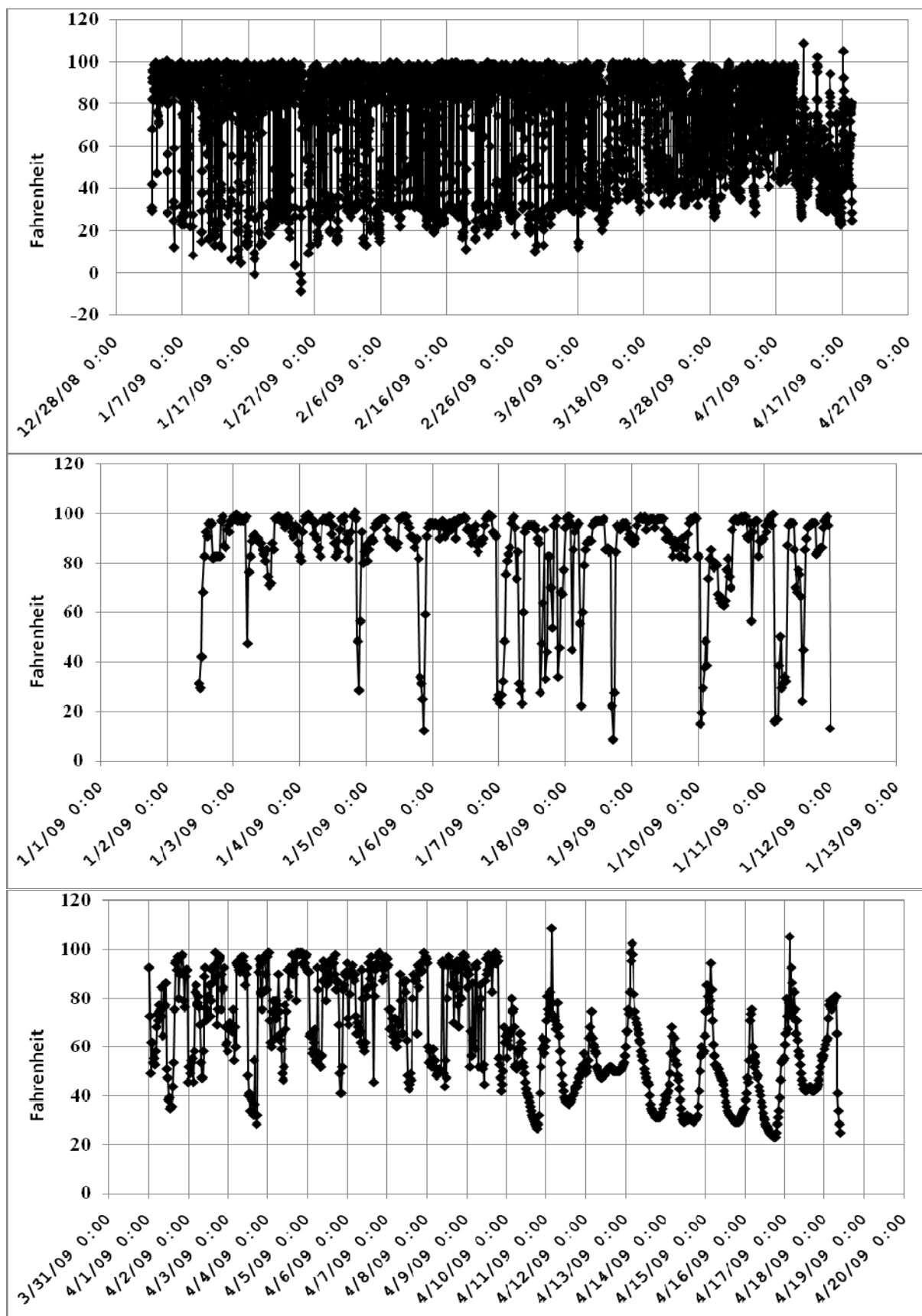


Figure 2. Temperature readings every 30 minutes from the radiocollar of a female fisher in Minnesota (Top: all readings from Jan. 2, 2009 – April 18, 2009; Middle: Readings for the first 10 days following release; Bottom: readings for the first 18 days of April)

## DYNAMICS OF A MINNESOTA MOOSE POPULATION IN A WARMING CLIMATE<sup>1</sup>

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### SUMMARY

1. Species on the southern edge of their distribution are especially at risk to climate-induced changes. One such species is the moose (*Alces alces*), whose continental United States distribution is restricted to northern states or northern portions of the Rocky Mountains cordillera. Moose are particularly vulnerable to climate change because of their intolerance to heat. Here, we examine the demographic implications of estimated survival and reproduction schedules for a moose population in northeastern Minnesota, USA, between 2002 and 2007.

2. Estimated age- and year-specific survival rates showed a sinusoidal temporal pattern during the course of the study and were lower for younger and old aged animals. Estimates of annual adult survival (constant across age classes) ranged from 0.74 – 0.85. Annual calf survival averaged 0.40 and the annual ratio of calves born to radiocollared females averaged 0.78.

4. Point estimates for  $\lambda$  from yearly matrices ranged from 0.71 to 0.97 during the 6-year study, indicative of a long-term declining population. Assuming each matrix to be equally likely to occur in the future, we estimated a long-term stochastic growth rate of 0.85. Population growth rate, and its uncertainty, was most sensitive to changes in estimated survival rates.

5. Maximum daily temperature at Ely, Minnesota, USA increased between 1960 and 2007 and displayed several short-term fluctuations. The telemetry data for this study were collected during a period with the highest maximum temperature values.

6. If heat stress is responsible for current levels of survival and temperatures continue to increase, survival rates are likely to decline even further and ultimately the southern edge of moose distribution will shift northward into Ontario.

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<sup>1</sup>Summary of paper submitted to Journal of Animal Ecology.

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## TEMPERATURE RELATIONSHIPS IN MOOSE HABITAT: A PILOT

Mark S. Lenarz

### SUMMARY OF FINDINGS

I deployed Hobo data loggers at 10 weather stations in northeastern Minnesota to determine the magnitude of variation in temperature metrics in various cover types used by moose (*Alces alces*). Maximum ambient temperature and dew point temperature varied little among stations. Maximum black globe temperature displayed more variation, presumably in response to differences in canopy closure at the weather stations. Soil moisture apparently affected variability in soil temperatures. Heat stress indices calculated from these metrics displayed little variation among stations.

### INTRODUCTION

Heat stress as experienced by moose is dependent on their immediate environment. Because of their dark color, moose are likely to absorb more solar radiation than light colored animals and more likely to benefit from using habitat as a thermal refuge. Ambient air temperature as used by Lenarz et al. (2009) as a measure of heat stress, explained variation in seasonal and annual moose survival but solar radiation and humidity may be even more important to the level of stress experienced by moose. Research into the effects of heat stress on humans and domestic cattle have utilized 2 different indices for heat stress. The Wet Bulb Globe Temperature (WBGT) is a composite of ambient air temperature, solar radiation, and humidity that was originally developed to determine appropriate exposure levels for humans (U. S. Army 2003). A similar measure, the Temperature Humidity Index (THI) was developed to determine the effect of summer conditions on human discomfort (AMS 2000).

The primary objective of this research was to determine the magnitude of variation in temperature metrics used to calculate WBGT and THI across a sample of habitats known to be used by moose. I also wanted to evaluate the limitations of a variety of data logging equipment that would be needed in any future research evaluating the thermal value of moose habitat.

### METHODS

To calculate WBGT, 3 metrics are required: the ambient air temperature in °C ( $T_a$ ), the dew point temperature in °C ( $T_d$ ), and the black globe temperature in °C ( $T_g$ ).

WBGT is derived from the formula:  $WBGT = 0.7T_d + 0.2T_g + 0.1T_a$ .

THI is derived from the formula:  $THI = 0.55T_a + 0.2T_d + 5.3$ .

At each station, I used a Hobo data logger (Onset, U23-001, Bourne, MA) enclosed in a thermometer shelter (Ben Meadows, Janesville, WI) to record  $T_a$  and  $T_d$ . The shelter was located approximately 1 m above the ground facing north. I used a Hobo (U23-003) enclosed within the shelter with 2 external probes, one of which was inserted into a 6" copper globe (Naugatuck Manufacturing, Waterbury, CT) painted flat black to measure  $T_g$ . The globe was located approximately 1 m from the shelter and 0.5 m above the ground. The second probe of this data logger was buried approximately 15 cm in the ground to measure soil temperature ( $T_s$ ). I also monitored light intensity ( $Lux/m^2$ ) as a covariate using a pendant Hobo UA-002064 that was hung on a hook on the north side of the shelter. Measurements were recorded every 30 minutes.

## RESULTS AND DISCUSSION

I deployed 6 weather stations on 7 Aug 2008 and 4 stations on 22 Aug 2008 and removed all stations on 12 Sep 2008. Weather stations were located in LandSat-Based Land Use Land Cover (Manitoba Remote Sensing Centre) cover types designated as Conifer Forest, Wetland Bog, Regenerated Young Forest, and Mixed Forest. I attempted to get a range in canopy closure across the locations. The most distant stations were separated by approximately 13 km.

The Hobo data loggers within the thermometer shelters recorded  $T_a$  and  $T_d$  without any problems. At 2 stations, however, snowshoe hare (*Lepus americanus*) or deer (*Odocoileus virginianus*) chewed through the cables connected to the external probes and all data for  $T_g$  and  $T_s$  were limited. In addition, the external probe fell out of 1 globe and all subsequent data on  $T_g$  were lost. Analyses involving  $T_s$  and  $T_g$  were restricted to the 7 functioning stations. All pendant data loggers were effective at recording light intensity. Any future studies using external probes should protect the cables (e.g. copper tubing) to prevent loss of data.

Maximum ambient temperature ( $T_a$ ) and dew point ( $T_d$ ) varied little among the 10 stations. Restricting data to the period 22 Aug to 12 Sep (when all stations were operational), the mean maximum daily  $T_a$  ranged from 18.2 – 20.7 °C ( $\bar{x} = 19.4$ , SE = 0.3,  $n = 10$ ) and  $T_d$  ranged from 12.3 – 13.1 °C ( $\bar{x} = 12.7$ , SE = 0.1,  $n = 10$ ). In contrast, mean maximum  $T_g$  ranged from 19.5 – 29.1 °C ( $\bar{x} = 25.3$ , SE = 1.4,  $n = 7$ ) which likely reflects differences in canopy closure among stations. Soil temperature ( $T_s$ ) was bimodal. Several stations were consistently warmer and tended to be correlated to  $T_a$  (e.g. station 1,  $r^2 = 0.816$ ). Other stations were 3 to 4°C cooler with little daily variation and were not correlated with  $T_a$  (e.g. station 4,  $r^2 = 0.293$ ). I suspect that the latter stations had water-saturated soils. Temperatures at the more variable stations declined and stabilized after 2 significant rain events. The amount of shading (total lux/m<sup>2</sup>/day) varied considerably among stations with an 11 fold difference between station 1 (29,600 lux/m<sup>2</sup>/day) and station 3 (2,600 lux/m<sup>2</sup>/day).

Despite substantial differences in the degree of shading, there was little variation in WBGT among stations. Mean maximum daily WBGT ranged from 13.6 – 15.9 ( $\bar{x} = 15.0$ , SE = 0.3,  $n = 7$ ). Daily maximum values of WBGT ranged as high as 24.3 during the period 7 Aug to 12 Sep 2009. Considering that  $T_g$  represents only 20% of WBGT, the lack of variation among stations is not surprising.

The Temperature Humidity Index (THI) incorporates only  $T_a$  and  $T_d$ , neither of which varied substantially among stations. Maximum daily THI ranged from 23.8 to 25.3 ( $\bar{x} = 24.5$ , SE = 0.2,  $n = 10$ ).

The difference between black globe temperature ( $T_g$ ) and ambient air temperature ( $T_a$ ) is a measure of the increased energy that a moose would absorb as a result of radiant energy (mostly solar). The maximum daily value for this variable was a linear function of the total amount of light at the site (Figure 1). This relationship, however, varied according to the site. At 3 stations (1, 8, and 9) where the total daily light (lux/m<sup>2</sup>/day) averaged 12,887 (1,637-29,600), the slope of the regression line was considerably higher (6.7E-4,  $r^2 = 0.868$ ) while at 2 stations (2 and 4) where total daily light averaged 76,800 (5,100-137,900) the slope was 1.0E-4 ( $r^2 = 0.935$ ). Most likely, the increased shrub layer at stations 1, 8, and 9 was radiating additional energy as measured in  $T_g$ .

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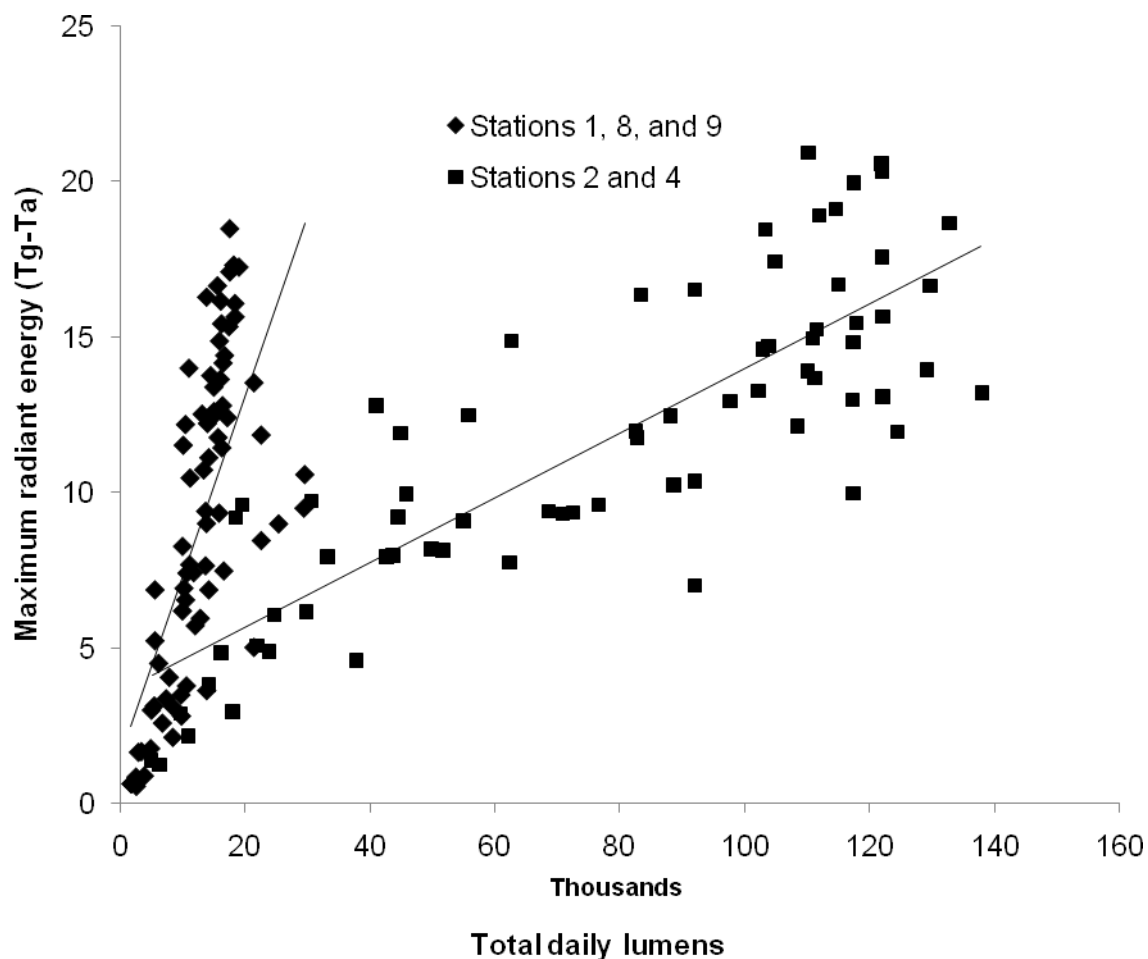


Figure 1. Maximum radiant energy as a function of light as measured at 5 weather stations in northeastern Minnesota, USA. At 3 stations (1, 8, and 9) the slope of the regression line was considerably higher ( $6.7\text{E-}4$ ,  $r^2 = 0.868$ ) than at the other 2 stations (2 and 4) where the slope was  $1.0\text{E-}4$  ( $r^2 = 0.935$ ). Most likely, the increased shrub layer at stations 1, 8, and 9 was radiating additional energy as measured in  $T_g$ .

## ASSESSING WINTER BODY CONDITION OF MOOSE (*ALCES ALCES*) IN A DECLINING POPULATION IN NORTHEASTERN MINNESOTA<sup>1</sup>

Glenn D. DelGiudice, Barry A. Sampson, Mark S. Lenarz, Michael W. Schrage,<sup>2</sup> and Andrew J. Edwards<sup>3</sup>

### ABSTRACT

Because winter nutrition of moose (*Alces alces*) and other northern ungulates has been strongly associated with mortality and reproduction, assessments of the condition of individuals may be particularly informative to understanding the dynamics of populations and other influential factors. During February–March 2003 to 2005, we assessed the nutritional condition of 79 moose (39 females, 40 males) in a declining population in northeastern Minnesota by ultrasonographic measurement of rump fat (Maxfat) and 2 body condition scoring (BCS) systems (whole body and rump-only). Our objective was to compare the 2 BCS techniques, relate them to a more quantitative measure of condition, and determine if condition was a contributing factor to non-anthropogenic mortality documented in a companion study and to pregnancy status of these moose. Scores of the 2 BCS systems were correlated ( $r = 0.81$ ;  $P < 0.0001$ ), and each was related to Maxfat ( $r^2 = 0.34$ – $0.35$ ) and ingesta-free body fat (IFBF;  $r^2 = 0.37$ – $0.41$ ), estimated from Maxfat. Body condition scores of males were less ( $P \leq 0.009$ ) than those of females, and there was a significant ( $P = 0.021$ ) sex  $\times$  capture-year effect on Maxfat (and IFBF) with no effect of age. Mean estimated IFBF was 9.9% ( $\pm 0.5$  [SE], range = 2.5–15.0%) for females and 8.8% ( $\pm 0.3$ , range = 6.2–11.4%) for males. During winter 2003, when the pregnancy rate was 55%, mean IFBF of females was 24–25% less than during 2004 and 2005 when all females were pregnant. During winters 2004 and 2005, mean Maxfat values, indicative of 7.8–11.5% IFBF, were consistent with winter and spring survival rates ( $\geq 95\%$ ). However, over all 3 winters, IFBF ( $< 5\%$ ) of 15–21% of these moose was indicative of a more compromised probability of subsequent survival. We did not observe a direct relationship between winter condition and non-anthropogenic mortality of these moose, but collective evidence suggests that heat stress, implicated in reported relations between January, spring, and “warm-season” temperatures and non-winter and annual survival of these moose, may have imposed an additive and cumulative adverse effect on their condition not detected by our sampling. Specific relations among nutritional limits, seasonal heat stress, and the use of thermal refuges by moose require more comprehensive and in-depth investigation.

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## MOOSE POPULATION DYNAMICS IN NORTHEASTERN MINNESOTA

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### SUMMARY OF FINDINGS

We captured and radiocollared a total of 150 adult moose (*Alces alces*; 55 adult males and 95 adult females) between 2002 and 2008. As of 1 April 2009, 100 collared moose (46 adult males and 54 adult females) have died. Annual mortality rates varied among years, and generally were higher than found elsewhere in North America. Estimates of fertility for this population were also low compared with other North American moose populations. Data analyses from this research are progressing and 1 manuscript was published and 2 other manuscripts were submitted for publication.

### INTRODUCTION

Moose formerly occurred throughout much of the forested zone of northern Minnesota, but today are restricted to the northeastern-most counties including all of Lake and Cook Counties, and most of northern St. Louis County. We initiated a research project in 2002 to better understand the dynamics of this population. Fieldwork on the first phase of this project ended in early 2008 and we are in the process of analyzing data and preparing manuscripts that discuss results from the first phase. The following report will discuss preliminary findings.

The project was a partnership between the Minnesota Department of Natural Resources, the Fond du Lac Band of Lake Superior Chippewa, the 1854 Treaty Authority, and the U. S. Geological Survey. A second research project was initiated in February 2008 with funding secured by the Fond du Lac Band. The Minnesota Department of Natural Resources and 1854 Treaty Authority will provide in-kind support and limited funding for this second phase of research.

### METHODS

We captured a total of 116 moose in southern Lake County and southwestern Cook County between 2002 and 2005, attached radiocollars, and collected blood, hair, fecal and tooth samples. See Lenarz et al. (2009) for greater detail on the study area and research methods. In February 2008, we captured and sampled an additional 34 moose and attached radiocollars. We monitored a sample of up to 78 radiocollared moose weekly to determine when mortality occurred. We calculated annual non-hunting mortality rates ( $1 - \text{survival}$ ) using the Kaplan-Meier procedure (Kaplan and Meier 1958) modified for a staggered-entry design (Pollock et al. 1989) and censored all moose killed by hunters, those that died from capture mortality, moose that had emigrated from the study area, and apparent transmitter failure. We used a Cox Proportional Hazard (CPH) model (Cox 1972, SAS PROC PHREG, SAS Institute 2008) to test for a difference in annual survival between sexes. Beginning in 2004, we used helicopter surveys in late May – early June (MJ) to estimate fertility of radiocollared females and additional surveys in late April – early May (AM) to estimate survival of calves born the previous spring.

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## RESULTS AND DISCUSSION

As of 1 April 2009, 100 of the 150-radiocollared moose (46 adult males and 54 adult females) have died. In addition, 1 moose slipped its collar, 1 moose moved out of the study area, and we lost contact (apparent transmitter failure) with 4 moose. Moose that died within 2 weeks of capture (6) were designated as capture mortality. Hunters killed 16 moose, 2 were poached, and 11 were killed in collisions with vehicles (cars, trucks, or trains). The remaining mortality (65) was considered to be non-anthropogenic and causes included wolf predation (6), bacterial meningitis (1), or unknown (58).

The unknown mortality appeared to be largely non-traumatic. In 50% of the cases, the intact carcass was found with only minor scavenging by small mammals or birds. Wolves and bears were the primary scavengers in 40% of the remaining cases. We were unwilling to attribute predation as the cause of death in these cases because there was little evidence that a struggle had preceded death. In 10% of the cases, we were unable to examine the carcasses or only found a collar with tooth-marks.

Annual non-hunting mortality rates (1 June to 31 May) for adult moose averaged 21% for males (7 to 40%, SE = 5,  $n = 6$ ) and 21% for females (5 to 30%, SE = 4,  $n = 6$ ; Table 1). Sex did not contribute to the prediction of survival ( $\chi^2 = 0.001$ ,  $P = 0.98$ ), which implies that there was no difference in survival rates (non-hunting) between adult male and female moose. 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) and similar to that observed for adult moose in northwestern Minnesota (Murray et al. 2006).

Serum samples from 91 radiocollared adult female moose were collected and analyzed using radioimmunoassay for levels of serum progesterone between 2002 and 2008. Using a pregnancy threshold of 2.0 ng/ml progesterone, annual pregnancy rate varied from 55 to 100% ( $\bar{x} = 80\%$ , SE = 8,  $n = 5$ ). Boer (1992), in his review of moose reproduction in North America found that adult pregnancy rate across North America averaged 84%. Although pregnancy rate of yearling moose is reduced (Schwartz 1968), our sample included only 1 yearling moose. Our estimates may be biased low because 4 cows that tested negative in 2003 (55% pregnancy rate) were subsequently observed with a calf.

Between 2004 and 2008, 161 radiocollared adult females gave birth to a minimum of 131 calves (76 singles, 26 twins, and 1 set of triplets; Schrage unpublished). The annual ratio of calves: radiocollared females ranged from 0.53 to 0.96 ( $\bar{x} = 0.79$ , SE = 0.07,  $n = 5$ ). These estimates were biased low because in 3 of 4 years, radiocollared females not accompanied by calves during the MJ survey were subsequently observed to be accompanied by a single calf (4 in 2004, 2 in 2005, 1 in 2007). It is also possible that post natal mortality occurred prior to the MJ survey. Nonetheless, these estimates are low compared with other locations in North America. Boer (1992), for example, reported estimates ranging from 0.88 to 1.24 calves/adult female, in moose populations above and below carrying capacity, respectively.

During the past year, 3 manuscripts discussing the results of this research have been prepared for publication. The first, entitled "Temperature mediated moose survival in northeastern Minnesota" was published in the May 2009 issue of the Journal of Wildlife Management. A second manuscript, entitled "Assessing winter body condition of moose (*Alces alces*) in a declining population in northeastern Minnesota" has been submitted to the Journal of Mammalogy. The final manuscript, entitled "Dynamics of a Minnesota moose population in a warming climate" has been submitted to the Journal of Animal Ecology. At least 2 additional manuscripts are planned from the data collected during the first phase of this research.

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Table 1. Annual adult mortality of moose in northeastern Minnesota, USA. Estimates censored for hunting, capture mortality, and apparent transmitter failure. Mortality calculated for period 1 June to 31 May.

Year	Male	Female	Combined
2002 <sup>1</sup>	7% (25) <sup>2</sup>	30% (29)	23% (54)
2003	25% (21)	20% (34)	21% (55)
2004	8% (32)	5% (42)	6% (74)
2005	24% (21)	29% (30)	26% (51)
2006	40% (10)	27% (22)	31% (32)
2007	20% (8)	19% (49)	18% (57)
Mean	21%	21%	21%

<sup>1</sup> Period: 1 June 2002 – 31 May 2003.

<sup>2</sup> Sample size as of 31 May.

## A REVIEW OF THE ECOLOGY OF *PARELAPHOSTRONGYLUS TENUIS* IN RELATION TO DEER AND MOOSE IN NORTH AMERICA

Mark S. Lenarz

### SUMMARY OF FINDINGS

It is well established that white-tailed deer (*Odocoileus virginianus*) are the normal host for *P. tenuis* and that this parasite either kills moose (*Alces alces*) directly or predisposes them to other causes of mortality. Despite the historical record of moose dying from this parasite, there is little evidence, that *P. tenuis* is a major cause of mortality in moose or that it was responsible for historic declines in moose populations. When white-tailed deer expanded their range northward into moose range following logging, they undoubtedly introduced *P. tenuis* to moose. While it may seem intuitive that higher deer numbers should translate into higher moose mortality, research has not corroborated this relationship. Rather, it has discovered that the transmission of *P. tenuis* between deer and moose is a complex relationship and after almost 45 years, this relationship is still poorly understood. Based on our current knowledge, reductions in deer density on moose range will likely have little effect on the population status of moose in Minnesota.

### INTRODUCTION

As early as 1912, a “moose sickness” was identified in Minnesota moose (Fenstermacher and Olson 1942). The disease was characterized by apparent blindness, lack of fear, aimless wandering, and ataxia (Karns 1967), which either killed moose directly or predisposed them to other causes of mortality. Histopathological analysis of diseased animals revealed irreversible damage to the central nervous system (Kurtz et al. 1966). Although moose sickness was associated with the presence of white-tailed deer as early as the late 1950s (Benson 1958), it was the experimental work by Anderson (1964), who demonstrated that the nematode lungworm (*Parelaphostrongylus tenuis*) caused moose sickness. Since this discovery, several hypotheses have been proposed regarding the relationship between deer, moose, and *P. tenuis*. It is the objective of this review to examine these hypotheses and subsequent research that either corroborate or refute them.

### LITERATURE REVIEW

The life cycle of *P. tenuis* normally incorporates a definitive host, white-tailed deer, and an intermediate host, which includes several species of gastropods (slugs and snails). Once a deer is infected, *P. tenuis* larvae develop into adults and live in association with nervous tissue in the spinal cord and in the subdural spaces and venous sinuses of the cranium (Lankester and Samuel 1998). After a complex journey through the deer’s body, first stage larvae are shed in the mucosal coating on feces and may survive as long as 10 months outside the host (Lankester and Anderson 1968). Gastropods that live in the litter on the forest floor crawl over the deer feces and become infected with the first stage larvae. Within the gastropod, the larvae ultimately molt into 3<sup>rd</sup> stage larvae that are infective to cervids if accidentally ingested (Lankester

and Samuel 1998). White-tailed deer apparently do not succumb to the neurologic disorders caused by *P. tenuis* in other cervids (Alibasogulu et al. 1961, Anderson 1963).

Shortly after Anderson identified *P. tenuis* as the cause of moose sickness, several authors hypothesized that *P. tenuis* was a major cause of mortality in moose and responsible for historic declines in moose populations (Karns 1967, Telfer 1967). By the mid-1970s, there was general agreement on this supposition, especially in areas of high deer densities in eastern North America (Gilbert 1974, Prescott 1974, Kearney and Gilbert 1976, Lankester 1987). In the 1980s, however, some scientists began to question this conjecture based on the low number of moose deaths attributed to “moose sickness” at a time when both moose and deer populations were increasing on shared range (Brown 1983, Upshall et al. 1987, Lenarz and Kerr unpubl.). Subsequently, Nudds (1990) questioned the hypothesis that *P. tenuis* was a major cause of mortality in moose and suggested that circular logic was used in making this inference. After reviewing all available data from Maine, Minnesota, New Brunswick, and Nova Scotia, Whitlaw and Lankester (1994a) indicated that the historical information available did not corroborate the hypothesis that *P. tenuis* had caused declines in moose populations.

Recent research in Minnesota also suggests that while present, *P. tenuis* is not a major cause of mortality. In northwestern Minnesota, for example, mortality of only 5% of radiocollared moose was attributed to *P. tenuis* (Murray et al. 2006). In northeastern Minnesota, 17% (18/108) of moose had positive titers for *P. tenuis* (Lenarz et al., unpublished data). Assuming that this parasite was responsible for the subsequent death of moose testing positive (except for 1 capture mortality and 1 hunter kill), annual cause specific mortality from *P. tenuis* averaged 4% (0 to 10%) and represented an average of 19% (0 to 32%) of the total mortality the population experienced each year (Lenarz et al., unpublished data). Considering the relatively low proportion of moose mortality attributed to *P. tenuis* in both northeastern and northwestern Minnesota, it is questionable whether this parasite represents a major threat to the moose populations.

Early researchers also suggested that the infection rate in moose increased as a direct response to increasing deer density (Anderson 1965, Karns 1967, Kelsall and Prescott 1971, Gilbert 1974). These early researchers reasoned that as deer density increased, more deer were infected, and more larvae would be shed. As a consequence of the increased number of larvae, more gastropods would be infected, and the probability that moose would consume an infected gastropod and die would increase. Subsequent research, however, has indicated that the relationships between deer density, *P. tenuis* infection rates, and moose mortality are complex and poorly understood (Anderson and Prestwood 1981, Whitlaw and Lankester 1994a, b; Lankester and Samuel 1998).

The hypothesis that more larvae are shed as deer density increases assumes that the prevalence of *P. tenuis* in deer is constant or increases as deer numbers increase. Based on meager evidence, Karns (1967) and Behrend and Witter (1968) suggested that the prevalence of *P. tenuis* increased as deer numbers increased. Gilbert (1973), however, found a lower prevalence at higher deer density after comparing 2 areas in Maine. Thomas and Dodds (1988) found no relationship between deer infection rates and deer density (2 levels) or moose density (3 levels) in Nova Scotia. Based on deer sampled from 17 Deer Management Districts, Bogaczyk et al. (1993) found that neither prevalence nor intensity of infection in white-tailed deer was associated with deer density over a range of 1.4 to 5.8 deer/km<sup>2</sup>. Hence, there are few data to suggest a relationship between prevalence and deer density. It is logical to

assume, however, that more deer will deposit more feces on the landscape and unless prevalence declines in response to deer density, there will be more infected feces.

Even if the density of infected feces is high, it doesn't imply that higher numbers of gastropods are infected. Lankester and Peterson (1996) found a prevalence rate of only 0.16% (7 out of 4,401) in a deeryard that seasonally supported 50 deer/km<sup>2</sup>. Other surveys in Minnesota and Ontario have generally found a prevalence rate less than 1% (Lankester 1967, Kearney and Gilbert 1978, Pitt and Jordan 1995, Lankester and Peterson 1996). Research that reported both infection rates of gastropods and local deer density is limited (Lankester and Anderson 1968, Maze and Johnstone 1986, Platt 1989, Lankester and Peterson 1989, Pitt and Jordan 1994). Pooling these data, there was no correlation ( $r = 0.09$ ,  $P = 0.86$ ) between prevalence and deer density. The infection rate of gastropods is likely dependent on the density, residence time, and defecation rates of infected deer; the survivorship of first-stage larvae on the feces or in the soil; and the abundance of and mobility of suitable gastropods (Lankester and Peterson 1996). Even in a situation with 120 deer living year around on a 1.3 km<sup>2</sup> island (240 deer/mi<sup>2</sup>), the prevalence of *P. tenuis* in gastropods was only 4.2% (Lankester and Anderson 1968).

Considering the extremely low infection rate of gastropods, it is unclear how large numbers of deer become infected. Based on the prevalence documented by Lankester and Peterson (1996) in northeastern Minnesota and assuming that a deer or moose could be infected by consuming a single infected gastropod, each deer or moose on summer range would need to consume 2,500 gastropods to become infected. Anderson and Prestwood (1981) proposed that infected gastropods might live in small concentrations that are not adequately sampled by researchers but encountered by foraging cervids. In a study of gastropod climbing behavior, McCoy and Nudds (1997) found that species were highly variable in the degree that they climbed; some species climbed infrequently while other were primarily arboreal. They suggested that data from studies which restricted sampling to the use of damp cardboard (e.g. Gleich and Gilbert 1976, Kearney and Gilbert 1978, Upshall et al. 1986, Lankester and Peterson 1996), resulted in estimates biased to the less arboreal species, which are more likely to encounter *P. tenuis* larvae. If correct, prevalence rates in gastropods would be even lower than currently estimated.

Finally, if moose mortality was a simple function of deer density, there should be an inverse correlation between changes in deer and moose density. In the 1980s and 1990s, several authors documented simultaneous increases in sympatric moose and deer populations in some eastern states and provinces (Clark and Boyer 1986, Upshall et al. 1987, Thomas and Dodds 1988, Boer 1992, Bogaczyk et al. 1993). Several hypotheses were proposed to explain this conundrum. Working in Maine, Clark and Bowyer (1986) found a high prevalence of *P. tenuis* larvae in moose feces and suggested that co-evolution favoring a reduction in the debilitating effect of *P. tenuis* upon moose may have occurred. McCullough and Pollard (1993), however, suggested that faulty lab procedures might have been responsible for the high prevalence of *P. tenuis* in moose feces found by Clark and Bowyer (1986). Upshall et al. (1987) found no larvae in New Brunswick moose and suggested that moose were feeding in different areas than deer, an argument first proposed by Telfer (1967) and subsequently challenged by Nudds (1990).

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## ANDEAN BEAR DENSITY AND ABUNDANCE ESTIMATES — HOW RELIABLE AND USEFUL ARE THEY? <sup>1</sup>

David L. Garshelis

### ABSTRACT

Estimates of bear abundance and density are difficult to obtain, for a number of reasons. Reliable estimates have been obtained for several populations of American black bears (*Ursus americanus*), brown bears (*U. arctos*), and polar bears (*U. maritimus*), and one population of sloth bears (*Melursus ursinus*). Efforts have been made to obtain estimates of the other bear species, but often with unresolved problems that are disregarded. Very few attempts have been made to estimate numbers and densities of Andean bears (*Tremarctos ornatus*). One very crude rangewide estimate, from 10 years ago, was made by applying the average density of American black bears (obtained from a compilation of many studies, mostly in hunted populations) to the range area of Andean bears. The resulting estimate of ~20,000 bears has been widely cited, without understanding how the value was derived. A more recent estimate (2003), based on the extent of genetic heterozygosity, and assuming that Andean bear numbers have remained stable for thousands of years, yielded a wide span of values (24,000–90,000); nevertheless, the closeness of the low value to the former estimate based on black bears seemed to strengthen the view that there are at least 20,000 Andean bears across the range. Three estimates in smaller study sites produced widely varying and contradictory estimates. At one site in Bolivia, a density estimate based on photographs of 3 bears at remote camera traps (4.4–6 bears/100km<sup>2</sup>) was one-half to one-third that of an earlier study in the same site, based on home ranges of 2 radiocollared bears (~12 bears/100 km<sup>2</sup>). Neither is reliable because of the small sample sizes, but both suggest that the original rangewide density, derived from black bears (25 bears/100 km<sup>2</sup>) could be an overestimate. Likewise, a DNA mark-recapture study in Ecuador also obtained a low density estimate (3–7 bears /100 km<sup>2</sup>). In this case, sample size was adequate, but there were strong indications that closure was violated, especially for males (twice as many males as females were detected), indicating that the density could be even lower. These issues are not simply incorporated into confidence intervals: CIs only include error due to sampling, not study design flaws; CIs around biased estimates may not include the true population number. I suggest that more work of a rigorous nature needs to be conducted if density estimates are to be useful in conservation and management.

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## ECOLOGY AND POPULATION DYNAMICS OF BLACK BEARS IN MINNESOTA

David L. Garshelis and Karen V. Noyce

### SUMMARY OF FINDINGS

During April 2008–March 2009, we monitored 40 radiocollared black bears (*Ursus americanus*) at 4 study sites representing contrasting portions of the bear's geographic range in Minnesota: Voyageurs National Park (VNP, northern extreme), Chippewa National Forest (CNF; central), Camp Ripley (southern), and a new site at the northwestern (NW) edge of the range. Hunting was the primary source (78%) of mortality in all areas, even though hunters were asked not to shoot radiocollared bears and bears cannot be legally hunted in 2 of the areas (but can be hunted when they wander outside). Reproduction was highest at the fringe of the bear range, at Camp Ripley (near the southern edge) and the NW site (western edge), due largely to an abundance of oaks and hazelnut in these areas, as well as agricultural crops consumed by NW and some Camp Ripley bears in the late summer–fall. Data from GPS-radiocollars indicated that males had home ranges of ~1000 km<sup>2</sup> (~400 mi<sup>2</sup>), 18x larger than those of females, and the largest yet measured for black bears across North America. In this highly-fragmented landscape, males travelled large distances, particularly in late summer to locate oaks and agricultural crops; they spent a third of their time in croplands during September. In contrast, only 2 of 7 GPS-collared females visited croplands. Although 15 bears had GPS-radiocollars in 2008, 7 of these failed, so both the bears and data were lost. A different brand of GPS-collar will be used in the coming year.

### INTRODUCTION

A lack of knowledge about black bear ecology and effects of harvest on bear populations spurred the initiation of a long-term telemetry-based research project on this species by the Minnesota Department of Natural Resources (MNDNR) 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 in sizes, growth rates, and productivity of bears across the state, apparently related to varying food supplies, we started other satellite bear projects in different study sites. Each of these began as a graduate student project, supported in part by the MNDNR. 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-wooded forests dominated by aspen (*Populus tremuloides*, *P. grandidentata*) of varying ages. Camp Ripley is unharmed, but bears may be killed by hunters when they range outside, which they often do in the fall, as the reserve is only 6–10 km wide. Oaks (*Quercus* sp.) are far more plentiful here than in the 2 study sites farther north, and agricultural fields (corn) border the reserve. VNP, being a national park, is also unharmed, but again bears may be hunted when they range outside. Soils are shallow and rocky in this area, and foods are generally less plentiful than the other sites.

In 2007 we initiated work in another study at the northwestern edge of the Minnesota bear range (henceforth NW). This area differs from the other 3 areas in a number of respects: (1) it is largely agricultural (including some croplands, like corn, oats, and sunflowers, that bears consume), interspersed with MNDNR Wildlife Management Areas, a National Wildlife Refuge, and small private woodlots; (2) the bear range in this area appears to be expanding and bear numbers have been increasing (most other parts of the bear range are stable or declining in bear numbers); and (3) hunting pressure in this area is unregulated (it is within the no-quota zone, so there is no restriction on numbers of hunting licenses, and each hunter is allowed to kill 2 bears).

## OBJECTIVES

1. Monitor temporal and spatial variation in cub production and survival;
2. Monitor rates and sources of mortality;
3. Compare body condition indices across sites and years (not covered in this report);
4. Assess habitat requirements for bears in an agricultural fringe area; and
5. Predict range expansion of bears in northwestern Minnesota.

## METHODS

We attached radiocollars with breakaway and/or expandable devices to bears either when they were captured during the summer or when they were handled as yearlings in the den of their radiocollared mother. We trapped bears this year only in the NW study site, using barrel traps baited with raw bacon, and anesthetized them with ketamine-xylazine. In this area, we used principally GPS-collars, programmed to collect locations every 2–4 hours. These data will be used to assess fine-scale movements and habitat use in this highly-fragmented landscape.

During December–March, we visited all radio-instrumented bears once or twice at their den site. We immobilized bears in dens 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 (removing GPS-collars for downloading data), attaching a first collar on yearlings, measuring, weighing, and obtaining blood and hair samples. We also measured bioelectrical impedance (to calculate percent body fat) and vital rates of all immobilized bears. Additionally, collaborators from the University of Minnesota (Dr. Paul Iaizzo) and Medtronic (Dr. Tim Laske) measured heart condition with a 12-lead EKG and ultrasound on a select sample of bears in early and late winter. Bears were returned to their dens after processing.

We assessed reproduction by observing cubs in dens of radiocollared mothers. We sexed and weighed cubs without drugging them. We evaluated cub mortality by examining dens of radiocollared 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 periodically through the summer. 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. During the hunting season, hunters reported collared bears that they killed.

We plotted GPS locations downloaded from collars on bears in the NW study site. We calculated home range areas as 100% minimum convex polygons (MCP) and 100% fixed kernels. We recognize that there is a recent trend toward using 95% MCPs or kernels, but in examining the data, we found that these estimators excluded a large number of points, which in this fragmented landscape accounted for a sizeable travel area.

We used a GIS overlay (Minnesota Land Cover – NLCD 2001) to categorize the covertypes of GPS locations, and then grouped these into 6 broad categories. We calculated percent use of these types by month for each bear, then obtained monthly averages among bears. We examined temporal trends in use of forests (deciduous, evergreen, and mixed types combined), wetlands (woody wetlands and herbaceous wetlands combined), and agricultural areas. Bears rarely used the other 3 categories, so they are not examined further here.

We conducted food sampling on plots in various woodlands in the NW study site, representing all the principle forest types in that area. Experience in our previous studies indicated that fruit production is often high at the forest edge, so we situated plots such that we sampled both the edge and interior of the woodlot. We sampled 12 circular plots, each 3-m radius, per stand. Within each plot, we separately estimated the percent areal coverage and productivity of all principal fruiting species that bears consume. We visually rated fruit production on a 0–4 scale, with 0 = no fruit, 1 = below average fruit production, 2 = average fruit production, 3 = above average fruit production, and 4 = bumper crop.

## RESULTS AND DISCUSSION

### Radio collaring and Monitoring

Since 1981 we have handled >800 individual bears and radiocollared >500. As of April 2008, the start of the current year's work, we were monitoring 28 collared bears: 5 in the CNF, 10 at Camp Ripley, 4 in VNP, and 9 in the new NW study site. Two bears at Camp Ripley and 7 in the NW had GPS-collars. The small sample sizes in the CNF and VNP were due to high harvests of radiocollared bears during the past several years.

We captured more bears in the NW study site during the month of June, concentrating in the far northwestern corner of the state. We captured 12 new bears (8 males, 4 females), and collared them, 6 with VHF radiocollars and 6 with GPS-collars.

During fall telemetry flights in the NW, 7 GPS-collars (all manufactured by ATS, Isanti, MN) could not be located; we assume that they failed.

### Mortality

Legal hunting has been the predominant cause of mortality among radiocollared bears from all study sites; 78% of mortalities that we observed were due, or likely due to hunting (Table 1). In earlier years of this study, 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 and habitat use 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. However, the mortality rate on collared bears has remained high because many hunters indicated that they could not see the collars in dim light conditions. Some hunters, though, saw collars on bears photographed by camera traps at their bait sites, and thereby avoided killing the bear.

This year (September–October 2008), hunters killed 1 of 5 CNF bears (unfortunately, a 23-year old individual whom we had monitored through 10 reproductive cycles and had hoped to track into her reproductive senescence) and 1 of 4 VNP bears (killed outside the park), but no bears at Camp Ripley and only 1 of 21 in the NW (another was killed after it lost its collar). The lower rate of loss in the NW site may have been due to more widespread publicity of this project in that area. It is also possible that some of the GPS-collared bears that could not be located during aerial searches of this area were killed by hunters, although normally hunters have returned collars, and in the rare cases where they have not, we have been able to locate collars left in the woods.

One other mortality occurred this year, a Camp Ripley bear that was hit by a vehicle outside the Camp. Camp Ripley has had the highest rate of vehicle-caused bear mortality, with 8 of 27 deaths (30%) attributable to this cause; in CNF, <5% of deaths were due to collisions with vehicles (Table 1). Statewide, only 27 car-killed bears were reported in 2008, but this is certainly an underestimate of the number actually killed.

Although nuisance kills have been the second-most common cause of bear mortality overall, across all study areas and years (Table 1), few collared bears have been killed as nuisances in recent years (most of the 25 nuisance-related mortalities among collared bears occurred in the 1980s). This corresponds with statewide records, which indicate that <30 bears were killed as nuisances each year for the past 10 years (vs. 100–400 killed annually during the 1980s and early 1990s).

Natural mortality is a relatively minor cause of death among Minnesota bears >1 year old. Natural mortalities were most common in VNP (Table 1). Now, however, with a small remaining sample of bears in this area, natural mortalities will be harder to detect. However, all of the remaining bears are unusually small females (2 4-year olds weighed only 69 and 99 pounds in March, 2009), so may be susceptible to undernutrition or cannibalism.

## Reproduction

Of 13 mature female bears checked in dens during late winter, 2009, 5 (38%) had cubs and 8 had yearlings. However, 1 mother with a yearling in the NW remained awake much of the winter, and separated from her yearling during January or February. Additionally, a 35-year-old has been post-senescent since 1999, when she was 25 years old.

Bears at Camp Ripley grow faster and thus have an earlier age of first reproduction than at CNF and VNP. Two years of data from the NW study site suggest that the bears there are also large, with early maturity (e.g., 1 male yearling weighed 131 lbs in the den in early March, the heaviest of this study). Both Camp Ripley and the NW site are at the fringe of Minnesota's bear range, where acorns, hazelnuts (*Corylus americana*, *C. rostrata*), and agricultural crops are plentiful, accounting for the high growth rates of these bears.

At Camp Ripley, 6 of 7 female bears produced their first litter at age 3; however, only half of these 6 litters survived the first year. Among 4–6 year-old females, the reproductive rate (cubs born/female) was nearly twice as high at Camp Ripley as in VNP (where no bears produced cubs at 4 years old, including 2 monitored this year); the reproductive rate of 4–6 year-olds was intermediate at CNF (Table 2). This gradient was also apparent in the reproductive rates of older bears, due to fewer missed reproductive opportunities in Camp Ripley versus 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. In both Camp Ripley and the NW, the proportion of adult females with cubs exceeded 50% due to an artifact of sampling (Table 2). This proportion was lowest in VNP and intermediate in CNF. Reproductive rates were also most variable, year-to-year, in VNP, and least so at Camp Ripley (Figure 1).

Mean litter size was not appreciably different among sites (2.3–2.6 cubs/litter, Tables 3–5). Data from the NW site produced a higher average litter size, but the sample size is still too small to make a reliable estimate in this area.

Average sex ratio of cubs shortly after birth was slightly, but consistently male-biased (pooled average across all areas = 52% male,  $n = 594$  cubs examined). Observed year-to-year variation in cub sex ratios (Tables 3–6) was likely attributable to sampling error, although it is possible that some real year-to-year variation may occur as a result of varying food conditions.

Cub mortality averaged 18–20% in CNF and Camp Ripley, and somewhat higher in VNP (Tables 3–5). Assessment of cub mortality in the NW was hampered by the failure of several radiocollars on adult females with cubs. Across all areas, the mortality rate of male cubs was significantly higher than (nearly 2x) that of females ( $\chi^2 = 10.3$ ,  $P = 0.001$ ), however, the predominant cause of cub mortality in Minnesota is not known.

Cub production (Figure 1) and cub mortality did not show either an upward or downward trend during our 28 years of monitoring. However, statewide bear harvests have contained an increasing proportion of young bears (Figure 2), suggesting a changing statewide age structure, likely due to increased hunting pressure.

## Movements and Habitat Use of NW Bears

Data from GPS-collars in the NW study site indicated that home ranges were especially large. The average male home range in this study was 900–1100 km<sup>2</sup> (350–420 mi<sup>2</sup>; depending on method of estimation, Table 7). We compiled results from 32 other studies of black bears across North America, which reported male home ranges of 5–465 km<sup>2</sup> (2–180 mi<sup>2</sup>; median = 108 km<sup>2</sup> = 42 mi<sup>2</sup>). Female home ranges in this study averaged 60 km<sup>2</sup> (23 mi<sup>2</sup>) based on MCPs. These were well above average from across North America (median = 25 km<sup>2</sup> = 10 mi<sup>2</sup>), but were within the range observed in other studies (2–295 km<sup>2</sup> = 1–114 mi<sup>2</sup>). The ratio of M:F home range sizes in our study (18:1 for MCP) is the largest yet reported.

These large home ranges, especially for males, are related to the fragmented habitat. Bears in this area must travel more because of the gaps between patches of suitable habitat. We purposefully included the gaps of cultivated land between forested patches as part of the home range area, as they are clearly relevant to the space use of these bears. Moreover, although kernel

estimators are now more commonly used than MCP, we feel that the MCP better represented the scope and scale of the land area necessary to support these bears, if not the area that they actually used. The normal concept of a home range (the area routinely used) does not apply very well in a fragmented, patchy landscape like this, because although large areas between the main habitat patches are rarely used, they stretch out the size of the home range. If only the small patches of actual use were summed (as tends to occur with kernel estimators), it would appear as though bears existed within much smaller areas (although 100% kernels were larger than MCPs for females, Table 7).

Monthly home ranges of males varied seasonally, and were not in accordance with temporal trends previously reported for bears in continuous forested habitats, including our other 3 study sites. The smallest monthly ranges were in June and July, during the breeding season, when males appeared to home in on several estrus females. Their home ranges during these months were 4–7x larger than those of females (Figure 3). Normally, in continuous habitat, males enlarged their ranges during breeding, to encompass more females. Afterward, in the late summer, males in this study tended to go on long excursions, attracted to scattered natural oak stands as well as agricultural crops; thus, their August and September home ranges were greatly expanded. Females generally did not expand their home ranges in late summer (Figure 3). The ratio of M:F monthly home range size peaked in September at 33:1 (excluding November, when most females had denned).

Male habitat use reflected the seasonal expansion of their home ranges. In September, 34% of their locations, on average, were in agricultural lands, presumably corn, oats, or sunflowers (Figure 4). All of the GPS-collared males used some agricultural crops, and they often travelled significant distances to find a cropfield to settle into (Figure 5). Conversely, only 2 of 4 solitary females and 0 of 3 females with cubs used agricultural cropfields in late summer and fall, and those that did so left these fields earlier than the males (Figure 4). It appeared that females only used cropfields that were adjacent to their home range. Most of the females had some agricultural lands adjacent to their home range, but the use of these lands was minimal (Figure 6), so we presume that they did not contain edible crops (we cannot ascertain this from the GIS overlays, but will determine what was grown there from county records and follow-up interviews with farmers).

Both males and females showed a trend of decreasing use of the “forest” category of land cover, and increasing use of “wetlands”, from May–November (Figure 4). We cannot yet speculate on the cause of this trend. We plan to investigate these wetland areas more thoroughly to see what sorts of habitats they really include, and from this we hope to be able to explain their increasing use through the year. This trend was opposite what we expected; in our other study sites, bears used wetlands primarily during spring.

## Fruit Sampling

From July 15 to August 20, we sampled 65 stands for soft mast and hazelnut production in the NW study site, including: 36 stands that were predominantly aspen (but varying in age, silvacultural treatment, and ground moisture), 18 oak, 5 balsam poplar (*P. balsamifera*), and 6 lowland hardwood. We quantified abundance of fruit-producing plants and fruit production of these plants. We also picked representative fruit samples to convert fruit production ratings to biomass estimates. These data have not yet been analyzed, however, it was clear that fruit production in 2008 was much less than in 2007, when juneberry (*Amelanchier* sp.), chokecherry (*Prunus virginiana*), and both species of hazel were especially productive.

## Denning

Bears in the NW study site tended to den during November or even December, later by >1 month than CNF bears. Some individuals moved dens, either on their own or due to our disturbance.

Two denning bears (1 large male, 1 female with yearlings) were found by local people in the NW study site, so we collared them. The female had a large wound on her back, probably a non-

fatal injury from a bullet or arrow. The male had apparently been flooded in its den during a rare, heavy rain in mid-February, so had moved and denned on the ground in an open field (Figure 7). He remained there for ~6 weeks.

We did not actively monitor when collared bears arose from hibernation, but we checked some in NW and found that by April 1, 7 (4 males, 3 adult females) of 8 bears had vacated their dens. The ground was saturated and underground dens were filled with water (ice). The only collared bear still in its den at that time was a young female whose underground den was not flooded. We did not check the den of the only collared female in the NW with newborn cubs, but she had denned underground in a very low, wet area, so we suspect that she too would have been flooded out. In the CNF during the 1980s, bears typically stayed in dens until at least the first week of April, and those with cubs did not emerge until mid-April.

## **FUTURE DIRECTION**

We plan to continue monitoring bears on these 4 study sites, although sample sizes have been greatly diminished by the exceedingly high harvest of collared bears in the past few years. Our main emphasis in the next few years will be at the new study site in Northwestern Minnesota, where we replaced GPS-collars with those of a different manufacturer. Our goal there is to assess the factors that may limit range expansion, including highly fragmented forested habitat, availability of agricultural crops that bears can eat, and human-related mortality.

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Table 1. Causes of mortality of radiocollared black bears  $\geq 1$  year old from the Chippewa National Forest (CNF), Camp Ripley, Voyageurs National Park (VNP), and northwestern (NW) Minnesota, 1981–2008. 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	NW	All combined
Shot by hunter	221	11	15	4	251
Likely shot by hunter <sup>a</sup>	8	1	0	0	9
Shot as nuisance	22	2	1	0	25
Vehicle collision	12	8	1	1	22
Other human-caused death	9	0	0	0	9
Natural mortality	7	3	4	0	14
Died from unknown causes	3	2	0	0	5
<b>Total deaths</b>	<b>282</b>	<b>27</b>	<b>21</b>	<b>5</b>	<b>335</b>

<sup>a</sup> Lost track of during the hunting season.

Table 2. Reproductive rates (cubs/female), mean litter size, and proportion of females with cubs (in all cases, counting only litters in which at least 1 cub survived 1 year) in winter dens (March) in 4 study sites (ordered from lowest to highest reproductive output): VNP (1997–2009), CNF (1981–2009), Camp Ripley (1991–2009), NW (2008–2009) ( $n = 4+$  year-old female-years of observation). Data from the new study site in the northwest are still too sparse to separate by age categories, and the overall values presented here are biased high by the loss of some bears due to radiocollar failure (see text).

Age of female	VNP ( $n = 59$ )			CNF ( $n = 407$ )			Camp Ripley ( $n = 51$ )			NW ( $n = 11$ )		
	Repro rate	Litter size	Prop w/ cubs	Repro rate	Litter size	Prop w/ cubs	Repro rate	Litter size	Prop w/ cubs	Repro rate	Litter size	Prop w/ cubs
4–6 yrs	0.53	2.0	0.26	0.84	2.3	0.37	1.04	2.2	0.48			
7–25 yrs	1.15	2.7	0.44	1.33	2.8	0.48	1.58	2.7	0.58			
4–25 yrs	0.93	2.5	0.37	1.15	2.6	0.43	1.29	2.4	0.53	1.75	3.0	0.58

Table 3. Black bear cubs examined in dens of radiocollared mothers in or near the Chippewa National Forest during March, 1982–2009. High hunting mortality of radiocollared bears has reduced the sample size in recent years to the extent that the data are no longer suitable for monitoring.

Year	Litters checked	No. of cubs	Mean cubs/litter	% Male cubs	Mortality after 1 yr <sup>a</sup>
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% <sup>b</sup>
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%	17%
2008	1	3	3.0	100%	33%
2009	1	3	3.0	33%	—
<b>Overall</b>	<b>174</b>	<b>455</b>	<b>2.6</b>	<b>52%</b>	<b>18%</b>

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

<sup>b</sup> 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, 1992–2009.

Year	Litters checked	No. of cubs	Mean cubs/litter	% Male cubs	Mortality after 1 yr <sup>a</sup>
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%	0%
2008	2	5	2.5	60%	0%
2009	3	7	2.3	29%	—
<b>Overall</b>	<b>28</b>	<b>66</b>	<b>2.4</b>	<b>50%</b>	<b>20%</b>

<sup>a</sup> 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. All adult collared females were killed by hunters in fall 2007, so there are no reproductive data for 2008-2009.

Year	Litters checked	No. of cubs	Mean cubs/litter	% Male cubs	Mortality after 1 yr <sup>a</sup>
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%	—
<b>Overall</b>	<b>24</b>	<b>54</b>	<b>2.3</b>	<b>52%</b>	<b>28%</b>

<sup>a</sup> 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.

Table 6. Black bear cubs examined in dens of radiocollared mothers in Northwestern Minnesota during March, 2007–2009. Loss of 1 large litter in 2007 skews the sample, so overall cub mortality is not calculated (value for 2008 may be more typical).

Year	Litters checked	No. of cubs	Mean cubs/litter	% Male cubs	Mortality after 1 yr <sup>a</sup>
2007	2	6	3.0	33%	100% <sup>b</sup>
2008	5	15	3.0	67%	22%
2009	1	3	3.0	33%	—
<b>Overall</b>	<b>8</b>	<b>24</b>	<b>3.0</b>	<b>52%</b>	<b>—</b>

<sup>a</sup> Cubs that were absent from their mother's den as yearlings were considered dead.

<sup>b</sup> Only one 5-cub litter was monitored, and all the cubs died (mother produced a litter of 4 cubs the next year).

Table 7. Estimates of annual home ranges of GPS-collared bears in Northwestern Minnesota, 2007 and 2008, based on 100% minimum convex polygons (MCP) and fixed kernels.

Sex	N	MCP (km <sup>2</sup> )			Fixed kernel (km <sup>2</sup> )		
		Mean	Min	Max	Mean	Min	Max
Males	6	1,081	100	2,976	893	160	1,837
Females	7	60	35	107	151	114	208

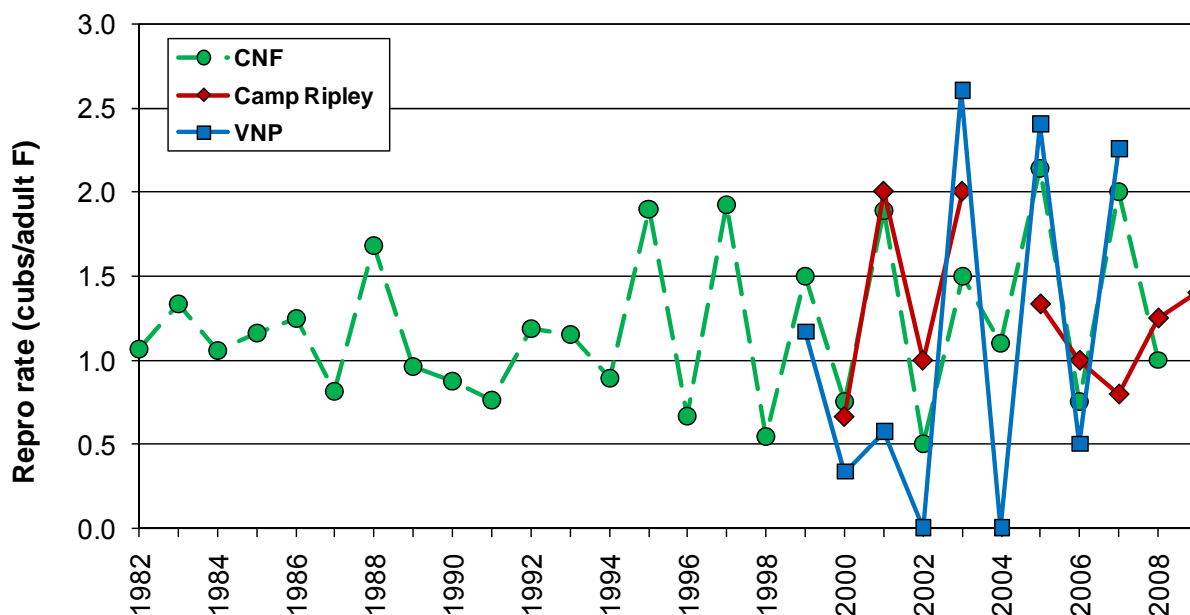


Figure 1. Reproductive rate (measured as [total cubs produced]/[no. of 4+-year-old females monitored]) in each of 3 study sites in Minnesota, 1982–2009. Gaps in data indicate inadequate sample size.

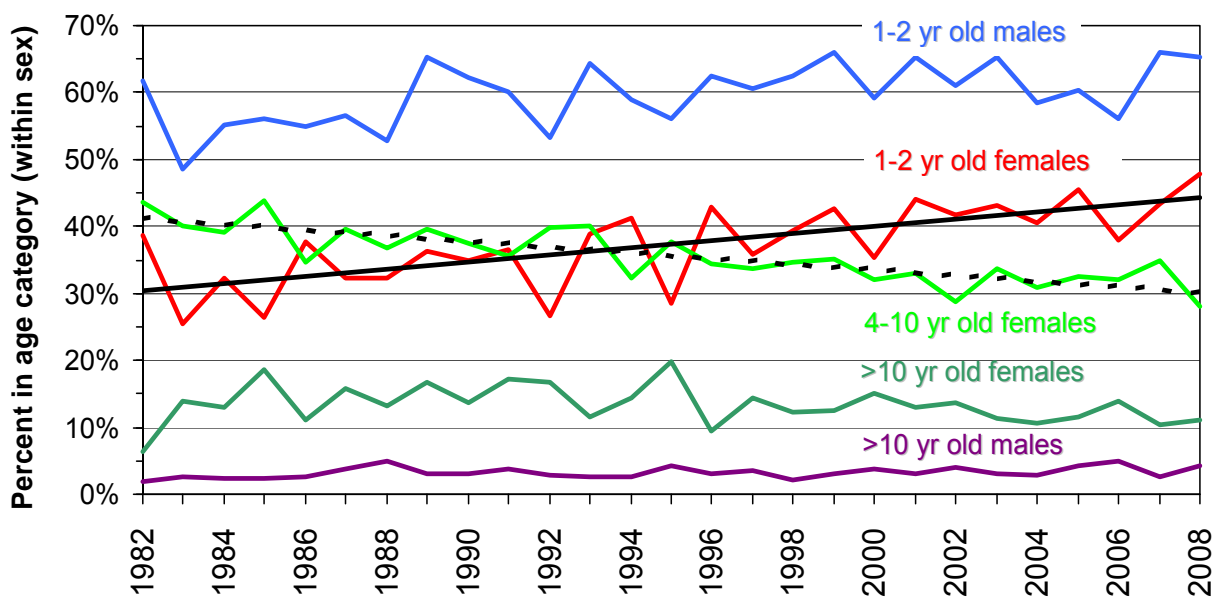


Figure 2. Proportion of harvested bears of each sex in indicated age groupings (3-year-olds excluded for clarity). Increasing trends for 1–2 year-old males and females, and declining trend for 4–10 year-old females are statistically significant ( $r^2 = 0.26$ ,  $P = 0.006$ ;  $r^2 = 0.48$ ,  $P = 0.0001$ ;  $r^2 = 0.66$ ,  $P < 0.0001$ ; respectively).

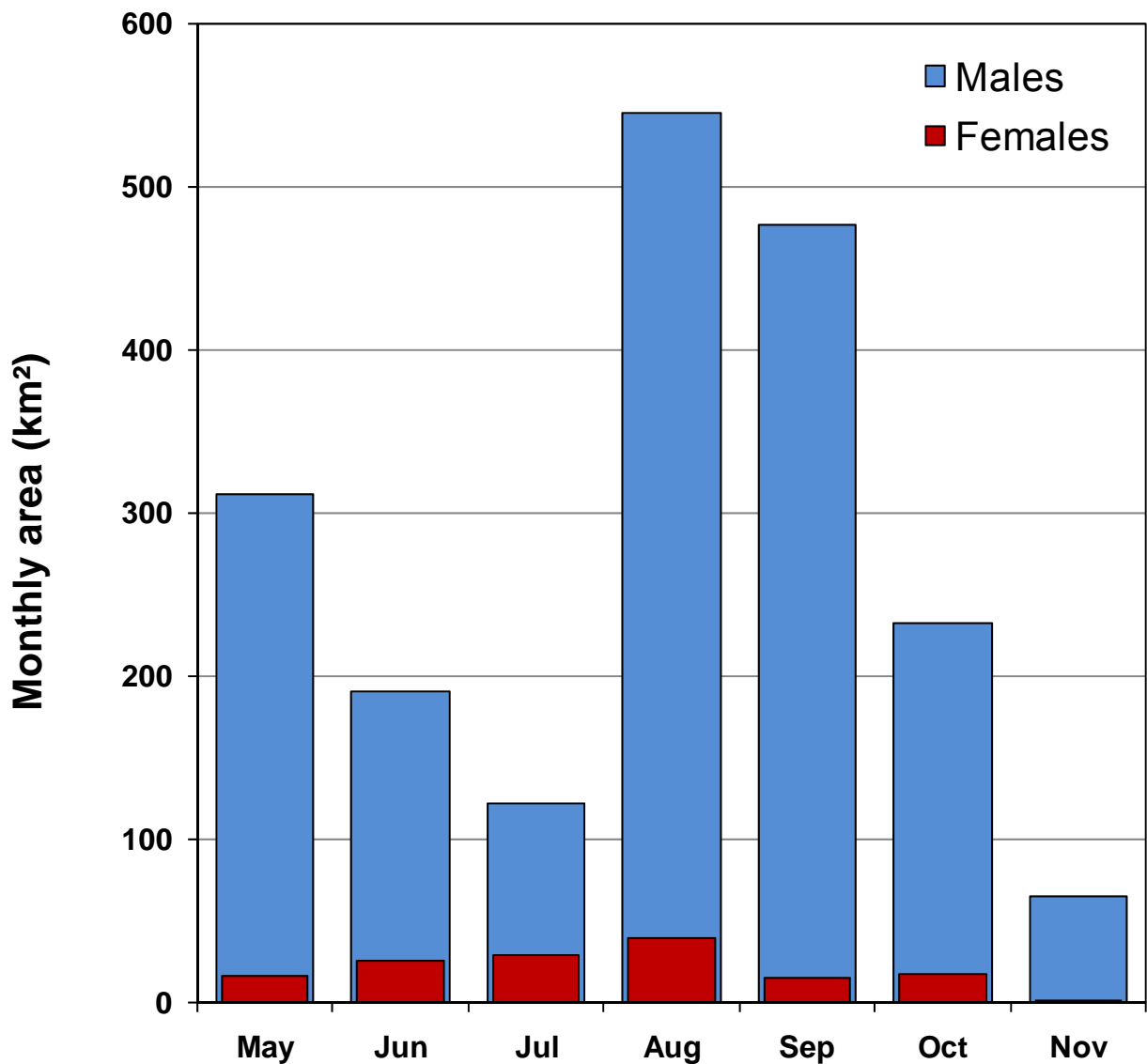


Figure 3. Mean monthly home ranges (100% MCP) of male and female black bears in Northwestern Minnesota, 2007 and 2008. Sample sizes were 4–6 males and 3–5 females each month. Females denned in November and males denned in November or December.

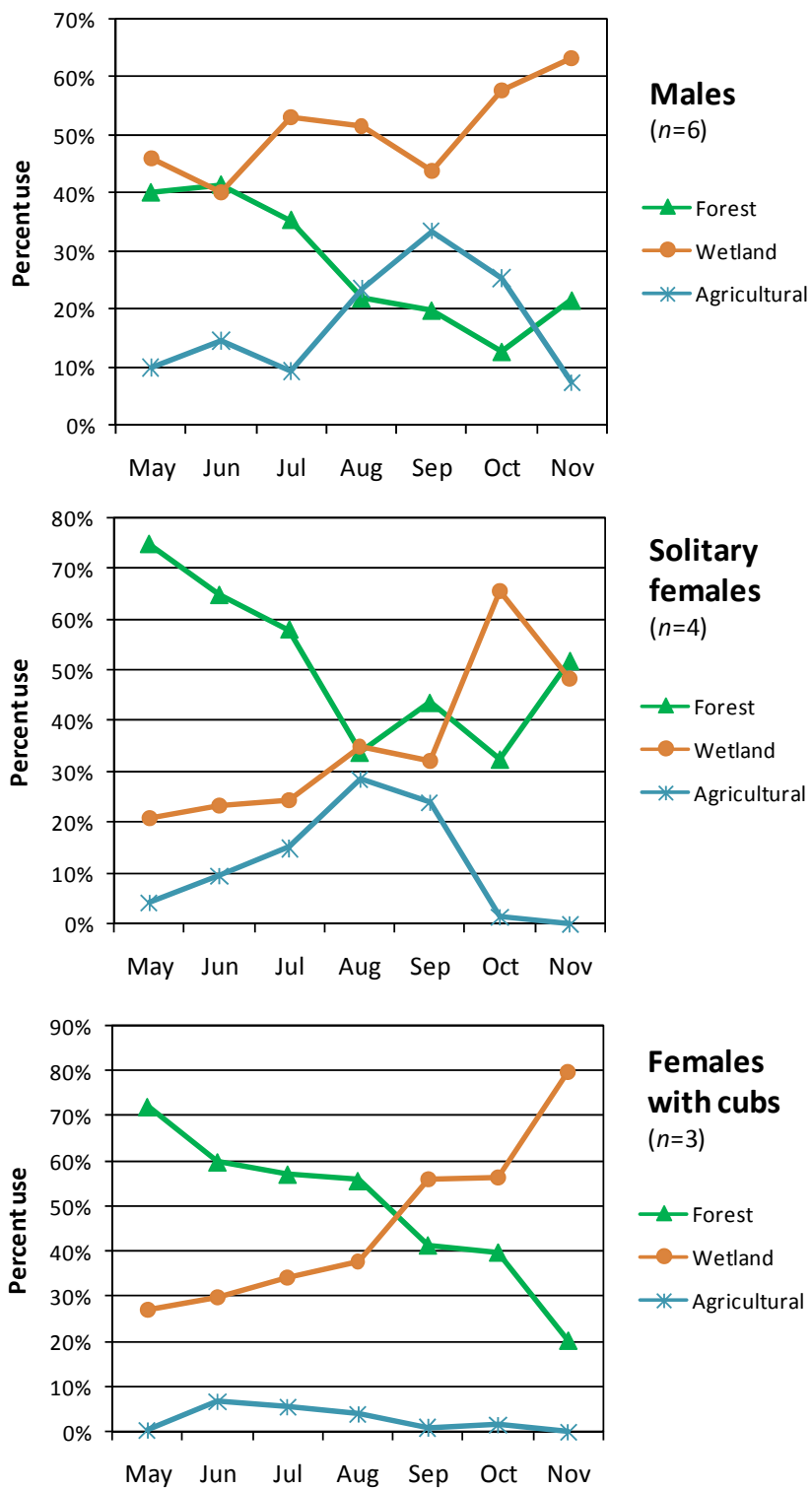


Figure 4. Trends in habitat use of 3 classes of black bears in Northwestern Minnesota, based on locations from GPS-radiocollars. Wetland areas include woody (shrub or forested) wetlands.

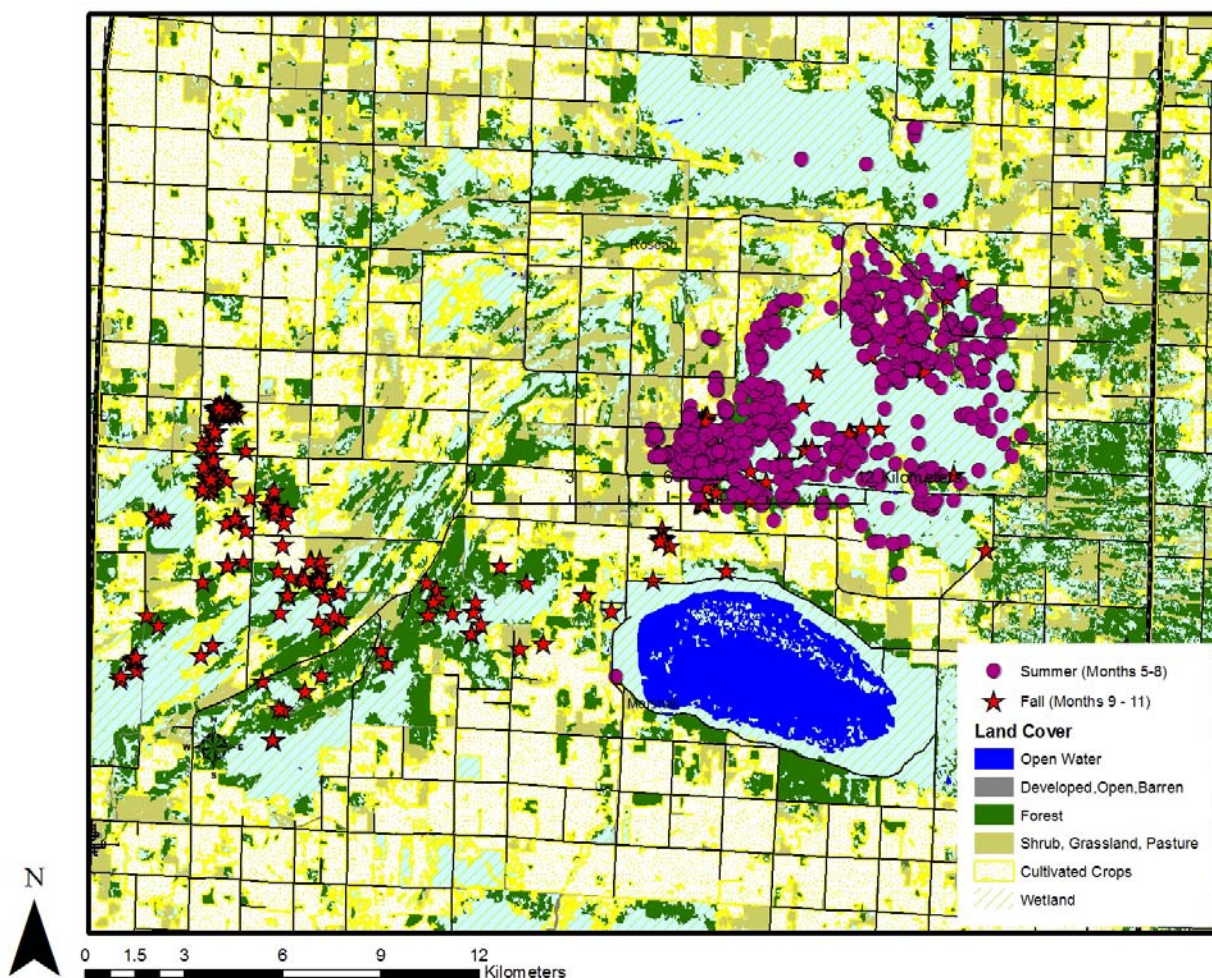


Figure 5. Home range of a GPS-collared male black bear, showing a distinct movement from the summer to fall range, 2007. The area that this bear moved to in the fall overlapped the area used by the female bear depicted in Figure 6. However, the male extensively used the agricultural fields to the north of the female's range. The northwestern-most cluster of stars represent locations during September and October, when it routinely visited a few cropfields.



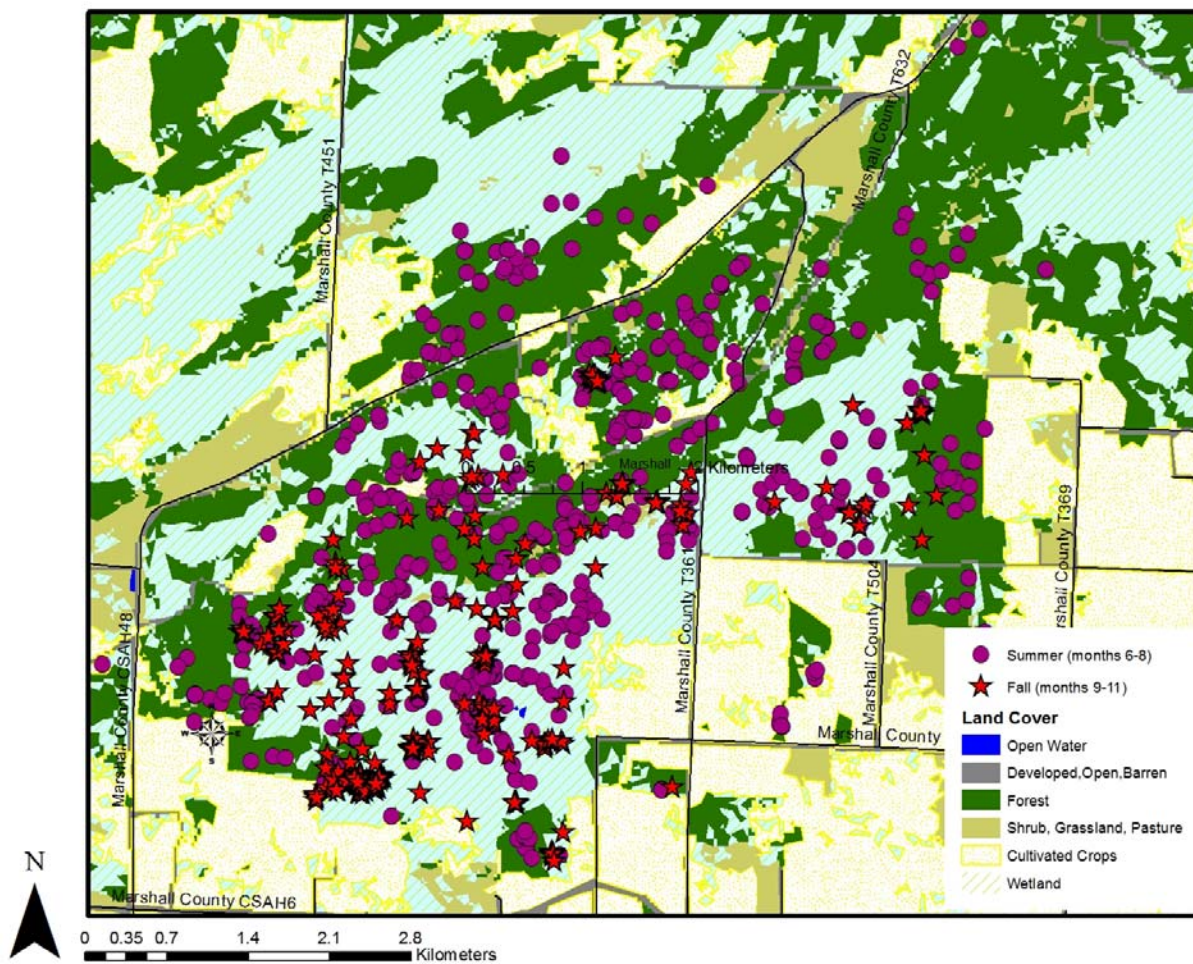


Figure 6. Home range of a GPS-collared female black bear on Thief Lake Wildlife Management Area, Northwestern Minnesota, showing areas of use during summer and fall months, 2007. This bear did not travel during fall, and although agricultural lands were adjacent to its range, it did not use them (presumably because they did not contain edible crops).





Figure 7. Bear denned in a CRP (Conservation Reserve Program) field in Northwestern Minnesota, 2009. Apparently this bear (weighing 347 lbs in early March) was flooded from its den by a heavy February rain. It remained denned in the field until March 27.

## PRELIMINARY RESULTS OF TETRACYCLINE MARK-RECAPTURE ESTIMATION OF MINNESOTA'S BLACK BEAR POPULATION, 2008

Karen V. Noyce and David L. Garshelis

### SUMMARY OF FINDINGS

During summer, 2008, we conducted tetracycline mark-recapture sampling of black bears (*Ursus americanus*) throughout their range in Minnesota. This was the 4<sup>th</sup> such survey in 17 years, providing independent estimates of Minnesota's bear population (bears  $\geq 1$  year old). These estimates are used to track population trends and aid in formulating harvest quotas. Bears were marked when they ingested baits of tetracycline-laced bacon, placed at approximately 4.8-km intervals across the bear range. During June–July, >3500 baits were set, marking an estimated 473–489 bears. This was lower than expected, in part due to use of a box to shield the bait from competing animals, high natural food abundance, and reduced bear abundance. During September–October, successful hunters submitted rib and tooth samples from harvested bears. These were screened microscopically for tetracycline, which fluoresces yellow under ultraviolet (UV) light. Of 1,498 samples examined, 57 were positive for tetracycline. We used a Lincoln-Petersen estimator for sampling without replacement to derive an initial population estimate of  $12,400 \pm 3000$  bears. Because estimates based on recapture samples collected in the same year as the marking have consistently been biased low (particularly in years like 2008, when natural foods were abundant), we expect the current estimate to significantly increase after samples from the 2009 harvest are added. Despite this, the estimate indicates that the bear population has declined since 2002.

### INTRODUCTION

Mark-recapture population estimates derived in 1991 and 1997 indicated that Minnesota's black bear population grew rapidly during the 1990s from about 15,500 ( $\pm 1500$  95%CI) bears in 1991 to around 24,000 ( $\pm 3000$ ) by 1997. In response to the perceived rapid growth rate, bear hunting permits were increased by 50% in 1998 and remained high through 2003, in an effort to curtail further population growth and avoid exceeding population levels that humans would tolerate, particularly in years when natural foods failed. At the time, some population modeling runs suggested that the bear population could “escape”, reaching levels that would not be controllable through hunting. However, a third tetracycline survey in 2002 yielded a population estimate of 25,000 ( $\pm 4500$ ) bears, indicating that high hunting pressure had stemmed population growth and bear numbers had leveled off. Following the 2002 estimate, hunting permits were progressively reduced in an attempt to keep the population from declining.

After 2002, however, several indicators suggested that the population had begun to decline: (1) the number of bears killed by hunters was consistently lower than expected, based on food availability and hunter numbers (regression model); (2) population reconstruction, based on harvest age structure, suggested a recent downturn in bear numbers; (3) the proportion of the harvest composed of reproductive females continued to increase; (4) for several years, radiocollared bears experienced unsustainably high hunting mortality, and (5) wildlife managers consistently reported low levels of nuisance activity and a paucity of bear sign. The persistence of these observations indicative of population decline, despite reductions in hunting permits, prompted a 4<sup>th</sup> tetracycline mark-recapture survey in 2008.

A growing problem in previous tetracycline surveys had been the increased incidence of non-target species, specifically raccoons (*Procyon lotor*), fishers (*Martes pennanti*), and martens (*Martes martes*) taking baits. This reduced both the precision and accuracy of the estimates. By 2002, about a third of tetracycline baits were taken by small carnivores, making them unavailable to bears. In anticipation of a similar problem, Alaskan researchers (Peacock 2004) enclosed baits in wooden boxes that bears, but not small carnivores (martens in this case), could dismantle. A similar box design was successfully employed in Wisconsin to also preclude

fishers and raccoons from taking tetracycline baits. We adopted this approach for the 2008 tetracycline survey in Minnesota.

## OBJECTIVES

1. Obtain an updated estimate of the number of black bears ( $\geq 1$  year old) in Minnesota;
2. Increase the precision of the population estimate by reducing the proportion of baits normally taken by non-target species;
3. Test the effectiveness of adding beaver (*Castor canadensis*) meat to baits to increase their attractiveness and thereby mark more bears.

## METHODS

### Marking

During June and July, 2008, baits with tetracycline were set across the bear range in Minnesota (Figure 1A). Bait sites were located within a grid approximately 4.8 km apart; adjacent baits were not set along the same road or trail. Where it was not possible to achieve optimal spacing, baits were set at alternative sites, but not closer than 3.2 km apart. Baits were checked 3–4 weeks after setting.

Each bait consisted of 0.4–0.5 kg of bacon, wrapped tightly around 9 500-mg capsules of tetracycline. The bait was completely enclosed within a wooden box. The box was stapled to a backboard that was then nailed 2.5-m high in a soft-barked tree. Prior tests indicated that bears, but not raccoons or fishers, could pull the box off the backboard and access the bait. Four 1.2-cm holes were drilled into the sides of the box to allow scent to emanate (but small enough to keep out squirrels). As an extra attractant, a 0.2-kg patty of ground whole beaver was added to 2/3 of the bait boxes. Again, prior tests suggested that ground beaver was more attractive to bears than bacon. After boxes were secured to tree trunks, 2 tablespoons of semi-solid grease (waste grease from a restaurant “rib-cooker”) was smeared on the outside of the box to enhance the scent at bait sites. There was not sufficient grease available to supply all bait sites, so some baits were set without added attractant and some were enhanced with other types of attractants supplied by survey participants.

Because of unexpectedly low visitation rates by bears at tetracycline baits during June and early July, 299 supplemental baits were set in late July at selected sites (Figure 1B). One objective was to try to mark more bears, although this is normally not an opportune time to bait because of the ripening of natural fruits. A second, more important objective was to assess the *cause* of the low visitation to the initial baiting. We posed 3, non-mutually-exclusive hypotheses: (1) lack of attractiveness of the baits due to poor scent emanation from inside the box; (2) lack of attractiveness of baits due to rotting of the beaver meat; or (3) low bear population. We tested these using the following 4 bait configurations, set at bait sites where bears had not taken the bait during June or early July:

1. Bacon packed in a plastic mesh bag and wired to a tree trunk; this was the method used in 1991, 1997, and 2002, so served as the control.
2. Bacon packed in a wooden box, with grease smeared on the outside, as in the 2008 survey.
3. Bacon packed in a wooden box, with beaver meat hung in a plastic mesh bag on a nearby tree.
4. Bacon packed in a wooden box, with bacon fat and molasses smeared onto the tree and loose bacon placed atop the box as extra attractants.

### Recapture and Population Estimates

We obtained “recapture” samples from hunters. During the fall bear hunting season,

hunters were required to submit 2 teeth (first upper premolars) and a 5-cm sample of rib bone from the bear they shot. Rib bones, cut in cross-section, were screened for tetracycline under a microscope illuminated with ultraviolet (UV) light; tetracycline incorporated into the bone tissue fluoresced yellow under UV illumination. Tetracycline reliably appears in rib bones of any bears that ingested a 4500-mg dose of tetracycline (Garshelis and Visser 1997). The mark persists until internal remodeling of bone tissue replaces older bone material, which, in young bears, can occur within 3 years. However, the amount of remodeling varies, and we could not reliably distinguish, using rib bones, marks caused by ingestion of other environmental sources of tetracycline in other years. In contrast, tetracycline marks in teeth are permanent, and their time of deposition can be determined by matching the position of the mark to cementum annuli. Thus, for all marked ribs, we examined a corresponding tooth. We also used teeth to distinguish cubs (bears <1 year old, which although not legally harvested, were sometimes shot inadvertently); these were excluded from the recapture sample.

We derived population estimates using a Lincoln-Petersen (L-P) estimator for sampling without replacement, as all sampled bears were dead and thus permanently removed from the population. Population estimates refer to the time of marking (July 2008), and exclude cubs. In marking years prior to 2008, we used a cumulative sample of recaptures from 2–3 years of harvesting, but these estimates still pertain to the year of marking.

We divided the data by Bear Management Unit (BMU) and made separate estimates for each. However, we had to group some units together either because sample sizes were too small or because there was strong evidence that bears moved between BMUs from the time of marking to the time of recapture (this is a problem in all cases, but more so for some units).

## RESULTS

### Marking

Wildlife managers set >3500 baits in 2008, more than in any previous survey (Table 1). New areas baited included the Red Lake and Bois Forte Indian Reservations, as well as several areas at the fringe of the Minnesota bear range. Bears visited only 17% of bait sites, considerably less than in past surveys (Table 1); visits were also surprisingly clumped, with large areas having no bear visits (Figure 2A). Also surprising, 18% of bears that visited baits either did not break into boxes or broke into boxes but did not consume the bait. In past surveys, <6% of baits visited by bears (ascertained from their claw marks on the tree) were not eaten. Consequently, fewer baits were eaten by bears in 2008 than in any previous marking year.

Both the addition of beaver meat to baits and the use of external attractants influenced the attractiveness (or detectability) of bait sites to bears. During initial baiting, bears investigated 19% of baits containing bacon plus beaver but only 14% of those with just bacon (Table 2;  $\chi^2 = 11.5$ ,  $P = 0.0007$ ). Also, baits with beaver meat were more likely to be consumed (85%), if investigated, than those without (78%;  $\chi^2 = 3.8$ ,  $P = 0.05$ ). The net effect of including beaver meat in baits was an extra 110 marked bears. Boxes smeared with the grease we supplied attracted more bears than boxes with no attractant, but boxes enhanced with attractants supplied by survey participants attracted the most bears ( $\chi^2 = 25.4$ ,  $P < 0.0001$ ). The effects of beaver meat and these other attractants appeared to be independent and additive (Figure 3).

The wooden boxes prevented most non-target animals from accessing baits and may have reduced their ability to detect baits. Visits by animals other than bears, primarily small carnivores, was down during initial baiting from 38% in 2002 to only 6% in 2008, and of those baits visited, only 17% were consumed (Figure 2B). During supplementary test baiting in late summer, small carnivores visited 12% of bagged baits, but only 9% of those in boxes. They ate 58% of the bagged baits they investigated, but failed to gain access into any of the boxes (Table 2).

However, boxes also may have interfered with bears' detection and consumption of baits, though this was not conclusive due to small sample sizes (Table 2). Bears visited baits in mesh bags nearly twice as often as baits in boxes with "rib-cooker" grease or beaver meat added to the site external to the bait box. Molasses and bacon grease, though, seemed to be more effective attractants; those baits were visited as frequently (or more) than bagged baits (Table 2). Boxes clearly interfered with consumption of baits; whereas bears ate 93% of the bagged baits that they investigated ( $n=14$ ), they consumed only 59% of the baits in boxes ( $n=22$ ;  $\chi^2 = 4.9$ ,  $P = 0.03$ ). Notably, even bagged baits were visited far less frequently than in previous years, when this was the method of tetracycline delivery (Tables 1, 2).

### Recapture and Population Estimates

The 2008 bear harvest was lower than in any of the previous 5 years, due in part to fewer hunters and very good fall foods. Harvest was similar to 2002, a year with excellent fall foods, and coincidentally the last time that bears were marked with tetracycline baits. Hunters in 2002 and 2008 submitted a similar number of usable teeth and rib bones and the number of samples that were positive for tetracycline was nearly identical (Table 3). However, because the total number of bears marked in 2008 was lower than in 2002, the number of marks recovered by hunters represented a larger proportion of the total in the population, yielding a considerably lower estimate of population size.

Samples obtained from the 2008 fall hunting season produced a population estimate of 12,400 (95% CI: 9400–15,600) bears. This was lower than any of the previous 3 estimates, based on the first year of recapture samples (i.e., from the 1991, 1997, and 2002 hunting seasons, Table 4). Population declines appeared to have occurred throughout the north-central, northeastern, and southern portions of the bear range, but not in the northwestern BMUs (Figure 4).

## DISCUSSION

Tetracycline marking of bears during summer 2008 was less successful than we had hoped and was particularly disappointing in light of the extra effort made to increase the number of bears marked by increasing the number of baits, increasing their attractiveness (with beaver meat and extra lure), and preventing non-target animals from obtaining them. We had hoped to increase the number of bears marked in order to narrow the confidence intervals bracketing a new population estimate. Confidence intervals had increased with each successive tetracycline survey to date (Table 4), due to non-target animals taking baits; this had hampered our ability to interpret population trends.

Several factors contributed to the relatively poor success at marking bears in 2008. First, wild summer bear foods were better than average in all parts of the bear range and more abundant than in any previous tetracycline-marking year (Garshelis and Noyce 2009). Wild sarsaparilla (*Aralia nudicaulis*), chokecherry (*Prunus pennsylvanicus*), Juneberry (*Amelanchier* spp.), and blueberry (*Vaccinium* spp) were particularly abundant and no species surveyed experienced crop failure. A number of berry species were somewhat delayed in fruiting due to early summer conditions, but, with abundant crops and good summer moisture, the fruit-bearing season lasted longer than usual. The widespread and prolonged availability of a variety of fruits during July and August (Noyce and Garshelis 1997) likely reduced the propensity for bears to investigate and consume tetracycline baits (Table 2). Second, housing baits in boxes probably reduced bears' detection of them and certainly interfered with their consumption. Anecdotal reports of signs of bears near baits that were not taken suggest that bears did not detect them from the range normally expected. There is no obvious explanation for at least 5 cases where bears visited baits and opened the box but did not consume the bacon. Addition of beaver to baits demonstrably increased both visitation and consumption rates, thus mitigating in part the poor detection and consumption rates attributable to the boxes. Results of test baiting suggest that finding the right attractants to use at bait sites could fully compensate for the negative

effects of enclosing bacon in boxes. Finally, visitation at baits was lower in 2008 because bear numbers appear to have declined significantly.

The tooth and rib “recapture” sample was also disappointingly small in 2008. The number of bears shot by hunters was 40% lower than the previous 10-year mean (Garshelis and Noyce 2009). This was attributable to better-than-average fall foods, reduced numbers of hunters, and reduced numbers of bears. Unlike 2002, the number of bears shot, after correcting for the number of hunters, was lower by about 25% than predicted based on fall food availability, continuing a trend that began in 2003 (Garshelis and Noyce 2009). Further exacerbating the relatively low number of recapture samples was a continued decline in the proportion of successful hunters that submitted samples (71% in 2008, compared to 91% in 1991; Garshelis and Noyce 2009).

The low sample sizes in both the mark and recapture phases of the 2008 tetracycline survey compromised both the accuracy and precision of the initial population estimate. Nevertheless, we believe there is ample justification for viewing this estimate as strong confirmation that Minnesota’s bear population is no longer increasing and has likely declined since 2002. The actual estimate, though, is most certainly biased low. Estimates based solely on samples collected during the fall immediately following marking have been consistently biased low (Noyce et al. 2001, Garshelis and Noyce 2006). This is because bears that consumed tetracycline baits during the summer are more likely than other bears to be shot over hunters’ baits the following fall. However, the linkage between bears consuming tetracycline baits and visiting hunters’ baits is much weaker a year later. Thus, the addition of another year’s samples (2009 hunting season) is likely to yield a higher and more accurate estimate. The amount by which the population estimate will increase is uncertain, but has ranged from 7% for the 1991 estimate to 45% for the 2002 estimate (Table 4). We assume, given the similarity of food conditions in 2008 and 2002, that the estimate could increase by an amount similar to 2002 (i.e., >45%) after another year of sampling. This would put the estimate at ~18,000 ( $\pm$ ~3500), which is markedly fewer bears than there were in 2002 (Table 4).

BMU-specific estimates should be viewed with caution, as they violate a basic assumption of the L-P mark-recapture procedure, namely geographic closure. We know that bears routinely move among BMUs during the interval between marking and recapture, and that the extent of these movements vary year-to-year with varying food resources. There is no way to account for such movements, and an argument could be made that BMU-specific estimates are therefore unreliable. We grouped units where it was evident that such movements occurred, and present the results with some diffidence. Nevertheless, they seem to confirm a widespread population decline, but indicate that this decline did not occur in the northwestern part of the state (despite some recent record high harvests in parts of this area). A better assessment of BMU-specific changes will be made after inclusion of samples from the 2009 harvest.

## **FUTURE DIRECTION**

We will continue to collect rib samples and teeth from dead bears at least through the 2009 bear hunting season. Depending on sampling success and the condition of fall bear foods in 2009, we will determine whether to extend sampling through a third hunting season in 2010 before calculating a final population estimate for 2008. Despite the limitations of the preliminary 2008 population estimate, we believe current evidence for recent declines in Minnesota’s black bear population is strong and that imposing annual reductions in bear hunting permits, as implemented since 2002, has been and continues to be a prudent course of action.

Contrary to popular belief, black bears in Minnesota have a relatively high reproductive rate and populations can rebound quickly when hunting pressure is reduced. Adult females are long-lived and have low natural mortality. Cub mortality in Minnesota is also low (18–20%), as is non-human-related mortality of juveniles. Control of hunter numbers constitutes a targeted means of effecting a quick turnaround in bear recruitment rates and population growth ( $\lambda$ ).



We recommend continued periodic statewide tetracycline surveys to track population trend, unless other, more suitable methods are discovered. Based on results of 2008 sampling, we recommend continued use of wooden boxes to contain baits, but with several modifications: (1) weaken the connection between the box and backboard to make it easier for bears to dismantle boxes; (2) include more attractant external to the bait boxes; and (3) conduct further experimentation with different attractants and box ventilation to determine ways of enhancing detection, attractiveness, and consumption of baits.

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Table 1. Tetracycline-marking data: 1991, 1997, 2002, and 2008.

	1991	1997	2002	2008
<b>Baits set and checked</b>				
Baits set	2905	2989	3122	3539
Baits not found	9	20	16	11
Baits checked	2896	2969	3106	3528
<b>Baits visited and eaten</b>				
Baits visited by other mammal or bird <sup>a</sup>	507	747	1181	218
Percent visited by other mammal or bird	18%	25%	38%	6%
Baits taken by a person	0	6	9	0
Baits taken by animal, not a bear	---	---	1015	37
Baits taken by ambiguous (possibly bear) <sup>b</sup>	2	64	30	16
Baits available for bears <sup>c</sup>	2701	2580	2572	3509
Baits visited by bears	1009	1214	755	594
Percent of available baits visited by bears	37%	47%	29%	17%
Baits eaten by bears	998	1213	707	490
Percent of visited baits eaten by bears	99%	100%	94%	82%
Percent of available baits eaten by bears	37%	47%	27%	14%

<sup>a</sup> Includes all baits visited by small mammals and/or birds. Some of these were not consumed; others were also visited by bears, in which cases they were recorded as taken by bears.

<sup>b</sup> These ambiguous cases were considered first as non-bears, then as bears in population estimates.

<sup>c</sup> Baits taken by small mammals or birds were considered to be available for bears half the time ( $\frac{1}{2}$  bait).



Table 2. Visitation rates of black bears and other carnivores at baits of several configurations deployed across Minnesota during June–mid-July (primary baiting period) and late July (supplemental test baiting), 2008. Percentages reflect the portion of the baits in the given configuration (*N*).

	<i>N</i>	Bears				Other carnivores			
		Baits visited		Baits eaten		Baits visited		Baits eaten	
		<i>n</i>	Percent of total	<i>n</i>	Percent of visited	<i>n</i>	Percent of total	<i>n</i>	Percent of visited
Primary baiting period									
Bacon only	1051	148	14%	115	78%	52	5%	4	7%
Bacon and beaver	2118	401	19%	340	85%	139	7%	21	15%
Total <sup>a</sup>	3169	549	17%	455	83%	190	6%	25	13%
Supplemental test baiting									
Bacon in bag	95	14	15%	13	93%	12	12%	7	58%
Bacon in box w. grease	67	5	7%	4	80%	5	8%	0	0%
Bacon in box w. beaver in bag	64	6	9%	2	33%	3	5%	0	0%
Bacon in box w. molasses and bacon fat	64	11	17%	7	64%	9	13%	0	0%
Total	290 <sup>b</sup>	36	12%	26	72%	27	9%	7	26%

<sup>a</sup> Totals for the primary baiting period are less than actual totals (see Table 1) because type of bait (with or without beaver) was not recorded at 62 original bait sites.

<sup>b</sup> Of 299 test baits deployed, 2 were not found again and 7 were miscoded as to type of bait deployed.

Table 3. Tetracycline recapture data: 1991, 1997, 2002, and 2008.

	1991	1997	2002	2008
Recapture				
Harvest	2143	3212	1916	2135
Ribs/teeth collected from harvest	1958 <sup>a</sup>	2594	1417	1488
Percent of harvest sampled	91%	81%	74%	71%
Ribs/teeth collected from nuisance or car killed bears	0	17	12	10
Cub samples excluded		13	16	23
Total samples checked for tetracycline	1958	2611	1429	1498
Positive samples				
Tetracycline-marked samples	122	149	56	57
Percent of samples marked	6.2%	5.7%	3.9%	3.8%
Double-marked samples	11	10	2	2
Percent of samples double marked	9.0%	6.7%	3.6%	3.5%

<sup>a</sup> Excluding cubs, which are not counted in population estimates.

Table 4. Tetracycline-based population estimates, 1991, 1997, 2002, and 2008. Final estimate for 2008 will not be available until after a second recapture sample (2009 harvest).

	1991	1997	2002	2008
Estimated no. marked bears <sup>a</sup>				
Excluding ambiguous cases	916	1134	680	473
Including ambiguous cases		1193	709	489
Population estimate from recaptures in year of marking (Yr 1)				
Mean of estimates with and without ambiguous cases	14,600	20,300	17,500	12,400
Lower 95% CI	12,300	17,000	13,000	9,400
Upper 95% CI	16,900	24,000	22,200	15,600
Population estimate from recaptures in year after marking (Yr 2)				
Mean of estimates with and without ambiguous cases	15,800	25,600	27,900	
Lower 95% CI	13,400	20,300	20,160	
Upper 95% CI	18,200	31,100	35,860	
Population estimate from recaptures in Yr 1 + Yr 2				
Mean of estimates with and without ambiguous cases	15,300	22,400	22,700	
Lower 95% CI	13,700	19,400	18,400	
Upper 95% CI	16,800	25,400	27,100	
% increase from first-year estimate	4.8%	10.3%	29.7% <sup>b</sup>	
Final population estimate <sup>c</sup>				
% increase from first-year estimate	6.8%	18.2%	44.6% <sup>b</sup>	

<sup>a</sup> Baits consumed by bears (Table 1) divided by estimated rate of double marking (Table 2).

<sup>b</sup> Abundant fall foods and low hunter success rate in 2002 created a much larger first-year under-estimate in 2002 than in 1991 or 1997. A similar situation existed in 2008.

<sup>c</sup> Final estimate is the average of Yr2 and (Yr1 + Yr2) estimates. Modeling suggests this to be least biased.

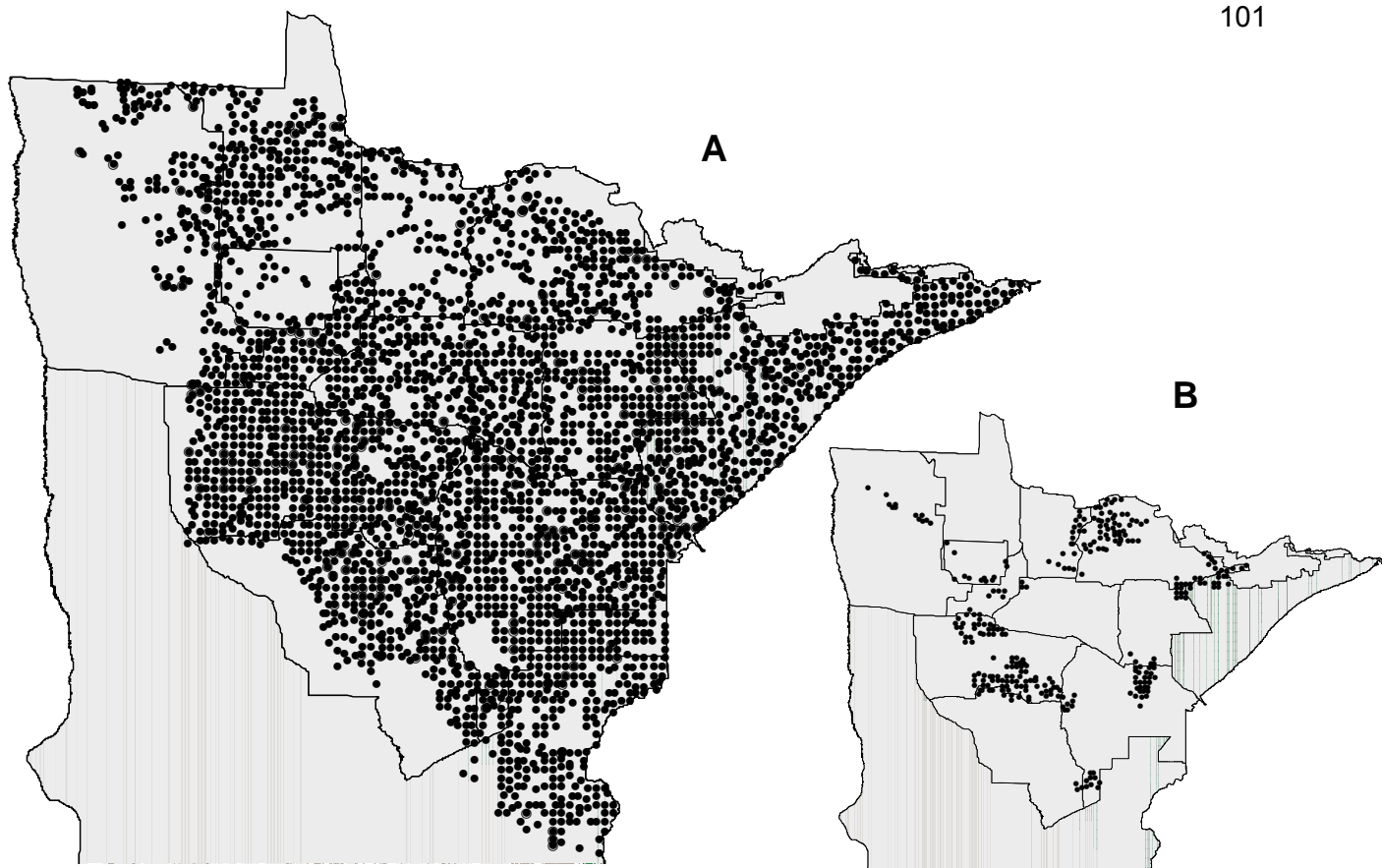


Figure 1. Locations of tetracycline baits across Minnesota's black bear range, summer 2008: (A) initial baiting, June and July, and (B) supplementary test baiting, late July.

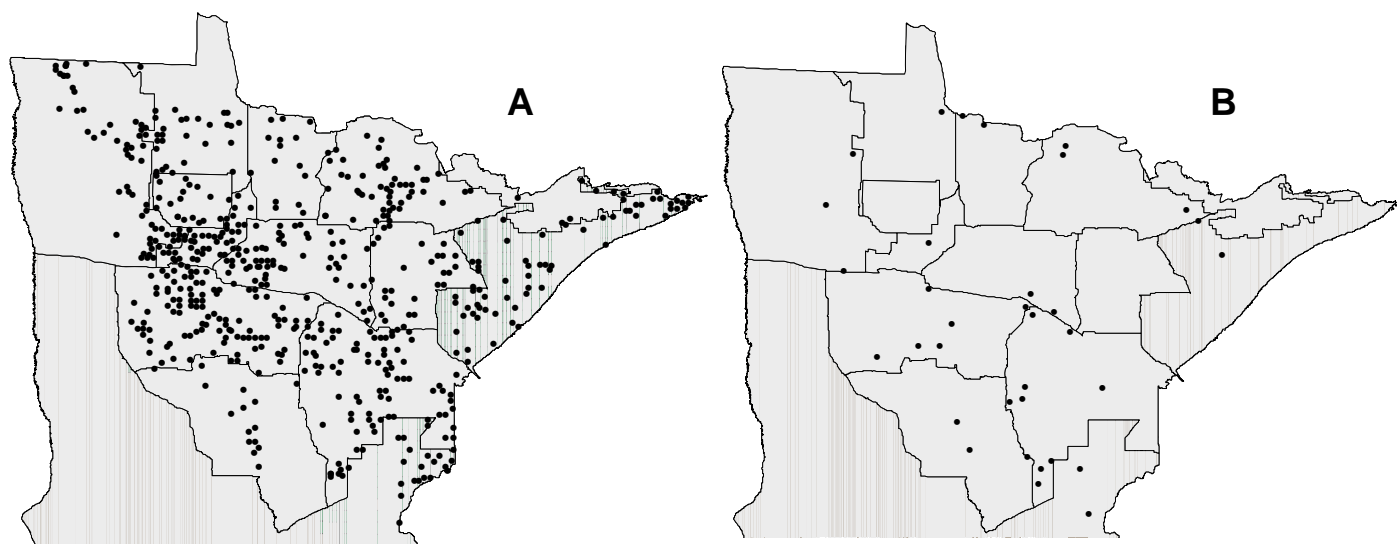


Figure 2. Locations of tetracycline baits consumed by: (A) bears and (B) small carnivores (fisher, marten, and raccoon) across Minnesota's bear range, summer, 2008.

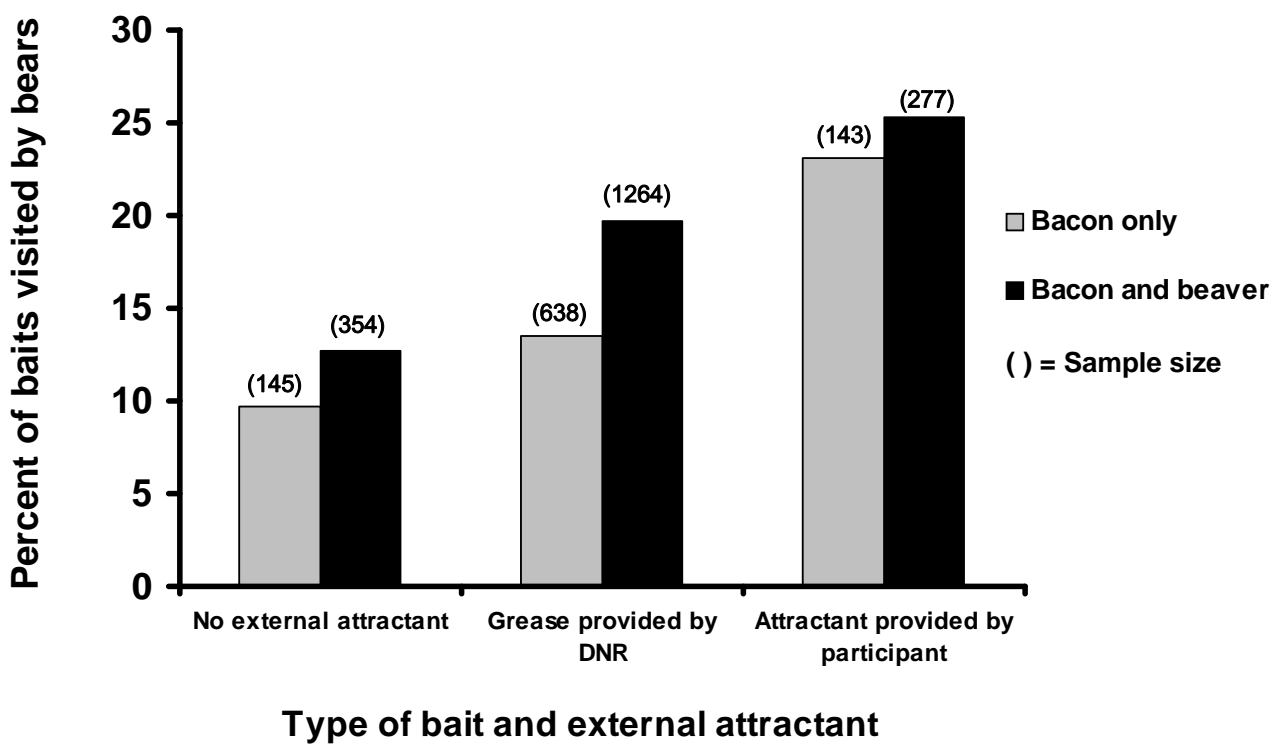


Figure 3. Influence of beaver meat and external attractants on the visitation rates of bears at tetracycline baits in Minnesota, June – July 2008.

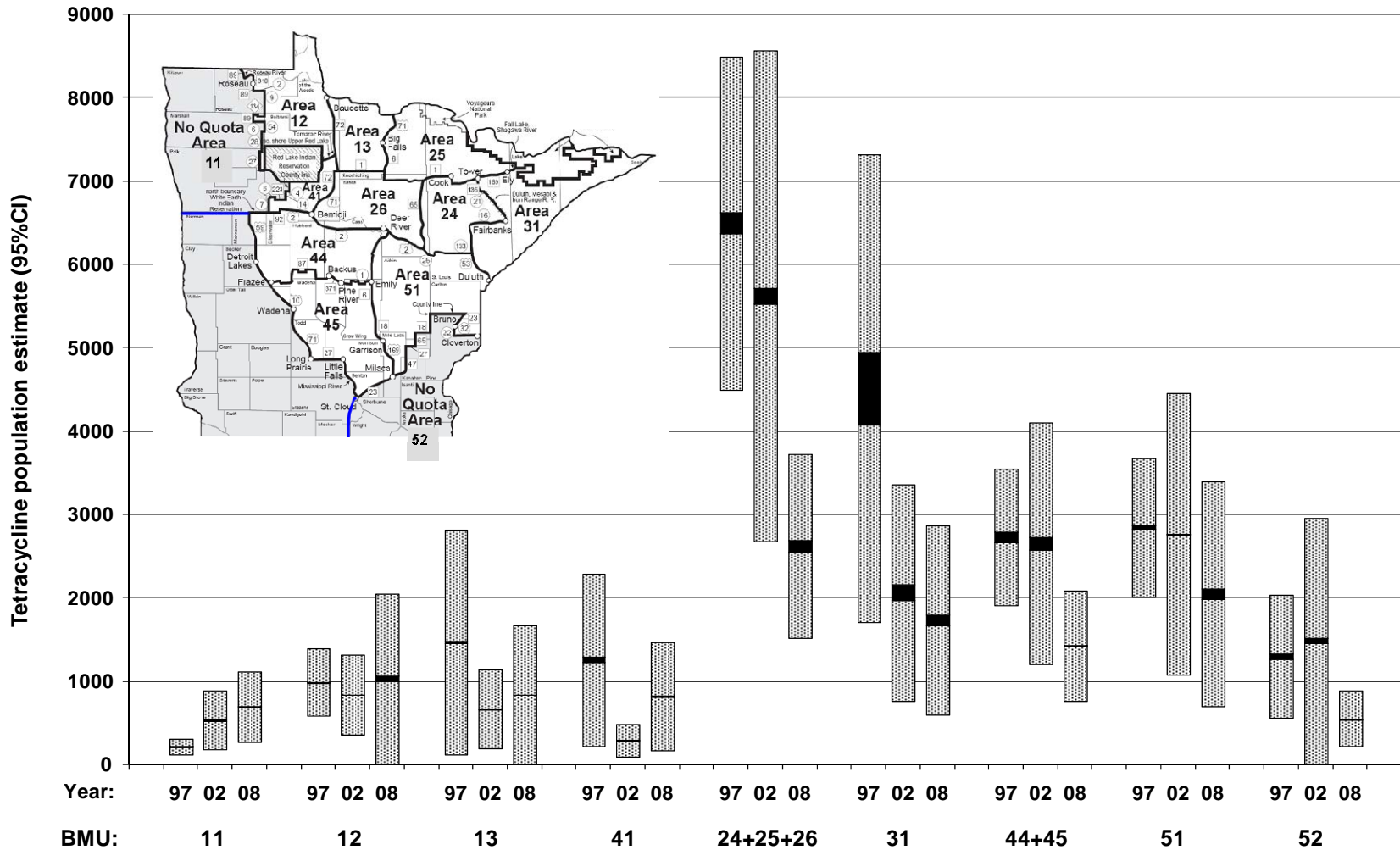


Figure 4. Population estimates of Minnesota black bears, by Bear Management Unit (BMU; boundaries shown on inset map), derived from tetracycline marking, based on sample recoveries only in the year of marking, 1997, 2002, and 2008. Black band in middle of each bar represents range of estimates; stippled bar shows span of 95% CI. Estimates based on recoveries from the year of marking tend to be biased low, by variable amounts, so across-year comparisons should be made with caution.

## HABITAT SELECTION BY MALE RUFFED GROUSE AT MULTIPLE SPATIAL SCALES

Meadow J. Kouffeld<sup>1</sup>, Michael A. Larson, and R. J. Gutiérrez<sup>1</sup>

### SUMMARY OF FINDINGS

No results are available yet for this study. Field work began during spring 2009.

### INTRODUCTION

The Minnesota Department of Natural Resources (MNDNR) set a goal of increasing the hunting harvest of ruffed grouse (*Bonasa umbellus*) from a mean of 561,000 birds/year (1976–2005; MNDNR, unpublished data) to a mean of 650,000 birds/year (MNDNR 2007). Achieving that goal likely will require increasing the quality or quantity of ruffed grouse habitat in Minnesota or both.

Although ruffed grouse occur in forest stands not dominated by aspen and in regions where aspen is sparse or does not exist (Devers et al. 2007), they reach their highest densities in aspen forests (Rusch et al. 2000). Young aspen stands provide dense vertical stems used as cover by grouse, particularly drumming males and females with broods. The flower buds of older male aspen trees are a favored winter food source for grouse. Classic grouse habitat, therefore, consists of close juxtaposition of multiple age classes of aspen in relatively small patches, so within an area the size of a typical grouse home range a grouse can access the various resources the different age classes provide (Gullion and Alm 1983, Gullion 1984).

All of the MNDNR's Subsection Forest Resource Management Plans (SFRMPs) that have reached the stage of defining "Desired Future Forest Conditions" have prescribed a conversion of many acres of managed forest land from an aspen cover type to another cover type [-5 to -33%, MNDNR 2001, 2003, 2004 (revised 2006)]. Recent plans for the 2 national forests in Minnesota call for similar conversions (USFS 2004a, 2004b). Restoration of a historical forest composition (i.e., range of natural variation or pre-settlement benchmark) was used to justify reducing the area of the aspen cover type in the future. Furthermore, global climate change is likely to influence conversions of forest cover types and other aspects of ruffed grouse habitat.

Although Gullion clearly showed an association between ruffed grouse and aspen (Gullion and Alm 1983), he did not explicitly investigate landscape patterns in ruffed grouse habitat. Furthermore, he left some uncertainty about the effect of pine stands in particular on ruffed grouse habitat by reporting high densities of drumming males associated with aspen clones in pine plantations under some unspecified conditions (Gullion 1990). Zimmerman (2006) conducted the only recent analysis of ruffed grouse habitat at a landscape scale. He found that the densities of drumming male grouse along ~5-km strip transects were most highly correlated ( $r \approx 0.53$ ) with an index of evenness in the distribution of land area among 6 types of land cover, including 4 types of forest overstory. Evenness was correlated with the proportions of aspen and conifer cover types (positively and negatively, respectively). The data, therefore, were inconclusive about the effects of specific forest cover types on the density of drumming grouse at a landscape scale. Thus, it remains uncertain what the effect of landscape-scale changes in forest overstory composition will be on ruffed grouse populations.

At the scale of a few forest stands, the preference of grouse for aspen in several age classes is well known (Gullion 1984, Rusch et al. 2000). Zimmerman (2006) found that variation in the number of drumming male grouse in individual forest stands was best explained by a model that included patch shape and 9 forest overstory types. More grouse were located in young aspen stands and stands with low edge density, and fewer were in mixed hardwood-conifer stands and mature spruce-fir stands. Less is known, however, about the influence on grouse of the following patch and adjacency characteristics of forest stands: the presence of

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conifers in aspen stands, the presence of aspen clones in conifer stands, the relative importance of different age classes of aspen, and variation in the density of woody stems regenerating after harvesting aspen.

We designed this study to address remaining uncertainties about the relationships between grouse habitat and forest characteristics at multiple spatial scales. Our results will help wildlife managers make forest management recommendations consistent with achieving the ruffed grouse harvest goal stated in the MNDNR's Strategic Conservation Agenda.

## OBJECTIVES

1. To determine forest characteristics that are correlated with the presence of male ruffed grouse in forest stands and at specific drumming structures.
2. To determine forest characteristics correlated with the abundance of male ruffed grouse within landscapes comprised of many forest stands.

## STUDY AREA

In the Laurentian Mixed Forest Province we identified several potential study sites that were: (1) relatively contiguous blocks of state or county ownership; (2) >200 km<sup>2</sup>; and (3) contained both aspen and conifer cover types. We based cover types on GAP level 3 classification data. For selecting study sites our aspen type was the aspen/white birch type in GAP, and our conifer type included the pine, spruce/fir, upland conifer, and upland cedar types in GAP.

Six of the 9 potential study sites had >7 times as much area in the aspen cover type than in conifer cover types. The other 3 potential study sites had the most conifer cover (10–24%, ratios of conifer:aspen area = 0.46–1.33). The site with the most conifer cover was adjacent to 1 of the 6 aspen sites, so we selected them to comprise our study area. These sites are in portions of Red Lake Wildlife Management Area and adjacent Beltrami Island State Forest.

## METHODS

Data for this study will come from 2 sources. We will collect new data by surveying grouse and measuring vegetation characteristics at a study area that is as representative as possible of forests in northern Minnesota. These data will be used to analyze habitat selection by grouse at all 3 spatial scales (i.e., drumming structure, forest stand, and landscape). We will also use existing data from the MNDNR's annual ruffed grouse drumming count survey routes to conduct an independent analysis of habitat selection at the landscape scale.

### Data Collection

*New field data*—We identified 60 3- to 5-km transects in the study area. Each transect was delineated by starting at a point along a road or trail nearest one of 30 randomly located points in the aspen study site and 30 in the conifer study site. We determined randomly the direction of each transect from that point and when each transect intersected another road or trail. Drumming grouse can be detected from approximately 200 m away (Zimmerman 2006), so we created a 200-m buffer around each transect to define sample landscapes. The transects were ≥400 m apart at all points. We divided the sample landscapes into 3 groups of 20 based on the proportions of aspen and conifer cover—those with the most aspen, those with the most conifer, and those with the most equal proportions. The aspen and conifer cover types comprised ≥50% of each sample transect. We randomly selected 10 transects from each of the 3 groups to sample for our study.

Each of the 30 selected transects will be surveyed on foot beginning 0.5 hours before sunrise during 8 different mornings during an 8-week period ending on the Friday nearest 31 May. When drumming grouse are detected during a survey, the exact location of each one will be determined by approaching it and identifying the log or other platform on which it was standing to drum, often indicated by the presence of fresh droppings. Drumming locations will be estimated with a hand-held GPS unit and can be confirmed by approaching again during subsequent surveys.

During Zimmerman's (2006) study, only 6% of detections were >200 m from the transect, and the probability of detecting a drumming grouse within 175 m of survey transects was not correlated with the distance from the transect. Assuming the mean probability of detection will be similar during our study (0.31), the probability that a drumming grouse that is present within 175 m of our transects will be detected at least once during 8 surveys will be approximately 0.95.

We will measure characteristics of ruffed grouse habitat at 3 spatial scales. The smallest scale will be the area immediately surrounding drumming locations identified during surveys. Characteristics at this scale will be measured in the field. The same variables will be measured at an unused but potential drumming platform (e.g., log or stump with no signs of use by grouse) nearest a randomly selected point within 85 m of each identified drumming location. A circle with a radius of 85 m represents the "core area" (2.3 ha) of a male's home range during the 2-month "drumming season" (6.7 ha, Archibald 1975). An 85-m radius will ensure that selected unused locations will be within the home range, whereas the 146-m radius of the home range would not.

The next scale will be the forest stand, which may be characterized by forest inventory data but will also be sampled in the field. The buffered transects will be the sampling unit for the landscape-level questions. Larger spatial scales for analysis (e.g., study area, Ecological Classification System land type association) may be possible by aggregating survey transects. Habitat characteristics at landscape scales will be quantified using the same forest inventory and land use/land cover data we use to identify study areas.

*Existing MNDNR annual survey data*—We will use existing ruffed grouse survey data, which are counts of drums heard at 10 points along road-side transects that have been surveyed once each year for many years. We will define sample landscapes consisting of the area within 175 m of each transect (i.e., to be more conservative about detection distance, given that each transect is surveyed only once each year) and seek existing Geographic Information System (GIS) data that represent land use and land cover information related to ruffed grouse habitat quality. We may randomly select a subsample of road-side landscapes to ground-truth remotely sensed data or digitize important features from aerial photos. We will quantify variables associated with ruffed grouse habitat in each road-side landscape using a GIS. We will select for analysis only drum count data collected within 2 years of when the landscape imagery was captured (i.e., 5 years total).

## Data Analysis

*New field data*—We will conduct a separate analysis at each spatial scale of interest. At the scale of specific drumming locations the analysis will follow a case-control logistic regression design in which the response variable is whether the point was used or not used (Keating and Cherry 2004). This may reveal selection for characteristics of drumming locations, given the constraint of occupying a limited home range. At all larger spatial scales we will use regression analyses in which the response variable is the count of drumming males (e.g., density within a forest stand or within 200 m of a transect). For all analyses we will define *a priori* models consisting of explanatory variables that represent hypothesized habitat relationships. We will use information-theoretic model selection procedures and consider multimodel inference (e.g., Burnham and Anderson 2002).

*Existing MNDNR annual survey data*—Annual drum counts are associated with specific points along each road-side transect. In most cases, however, much uncertainty exists about



the location of the points because they may not be documented and observers may not stop at exactly the same points each year. We will use the entire transect, therefore, rather than survey points as the sampling unit. We will sum the counts from all survey points on each transect for each annual survey. There may be much interannual variation in counts along a transect that is not associated with either habitat quality or the long-term grouse population cycle, so we will use the mean of 5 consecutive annual sums, rather than counts from a single survey, as an indication of the relative quality of grouse habitat along each transect. We will use the 5-year mean of annual counts as the response variable in regression models. Landscape metrics will be used in various combinations that represent our *a priori* hypotheses about ruffed grouse habitat relationships. We will use information-theoretic model selection procedures and consider multimodel inference (e.g., Burnham and Anderson 2002).

## ACKNOWLEDGEMENTS

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## HABITAT SELECTION, NEST SUCCESS, AND SURVIVAL OF SHARP-TAILED GROUSE IN RELATION TO MANAGEMENT OF OPEN LANDSCAPES IN MINNESOTA

J. Wesley Bailey and Michael A. Larson

### SUMMARY OF FINDINGS

The Forest Wildlife and Populations Research Group of the Minnesota Department of Natural Resources (MNDNR) initiated research to examine habitat selection, nest success, and survival of sharp-tailed grouse (*Tympanuchus phasianellus*) in relation to management of open landscapes in east-central Minnesota. The 2009 breeding season served as the pilot period of a planned long-term study. During spring 2009 we trapped and radiocollared 17 sharp-tailed grouse; 10 hens and 7 cocks from 4 dancing grounds. Because of late trapping efforts, we only caught hens at 2 of the 4 dancing grounds. As of July 6, 2009, unknown predators killed 3 hens and 1 cock. We found 8 nests all within a variety of habitats ranging in succession from wooded bog edge to willow brush-grass, to treeless residual grass within a burned meadow. The mean distance from nearest dancing ground to nest locations was 973.97 m (min. 504.17, max. 1899.74). The mean distance between neighboring nest locations for all samples was 1546.80 m (min. 528.25, max. 3074.80). Of the 8 nests, 1 was excluded from survival analysis because the hen died while foraging away from the nest; all 13 eggs remained in the nest. Of the 7 nests considered for analyses, predators destroyed 4 nests during late incubation and 3 nests successfully hatched at least 1 sharp-tailed grouse chick. For all nests, mean clutch size was 11.0 eggs (7 minimum, 13 maximum). To date, data collection is ongoing; therefore, complete results are not available at this time. We do, however, anticipate results from this study will help managers in east-central Minnesota focus their brushland management efforts, and provide the context to develop refined management objectives for managing open landscapes for sharp-tailed grouse consistent with MNDNR's Strategic Conservation Agenda.

### INTRODUCTION

Brush and open-landscape management efforts comprise a considerable portion of MNDNR's Section of Wildlife budget. Despite the time and resources allocated to open-brushland management, it is uncertain how these efforts have influenced sharp-tailed grouse populations. Annual dancing ground surveys may provide some insight into sharp-tailed grouse response to management. Information on sharp-tailed grouse habitat selection and survival is lacking and would better guide management efforts in Minnesota. This research will evaluate habitat selection of sharp-tailed grouse hens during the breeding season, determine habitat characteristics that best predict nest-sites and influence nest survival, and finally, determine factors contributing to hen mortality.

Sharp-tailed grouse have the most extensive range of all prairie grouse (Silvy and Hagen 2004), but for several decades have suffered marked population declines (Connelly et al. 1998) because of poor management or habitat loss (Riley 2004). In Minnesota, sharp-tailed grouse populations have declined sharply because open brushland habitat they inhabit is vulnerable to destruction, senescence, and conversion to incompatible cover (Berg 1997). Annual spring surveys of dancing grounds in Minnesota indicate the mean number of sharp-tailed grouse per dancing ground has fluctuated since 1980 but the overall trend has been negative, although a positive trend occurred within the past 5 years (Larson 2008).

Because of the population decline, and subsequent lower harvest, MNDNR set a long-term goal to nearly quadruple the annual sharp-tailed grouse harvest from a 5-year average (2000-2004) of 11,400 to 40,000 (MNDNR 2007). An increase in harvest may indicate a positive response to management. To attain this goal, forest planning efforts have identified priority open landscapes for brushland management (MNDNR 2007) with an objective of increasing the quantity and quality of sharp-tailed grouse habitat throughout its range in Minnesota. However, brushland management is expensive and this management activity

comprises a considerable portion of MNDNR Section of Wildlife budget. Openland/brushland management ranked first in average annual expenditures of all habitat management activities within the Forest Habitat Program for FY06-FY08 with mean expenses averaging \$800,000 (MNDNR, unpublished data).

Despite the amount of resources allocated to brushland management, significant information gaps about sharp-tailed grouse habitat relationships exist in Minnesota. Managers are unsure whether their actions benefit the species. Managers suspect their efforts are at least maintaining sharp-tailed grouse populations, but do not have current information to bolster their assumptions. A cursory examination of annual dancing ground counts compared to management in Aitkin County revealed no statistical correlation between the mean number of sharp-tailed grouse counted at dancing grounds and acres managed (Figure 1). Although annual dancing ground surveys may provide some insight to a population level response to management (Connelly et al. 1998), linking survival and productivity to habitat features would better direct management resources (Martin 1992).

Information on sharp-tailed grouse habitat in Minnesota stems from studies conducted in northwestern MN (Artmann 1971, Schiller 1973, Wells 1981) and in east-central MN (Niemi and Hanowski 1992, Hanowski et al. 2000). Based on results from these studies, MNDNR and Minnesota Natural Resources Conservation Service (MN NRCS) provide management guidance on recommended habitat characteristics for sharp-tailed grouse. However, without basic information on nesting habitat and factors that limit adult and nest survival, managers make habitat management decisions with limited information, which results in management of unknown efficacy. Results from this study would help managers make informed management decisions by linking on-the-ground efforts to positive responses by sharp-tailed grouse populations. MNDNR would achieve its goals for the sharp-tailed grouse more quickly and efficiently by understanding the impact of management activities.

## **OBJECTIVES**

1. To determine which habitat features most influence nest-site selection;
2. To determine which habitat characteristics and time-specific factors most influence nest survival;
3. Determine if sharp-tailed grouse hen mortality is influenced by time-specific factors, habitat characteristics at multiple scales, and site management history; and
4. Determine if site management history is correlated with habitat use of hen sharp-tailed grouse.

## **STUDY AREA**

During the 2009 pilot period, we conducted field work during late April through July on Willowsippi Wildlife Management Area and two private parcels in Aitkin County near Palisade, Minnesota (Figure 2). Each site has a history of open-brushland management and supports at least one sharp-tailed grouse dancing ground.

## **METHODS**

### **Capture, Marking, and Monitoring Sharp-Tailed Grouse**

We utilized radio-telemetry to determine sharp-tailed grouse habitat use and selection, nest success, and hen survival. We captured sharp-tailed grouse hens at active dancing grounds in late April and early May using walk-in funnel traps (Toepfer et al. 1987). To efficiently trap hens from several dancing grounds during any given trapping period (i.e., 2-4 days), we did not trap at dancing grounds more than 10-15 miles apart. We deployed and operated 4 traps, 1 per dancing ground, from 05:00-09:00 am and from 17:00-20:30 pm from 28 April through 07 May 2009. We fit sharp-tailed grouse with 14-15 g necklace radiocollars

(Advanced Telemetry Systems, Model A3960 and Holohil Systems, Model RI-2BM) with a lifespan of 18-24 months. Each transmitter included a 12 hour mortality sensor. We located each bird using a null-peak vehicle mounted telemetry system (Brinkman et al. 2002) and with a portable receiver and 2-element H-style antenna. To locate individual birds, we triangulated locations by obtaining 3 directional azimuths to the radiocollared bird's position (White and Garrott 1990). We used homing to find nests and locate dead birds (White and Garrott 1990).

### **Sampling Design for Habitat Selection During Breeding Season**

The most important reproductive decisions a hen must make is selecting where to nest and selecting sites where broods can grow and be protected from predators (Bergerud and Gratson 1988). Furthermore, nest success and survival of sharp-tailed grouse hens and broods during the breeding season may be the most important vital rates affecting population viability (Schroeder and Baydack 2001, Manzer and Hannon 2007). Therefore, our objective is to determine habitat selection during the breeding season with emphasis on obtaining locations pre- and post-hatch. To determine habitat selection, we will triangulate each hen's location daily to once every 5 days, depending on if hens are actively nesting or brood-rearing, until the end of the breeding season.

We will attempt to collect  $\geq 30$  locations per 3-month spring-summer field season for each sharp-tailed grouse hen (Seaman et al. 1999, Leban et al. 2001). Before hatch, hen movements will likely be limited to foraging bouts near the nest (Connelly et al. 1998, Roersma 2001). Therefore, locations may be more concentrated in space, in which case, fewer observations may be needed to characterize habitat use during laying and incubation.

We will evaluate habitat selection by determining the number of locations per hen within defined cover types and disturbance categories during the breeding season. Cover types are defined as those amenable to management (e.g., alder/willow swamp). For each hen, we will use a Geographic Information System (GIS) to delineate cover types of interest on digital aerial photographs covering breeding home ranges as defined by the minimum convex polygon of telemetry locations from all hens. If cover types cannot be distinguished from digital imagery, we may ground-truth these as needed. To determine cover type categories, we will use Minnesota Gap Analysis Project GAP cover types and cover types within the modified Minnesota land cover classification system. Because management histories at study sites will likely include more than 1 management type (e.g., burn, shear, herbicide), we will evaluate time since disturbance rather than management technique. We will determine time since disturbance for lands with a history of management during 1999-2008 and categorize time since disturbance into 2-year intervals (e.g., 0-1, 2-3, etc.). We will include 2009 management information as data become available.

### **Sampling for Nest-Site Selection**

We will collect nest-site habitat data immediately after nest fate is determined. Breeding home ranges define available habitat during the breeding season; however, nesting habitat requirements may differ from brood-rearing habitat (Connelly et al. 1998). For each nest, we will identify 2 non-nest points by selecting a random bearing and distance from the nest to measure available habitat. Selection of nest-sites occurs after hens select nesting areas (Johnson 1980; third-order selection). Because hens will nest near locations from the previous season (minimum distance 0.2 km; Roersma 2001) we will constrain sampling of random non-nest points to within 55-120 m of the nest to ensure we are sampling habitat available to each hen. We will ensure that non-nest points are in suitable habitat by selecting another random point and distance should a non-nest point occur in unusable habitat (e.g., forest). We will search non-nest points to be sure a nest is not present. At each non-nest point we will measure the same habitat variables used at the actual nest.

We will evaluate nest-site selection by measuring used habitat variables from a variety of spatial scales and compare these to available habitat. The scales of interest include habitat

features directly at the nest, within 30 m, 100 m, and 3200 m of the nest. We chose a 30 m radius because this scale represents the minimum mapping unit for MN GAP data and approximates the smallest scale at which management may appreciably affect sharp-tailed grouse habitat. We chose a 100 m radius because at this scale available habitat should include many potential nest sites, and the 3200 m represents the maximum distance sharp-tailed grouse are known to nest away from a dancing ground (Connelly et al. 1998).

Because we are interested in structural features of sharp-tailed grouse habitat amenable to management, we will focus on type of cover selected for nesting, vegetation height, and density at and surrounding the nest. We will estimate overhead cover by placing a black 20-cm diameter disk with 9 25-mm<sup>2</sup> equally spaced white squares in the nest bowl (or random point), and count the number of white squares  $\geq 50\%$  blocked by vegetation measured from 1-m away directly overhead (Roersma 2001, Manzer 2004). We will measure the height of woody and herbaceous vegetation at the nest and random points to the nearest 0.1 m.

Because sharp-tailed grouse often nest under or near brush (Connelly et al. 1998), our measures focus on quantifying the characteristics of the brush patch associated with the nest. We will use the area of an ellipse to calculate the area of the brush patch. We will then calculate mean vegetation height by measuring the height of the tallest stem in each cardinal direction from the center of the brush patch using a telescoping measuring pole. We will calculate brush patch volume by multiplying patch area by mean patch height and will use volume in analyses. To measure low vegetation cover we will use a density board (Nudds 1977, Noon 1981) 2 m high and 0.3 m wide and divided into 0.25-m sections each containing 25 5x5 cm squares. We will place the board at the nest and determine the percent cover of woody and herbaceous vegetation in each 0.25-m section from 5 m away in each cardinal direction by subtracting the number of squares  $< 50\%$  covered from 100. For analysis, we will use the average value of each 0.25 m section. If during the pilot season we determine sharp-tailed grouse are nesting in grassland more than brush, we may instead use Robel pole readings, which are a standard measure for grassland bird nests (Robel et al. 1970). Within 30 m of the nest, we will count the number of stems  $\geq 1.50$  m in height because these may provide perches for avian predators (Manzer and Hannon 2005).

We will classify cover types on digital aerial photography based on MN GAP cover types and cover types within the modified Minnesota land cover classification system, and convert vector data to raster to facilitate analyses. We will use GIS to measure spatial pattern (i.e., configuration) and characteristics of woody and herbaceous cover. We will use FRAGSTATS (McGarigal and Marks 1995) to measure the following characteristics: area (ha) of cover types, cover type patch size (ha), mean cover type patch size (ha), number of cover type patches, cover type density (number/100 ha), edge density (m/ha) between cover types, core area metrics (e.g., total core area), and nearest neighbor metrics (e.g., average proximity of brush patches to determine if sharp-tailed grouse select brush cover on average closer to other brush patches). We will record the distances from nests to "hostile" features, both natural and anthropogenic, (e.g., shelterbelts, buildings, improved roads, unimproved roads, transmission lines, hard forest edges) to determine whether the features are related to nest location (Pitman et al. 2005).

### **Sampling for Nest Survival**

After capture and allowing for an adjustment to the transmitter to lapse (1 week), we will triangulate hens daily to locate early nesting attempts. Once a hen has initiated a sedentary behavior (i.e., is found repeatedly at same location), we will home in on the hen and visually confirm the nest location, mark it with GPS, photograph the nest location and surrounding vegetation, and record the number of eggs. We will remotely monitor hens every 1-2 days during the nesting period. We will obtain a final clutch-size count when the hen is absent from the nest. On subsequent visits, we will assess if nests are still active by triangulating locations every 1-2 days. To avoid flushing the hen, we will only visit nests if monitoring indicates incubation has ceased (i.e., multiple absences) or if the hen substantially changes movements

(e.g., long-distance away from the nest or more frequent movements). We will visually inspect the nest at that time to determine nest stage, nest fate, and number of hatched eggs. We will consider all nests hatching  $\geq 1$  egg successful. If a nest is depredated, we will attempt to locate the hen and repeat daily sampling until a new nest location is determined; afterwards, we will repeat our nest monitoring protocol of triangulating hens daily to once every 2 days until a fate is determined.

During the course of nest monitoring and post-hatch monitoring, we will check for mortality signals to the end of the breeding season. Although our focus is on survival during the breeding season, and because transmitters will have a 24-month battery life, we will remotely check for mortality signals approximately 2 times a month throughout the year. Dead hens will be recovered as quickly as possible and categorized by the probable cause of mortality into 4 classes: avian predation, mammalian predation, unknown cause, and exposure. We will classify predation type while in the field by assessing any damage to the transmitter and carcass, and the location of the recovery site. Exposure is assigned if a severe weather event occurred immediately before mortality is indicated. If no damage is found on the transmitter and/or carcass, we will assign this as unknown.

## Data Analysis

Before collecting radio-telemetry observations, we will evaluate telemetry error to estimate precision of directional azimuths (White and Garrott 1990, Withey et al. 2000). To test error for actual field conditions (Withey et al. 2000), and to emulate transmitters on birds, we will place 5 transmitters 10 cm off the ground attaching them to saline filled bottles (J. Giudice, personal communication). Locations will be unknown to the observer in habitats representative of our study area. We will obtain 60 directional bearings for each transmitter from 4 known locations (Lesmeister et al. 2009). We will report linear error of each estimate, the standard deviation of bearing errors, and estimated size (ha) of each error polygon (confidence ellipse associated with location estimate). We will use a regression model to relate linear error to observer, distance of transmitter from receiver, and geometry of bearing intersections (Withey et al. 2000). To account for telemetry error, we will exclude all locations with error ellipses larger than 2.0 ha or the mean size of grass-brush patches within each dancing ground complex, whichever is larger.

To evaluate habitat selection, we will use a type 2 use-availability design where individual sharp-tailed grouse hens are identified and use is measured for each hen and availability is measured at the population level (Thomas and Taylor 1990, Alldredge and Griswold 2006, Thomas and Taylor 2006). We define the population as each individual dancing ground complex; therefore, availability will be defined as the extent of a minimum convex polygon around the collection of breeding season home ranges (Lesmeister et al. 2009) of hen sharp-tailed grouse within each dancing ground complex. Home range characteristics will be compared to those of the study area (i.e., dancing ground complex; second-order selection; Johnson 1980) because managers manage for large blocks of habitat. The experimental unit is an individual sharp-tailed grouse hen. We will define use as the proportion of time spent within individual cover types based on the number of observations per hen. We will not include successive locations of hens at nest sites in these analyses. We will use the Johnson rank-order method (1980) to rank the difference in proportional use of habitats and proportional availability of habitats for each sharp-tailed grouse hen using Hotelling's *T*-square or equivalent approximate *F* statistic (Johnson 1980, Alldredge and Griswold 2006). Johnson's procedure allows for inference only at the population level and does not require independence of animal relocations, which allows for the continuous observation of individuals (Alldredge and Griswold 2006).

Because we determined the number of non-nest points and will constrain sharp-tailed grouse selection to within the same habitat area as the nest, this represents a discrete-choice type analysis (Keating and Cherry 2004). We consider sharp-tailed grouse nests uncommon given the amount of available nesting cover types within study sites and we will search non-nest

points to be sure a nest was not present; therefore, we will meet assumptions for case-control or discrete-choice analyses (Keating and Cherry 2004). To assess nest-site selection, we will employ case-control conditional logistic regression (Allison 1999, SAS Institute 2004 [PROC LOGISTIC with STRATA statement]) to compare nests to paired non-nest points. Each matched set will consist of the nest (case) and 2 random points (controls) located within 55-120 m of the nest.

To assess nest survival, we will use the logistic-exposure method (Shaffer 2004) to model the effects of habitat variables from our nest-site selection analyses and time-specific variables (e.g., date, nest stage) to estimate nest survival. The logistic-exposure method allows values for time-dependent explanatory variables, such as nest stage, to change among nest-observation intervals, but assumes they are constant within an interval (Schaffer 2004). We will fit logistic exposure models with the GENMOD procedure (SAS Institute 2004) by selecting a binomial response distribution (i.e., nest failed or survived the interval) and supplying the user defined link function described by Schaffer (2004).

We will use an information-theoretic framework (Burnham and Anderson 2002) to evaluate support for our sharp-tailed grouse nest-site selection and nest survival hypotheses. Nest-site selection will evaluate habitat measured at multiple scales while nest survival will evaluate temporal and habitat effects. For the nest-site selection analyses, we will plot predicted probabilities that a site was a nest site as a function of covariates that had effects on nest site selection. We also will report descriptive statistics for nest-sites and non-nest points. For the nest survival analysis, we will first determine the most supported temporal effects (e.g., date of season, nest stage) and then include these in all habitat models; holding these nuisance effects constant in each habitat model will reduce the total number of models we fit and let us focus on habitat effects in the second stage of the analysis (Grant et al. 2005). For nest-site selection and nest survival analyses, we will examine each global model for multicollinearity by calculating tolerance values for variables (Allison 1999) and evaluate the goodness-of-fit with the Hosmer and Lemeshow (2000) goodness-of-fit test and will examine the overdispersion parameter (Burnham and Anderson 2002).

We will evaluate support of candidate models based on Akaike's Information Criteria for small sample sizes ( $AIC_c$ ) and report likelihood values,  $AIC_c$ ,  $\Delta AIC_c$ , and Akaike weights ( $w_i$ ) for each model (Burnham and Anderson 2002); we will use the effective sample size (Rotella et al. 2004) to compute  $AIC_c$ . Because of the potential for substantial model selection uncertainty, we will present model-averaged coefficients, unconditional standard errors, odds ratios and 95% confidence intervals from models with  $\Delta AIC_c \geq 2$  (Burnham and Anderson 2002). We will limit interpretation of effects to those with odds-ratio confidence intervals that did not overlap 1. We will estimate the relative importance of covariates from averaged models by summing the Akaike weights ( $w_i$ ) from all competing models in which the variable appeared (Burnham and Anderson 2002).

Survival over the breeding season will be measured from 15 April to 15 July annually. We will estimate breeding season survival using known-fate models in Program MARK (White and Burnham 1999) and use model selection to evaluate hypotheses about differences in survival during this time-period. We will calculate survival probabilities for radio-marked hens using the Kaplan-Meier product limit estimator (Kaplan and Meier 1958) with the staggered entry design (Pollock et al. 1989). We will code each encounter as live, dead, or censored. Because sharp-tailed grouse may need to adjust to radiotransmitters after capture, we will exclude any deaths within 1 week of capture from survival modeling.

## RESULTS

During spring 2009 we trapped and radiocollared 17 sharp-tailed grouse; 10 hens and 7 cocks from 4 dancing grounds. We caught 3 hens at Willowsippi WMA and 7 hens at Rono dancing ground; we did not capture hens at Sherman nor Gun Lake dancing grounds. As of July 6, 2009, unknown predators killed 3 hens and 1 cock.



We found 8 nests all within a variety of habitats ranging in succession from wooded bog edge to willow brush-grass, to treeless residual grass within a burned meadow. The mean distance from nearest dancing ground to nest locations was 973.97 m (min. 504.17, max. 1899.74). The mean distance from nearest dancing ground to nest locations was 1160.06 m at Rono and 663.82 m at Willowsippi; for all samples, the mean distance was 973.97 m. The maximum distance between nest locations at Rono was 3074.80 m; the minimum distance was 528.25 m. At Willowsippi, the maximum distance between nest locations was 1357.80 m; the minimum distance was 702.73 m. The mean distance between neighboring nest locations for all samples was 1546.80 m (min. 528.25, max. 3074.80).

Of the 8 nests, 1 was excluded from survival analysis because the hen died while foraging away from the nest; all 13 eggs remained in the nest. Of the 7 nests considered for analyses, predators destroyed 4 nests during late incubation and 3 nests successfully hatched at least 1 sharp-tailed grouse chick. For all nests, mean clutch size was 11.0 eggs (7 minimum, 13 maximum).

## DISCUSSION

While these data have not been analyzed, we anticipate results from this study should provide managers in east-central Minnesota with new insight into their brushland management efforts, and provide the context to evaluate and adjust their management of brushlands for sharp-tailed grouse consistent with MNDNR's Strategic Conservation Agenda (MNDNR 2007). By evaluating results of management practices, specific habitat features that directly influence reproductive success should be identified, increasing our ability to effectively conserve habitats (Martin 1992). Our results will allow managers to set quantitative goals for vegetation management by providing the range and configuration of cover types that benefit reproduction and survival of sharp-tailed grouse. All information gained from this study will help formulate cost-effective strategies for sharp-tailed grouse management and provide information on habitat relationships needed to effectively manage sharp-tailed grouse.

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Table 1. Distance (meters) to nest locations from dancing grounds where hens were captured during 2009 breeding season in Aitkin County, Minnesota.

Hen	Dancing ground	Distance to nest (m)
166.139	Rono	1246.45
166.238	Rono	1899.74
166.434	Rono	746.18
166.445	Rono	804.43
166.395	Rono	1103.50
166.120	Willowssippi	784.49
166.161	Willowssippi	702.81
166.483	Willowssippi	504.17

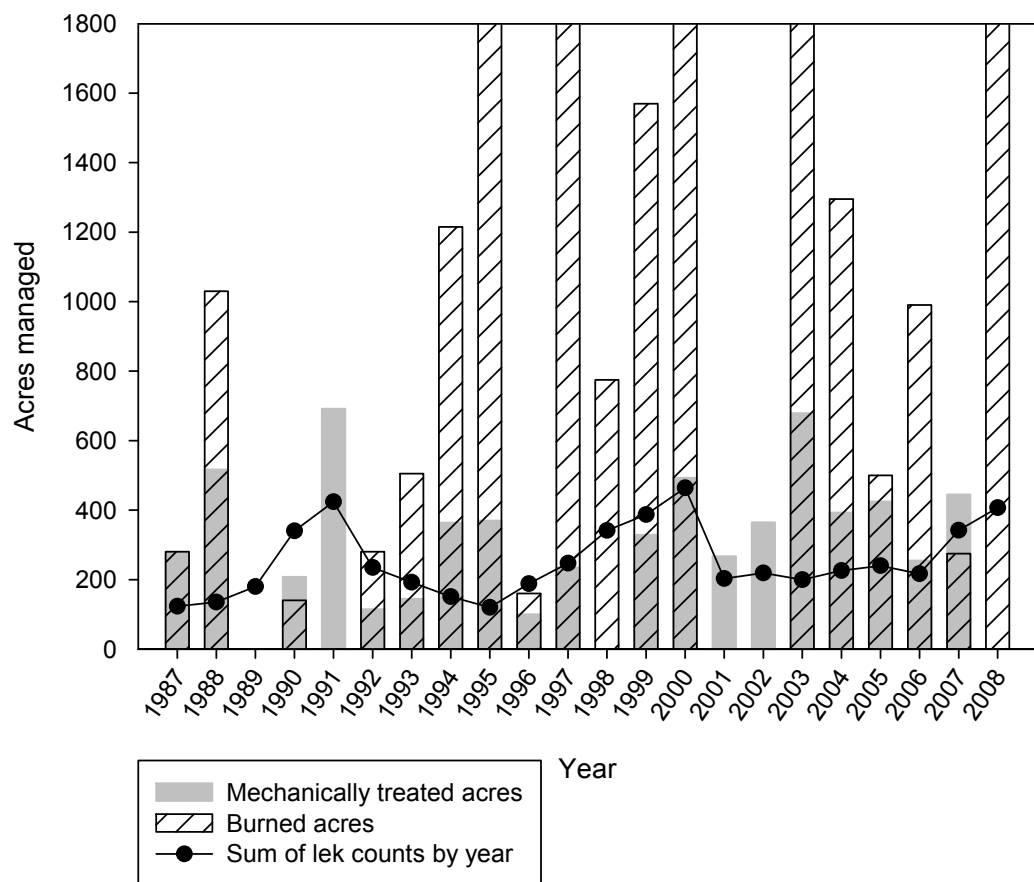


Figure 1. Relationship between mean count of male sharp-tailed grouse from annual dancing ground surveys and acres of managed brushland by treatment type within the Aitkin Wildlife Work Area during 1987-2008 in Aitkin County, Minnesota. Acres managed scaled to best show relationship; annual burned acres exceed 1800 in various years.

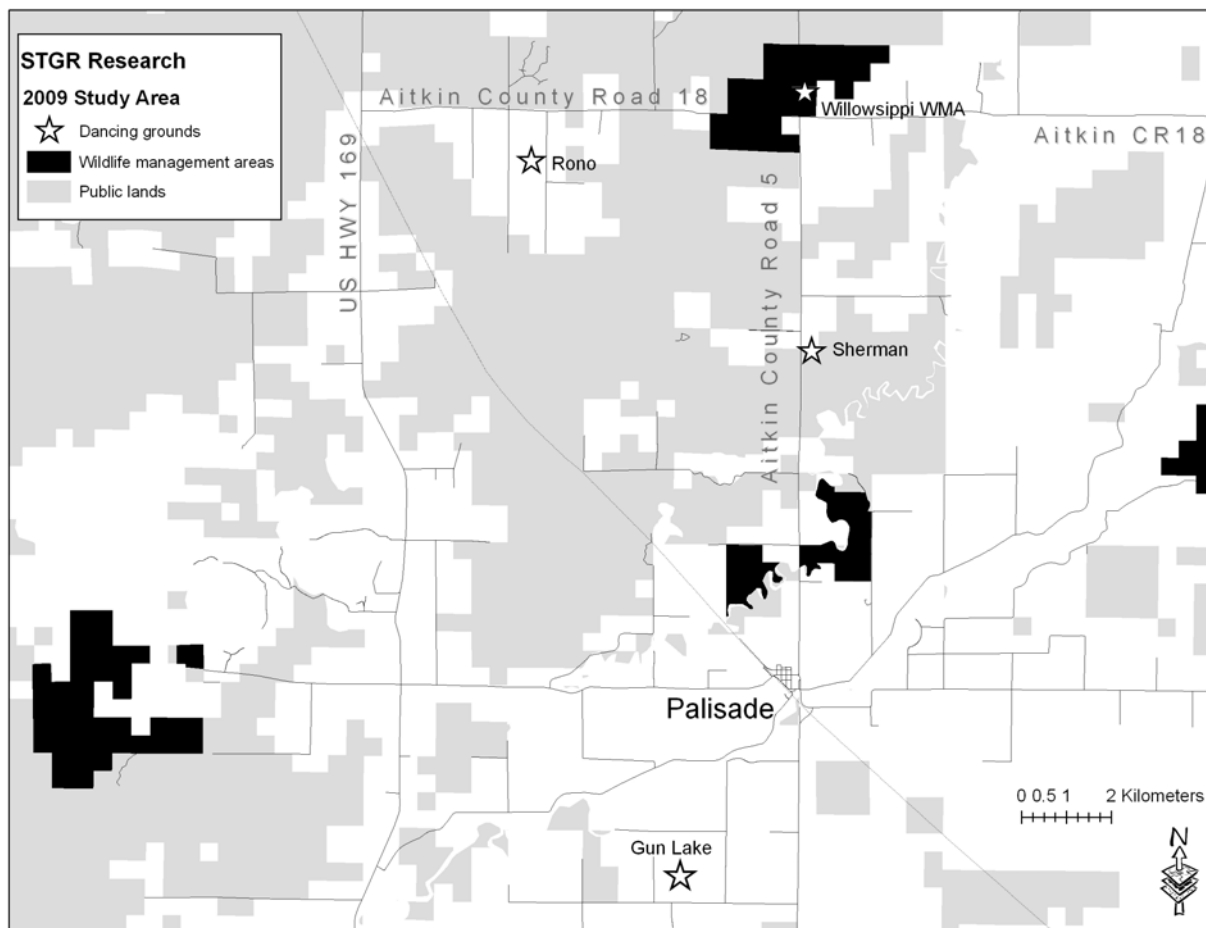


Figure 2. East-central study area in Aitkin County, Minnesota, during 2009 pilot season. Stars indicate locations of sharp-tailed grouse dancing grounds where trapping occurred and indicate focus areas for radio telemetry observations.

## 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 and Barry A. Sampson

### SUMMARY OF FINDINGS

The goal of this long-term (1991-2005) investigation is to assess the value of conifer stands as winter thermal cover/snow shelter for white-tailed deer (*Odocoileus virginianus*) at the population level. Over the course of this 15-year study period, we radiocollared and monitored a total of 452 female deer, including 43 female newborn fawns. Data generated from this study provided the basis for scientific and popular articles addressing supplemental feeding effects on winter food habits of white-tailed deer; age-specific survival and reproduction; cause-specific mortality; seasonal migration; safe capture, chemical immobilization and handling; wolf predation; bait selection and capture success; and diseases of deer; as well as progress in applied geographic information system (GIS) technology. The focus of several of these papers was to explore new, more scientifically rigorous analytical approaches to viewing the diverse data sets we were accumulating. During the past year, we've completed the publication process for the last of these papers, addressing seasonal migration and approaches to analysis (i.e., time-scale and origin) of deer survival from birth to 18.5 years old. We have completed organization and quality control checks of several large data sets, and we have begun statistical analyses of relationships between environmental variation (e.g., severity of winter weather conditions, harvests of conifer cover), the physiological status of study deer, their use of habitat, and survival, focusing on nutrition as a mechanistic thread. Below we highlight some of these preliminary findings.

### INTRODUCTION

The goal of this long-term investigation is to assess the value of conifer stands as winter thermal cover and snow shelter for white-tailed deer at the *population level*. Historically, the availability of conifer stands has declined markedly relative to the increasing numbers 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 (to varying degrees) harvests of conifers compared to hardwoods, because of evidence at the *individual-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

In proposing and planning this study in autumn 1990, we hypothesized that winter severity and the availability of moderately dense (40-69% canopy closure [Class B]) and dense ( $\geq 70\%$  canopy closure [Class C]) conifer stands on winter range affects their use by female white-tailed deer as thermal cover or snow shelter, deer movements (i.e., migration) and distribution. Further, we hypothesized that nutrition is likely the mechanistic thread between this environmental variation (e.g., ambient temperature, snow depth, conifer availability) and the population performance (survival and reproduction) of deer. Relative to varying winter severities, the objectives of the comprehensive, quasi-experimental approach of this study have been to:

- monitor deer movements (i.e., migration) between seasonal ranges and on winter ranges by radio-telemetry (very high frequency [VHF] and Global Positioning System [GPS]) to assess spatial distribution;
- determine habitat composition of winter range study areas and deer use of specific vegetation types;
- monitor winter food habits;
- physiologically monitor winter nutritional restriction and condition via serial examination of deer body mass and composition, blood and bladder-urine profiles, and chemistry profiles of fresh urine voided in snow (snow-urine);
- monitor age-specific survival, cause-specific mortality, and reproduction; and
- collect detailed weather data in conifer, hardwood, and open habitat types to determine the functional relationship between the severity of winter conditions (including micro-climates), deer behavior (e.g., use of habitat) and their survival.

## STUDY DESIGN AND PROGRESS

This study (1991-2005) employed a replicated manipulative approach, which is a modification of the Before-After-Control-Impact design (BACI; Stewart-Oaten et al. 1986; see DelGiudice and Riggs 1996). The study involves 2 control (Willow Lake and Dirty Nose Lake) and 2 treatment sites (Inguadona Lake and Shingle Mill Lake), a 5-year pre-treatment (pre-impact) phase, a 4-year treatment phase (conifer harvest served 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<sup>2</sup> (5.0-9.1 mi<sup>2</sup>) in area. The study began with the Willow and Inguadona sites during winter 1990-1991. The Shingle Mill and Dirty Nose sites were included beginning in winter 1992-1993. The objective of the experimental treatment (impact) was to reduce moderately dense and dense conifer stands (good and optimum thermal cover/snow shelter, respectively) to what is considered poor cover (< 40% canopy closure [Class A]).

Data collected on all 4 study sites included the following: (1) descriptive quantification of deer habitat by color infrared air photointerpretation, 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 (e.g., rectal temperature, response times to immobilizing chemicals); (4) age determination by last incisor extraction and cementum annuli analysis; (5) data generated by laboratory analyses of physiological samples of all captured and recaptured female deer, 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 ultrasonography; (6) morphological measurements; (7) physiological assessment of winter nutritional restriction by sequential collection and chemical analysis of snow-urine; (8) seasonal migrations and other movements via VHF and GPS radiocollars; (9) habitat use; (10) annual and seasonal cause-specific mortality; (11) age-specific survival rates; (12) pregnancy determination; (13) winter food habits; and (14) movements, territory size, survival, and cause-specific mortality of radiocollared wolves. See DelGiudice and Sampson (2008), other previous issues of the Minnesota Department of Natural Resources' annual Summaries of Wildlife Research Findings, and associated publication lists for further details of this study.

### Winter Severity and Nutritional Status of Study Deer

Winter nutritional restriction or deprivation of white-tailed deer, moose (*Alces alces*), and

elk (*Cervus elaphus*) can be assessed by sequential collection and chemical analysis of fresh urine voided in snow (DelGiudice et al. 1988, 1989, 1997, 2001; Ditchkoff 1994; and others). Collecting urine specimens associated with fresh deer tracks within 72 hours of a recent snowfall allowed us to associate urine chemistry values and nutritional assessments to known times ( $\leq 72$  hours) with a high degree of certainty (DelGiudice et al. 1988).

Pooling across our 4 study sites, mean urea nitrogen:creatinine (UN:C) ratios of the snow-urine samples collected during winters of varying severities clearly shows that higher values, caused by starvation-level nutritional restriction and accelerated net catabolism of endogenous protein (DelGiudice et al. 1987, 1991, 1994), occurred during the 3 winters with the highest winter severity index values (1995-96, 1996-97, and 2000-01; Figure 1). The winter severity index (WSI) is calculated by accumulating 1 point for each day with an ambient temperature  $\leq -17.7^{\circ}\text{C}$  and 1 point for each day with snow cover  $\geq 38$  cm during November-May. During these 3 severe winters, maximum snow depths were 86-97 cm, more than twice as deep as threshold depths that cause deer to engage in energetically-costly bounding to move about (Kelsall and Prescott 1971, Moen 1976) and when forage availability is markedly diminished. When we examined the proportion of snow-urine specimens collected during each winter with UN:C ratios indicative of moderately severe ( $3.0 \leq x < 3.5$  mg:mg) and severe or starvation-level ( $\geq 3.5$  mg:mg) nutritional restriction, a clear pattern was revealed in each of the 4 study sites (Figure 2). Generally, severe nutritional restriction was most evident during winter 1995-96, 1996-97, and 2000-01, when winter conditions were most severe, but patterns varied somewhat among the 4 sites (Figure 2). Severe nutritional restriction was most apparent throughout the period 1992-93 to 2004-05 at Shingle Mill Lake and least apparent at Inguadona Lake. Maximum WSI values were significantly ( $P = 0.057$ ) related to the proportions of snow-urine specimens with UN:C ratios indicative of severe nutritional restriction of deer during the study period; WSI values accounted for about one-third ( $r^2 = 0.35$ ,  $y = 2.958 + 0.051x$ ) of the variation of these proportions (Figure 3). Of these WSI values, temperature-days ( $r^2 = 0.39$ ,  $y = -1.982 + 0.190x$ ,  $P = 0.039$ ,) exhibited a stronger relationship to these indicators of severe nutritional restriction than snow-days ( $r^2 = 0.30$ ,  $y = 5.031 + 0.065x$ ,  $P = 0.080$ ). Additionally, this indicator of severe nutritional restriction in our study deer was related to percent winter mortality ( $r^2 = 0.52$ ,  $y = 3.942 + 0.381x$ ,  $P = 0.013$ , Figure 4) and percent mortality by wolf predation ( $r^2 = 0.38$ ,  $y = 5.161 + 0.415x$ ,  $P = 0.044$ ). We have reported wolf predation as the primary source of natural winter mortality for these deer (DelGiudice et al. 2002, 2006). Additional statistical analyses of snow-urine chemistry profiles will focus on specific relationships within control and treatment sites relative to winter severity, experimental harvests of conifer cover, and use of habitat

### Winter Severity and Use of Conifer Cover

The 4 study sites are mosaics of various vegetative types (polygons) that we classified by the dominant 2-3 tree species, tree height, and for conifers, by canopy closure class as well. Using ArcMap of ArcGis (Version 9.3), we measured the nearest distance (m) of diurnally radio-located female deer (Dec-May) to conifer stands with moderately dense (Class B) and dense (Class C) canopy closures, which based on findings in the literature, serve as good to optimal thermal cover and snow shelter, respectively, for deer. In our preliminary examination of deer location and habitat data from 3 winters ranging in maximum WSI values from 60 to 195, mean nearest distances of deer were closer to dense (Class C) conifer stands and to moderately dense or dense (Class B or C) stands *within* each site during the more severe winter, except at Dirty Nose Lake (Figure 5). At Dirty Nose Lake deer were located relatively close to conifer stands of Classes B or C ( $\leq 75$  m) during all 3 winters, regardless of severity, and this may be related to the overall vegetative composition of this site, as well as the availability and arrangement of these conifer classes. We will be studying this and associated issues in more detail. On the other 3 sites (Willow, Inguadona, and Shingle Mill), there were significant ( $P \leq 0.05$ ) differences in the mean nearest distance to moderately dense or dense conifer cover



during the mildest (1994-95) versus the most severe (1995-96) winters; distances tended to be intermediate during the moderately severe winter (1993-94). Among all 4 sites, mean nearest distances of female deer to dense conifer stands were inversely related ( $r^2 = 0.30$ ,  $y = 323 - 0.978x$ ,  $P = 0.028$ ) to WSI values (Figure 6), with a slightly stronger association with snow-days ( $r^2 = 0.31$ ,  $y = 285 - 1.376x$ ,  $P = 0.026$ ) than with temperature-days ( $r^2 = 0.28$ ,  $y = 413 - 3.330x$ ,  $P = 0.036$ ). Mean nearest distances of deer to moderately dense or dense conifer stands were also related ( $r^2 = 0.35$ ,  $y = 158 - 0.488x$ ,  $P = 0.015$ ) to WSI (Figure 6). Interestingly, we also observed a marginally-significant inverse relation ( $r^2 = 0.19$ ,  $y = 134 - 2.790x$ ,  $P = 0.095$ ) between mean nearest distances of deer to moderately dense or dense conifer stands and the proportion of snow-urine UN:C ratios indicative of severe nutritional restriction (Figure 7). This lends evidence to the notion that during increasingly severe winters of deep snow, deer spend more time close to and within moderately dense to dense conifer stands, where nutrition tends to be less available and nutritional restriction is more severe.

These promising preliminary findings suggest that the diverse data we've collected during our 15-year study period will allow us to more closely assess potential relations between environmental variation, deer behavior (e.g., habitat use), and the vital rates that most strongly affect population performance. Additional analyses will focus more specifically on the potential effects of the experimental reduction of conifer cover on the 2 treatment sites versus little to no reduction on the 2 control sites.

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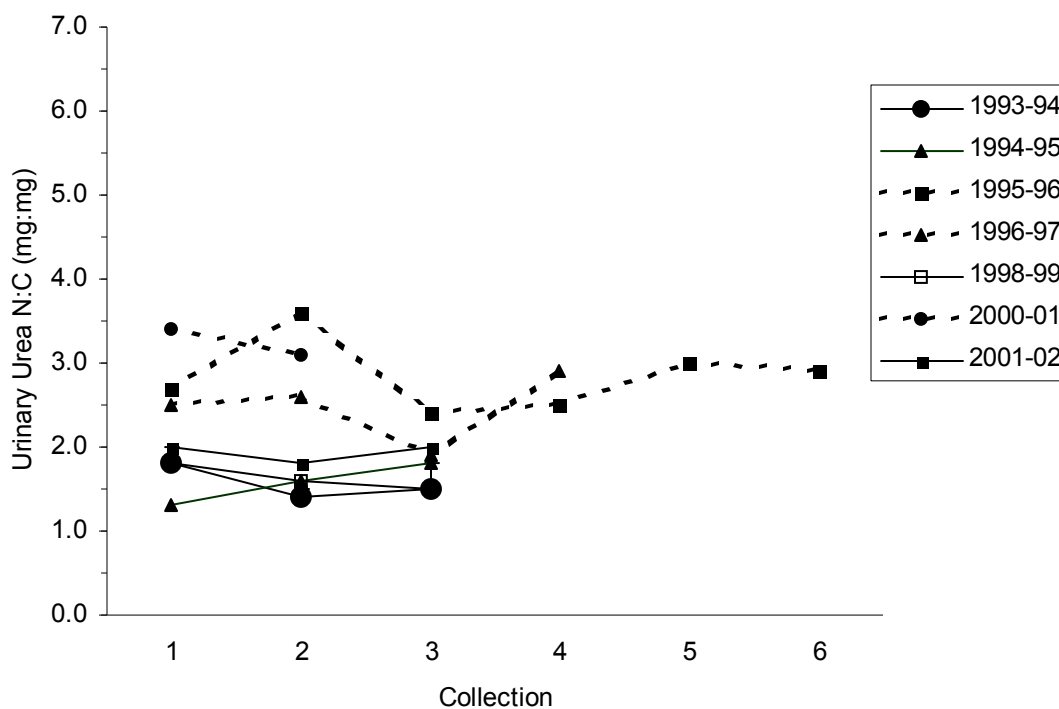


Figure 1. Mean urea nitrogen (N):creatinine (C) ratios in urine recently ( $\leq 72$  hours) voided in snow (snow-urine) by free-ranging white-tailed deer, all 4 study sites pooled, north-central Minnesota, January-March 1993-94 to 2001-02 (as indicated in the legend). Samples sizes per collection ranged from 94 to 143. Maximum winter severity index (WSI) values were 126, 61, 195, 159, 46, 153, and 45 (see text for definition).

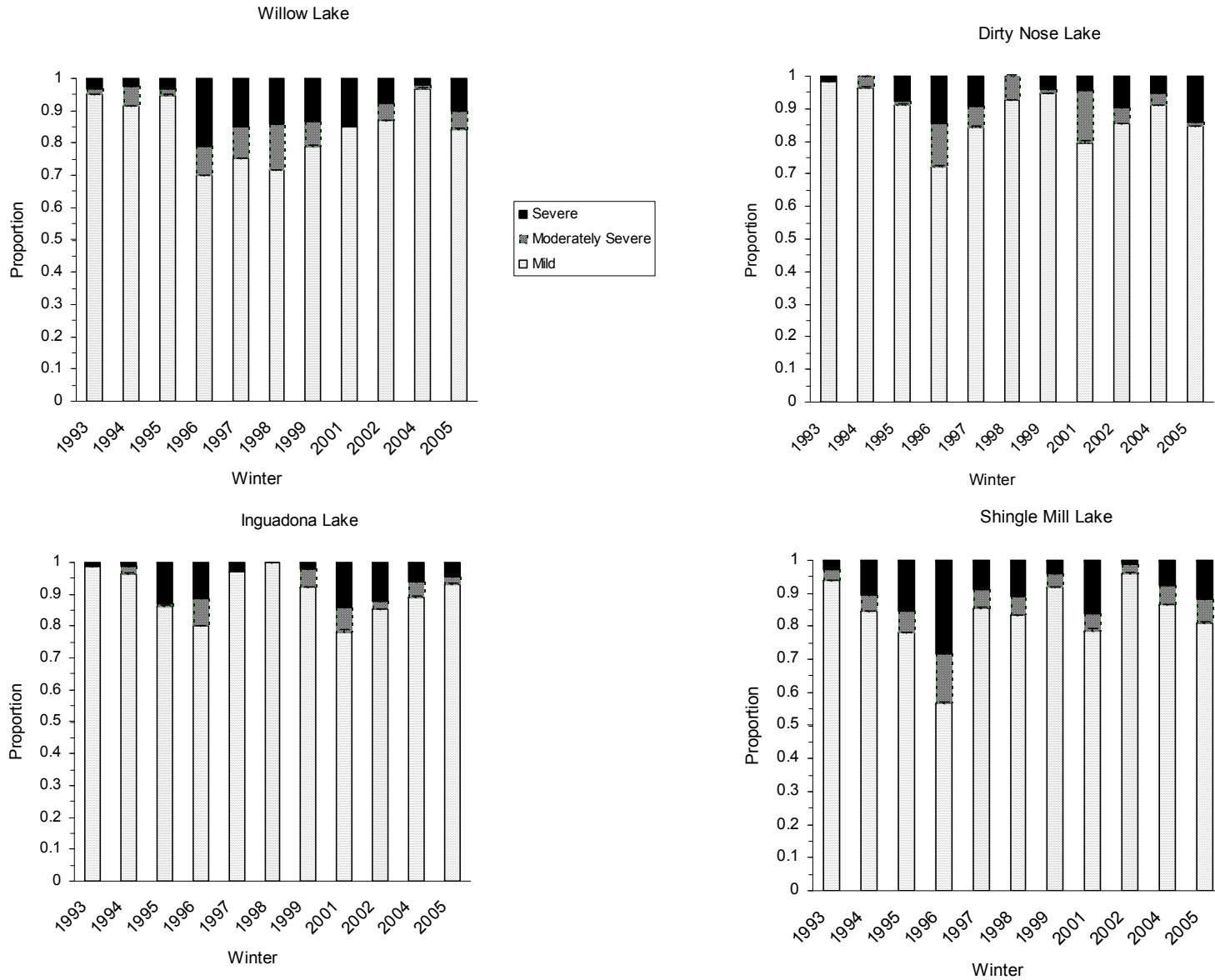


Figure 2. Proportion of urine specimens in snow (snow-urine) with urea nitrogen:creatinine ratios indicative of mild ( $< 3.0$  mg:mg), moderately severe ( $3.0 \leq x < 3.5$  mg:mg), and severe ( $\geq 3.5$  mg:mg) nutritional restriction in white-tailed deer on 2 control sites (Willow Lake, Dirty Nose Lake) and 2 treatment sites (Inguadona Lake, Shingle Mill Lake), north-central Minnesota, January-March 1992-93 to 2004-05 (as indicated on the x-axis).

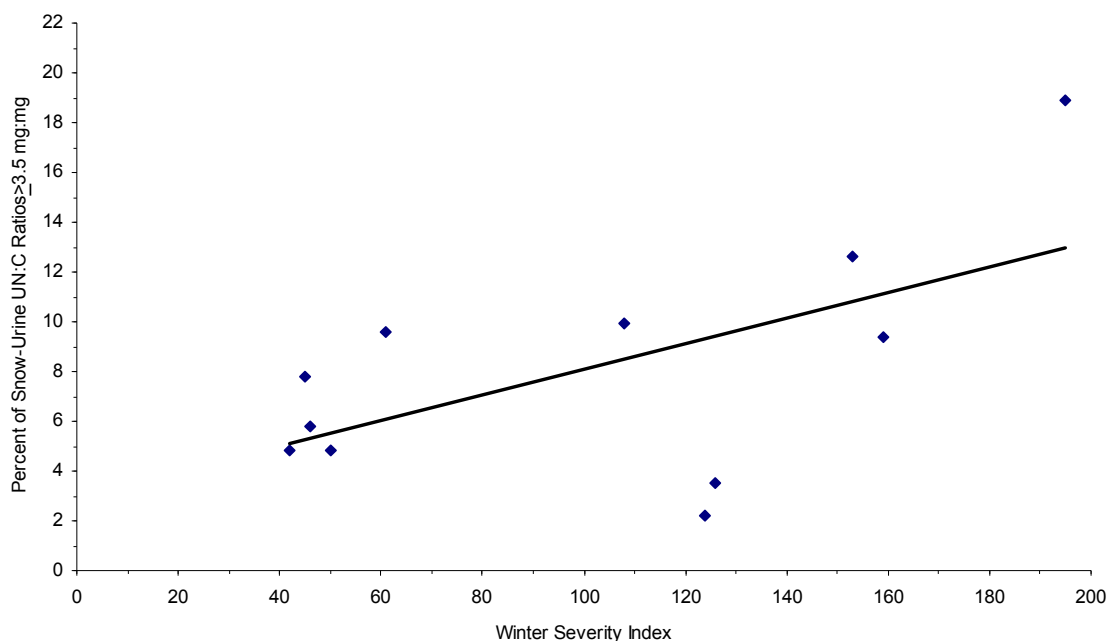


Figure 3. Relationship ( $r^2 = 0.35$ ,  $y = 2.958 + 0.051x$ ,  $P = 0.057$ ) of the annual maximum winter severity index (see text for definition) to the percent of urine samples in snow (snow-urine) of white-tailed deer with urea nitrogen: creatinine (UN:C) ratios indicative of severe nutritional restriction ( $\geq 3.5$  mg:mg), all 4 study sites pooled, north-central Minnesota, winters 1992-93 to 1998-99, 2000-01, 2001-02, 2003-04, and 2004-05.

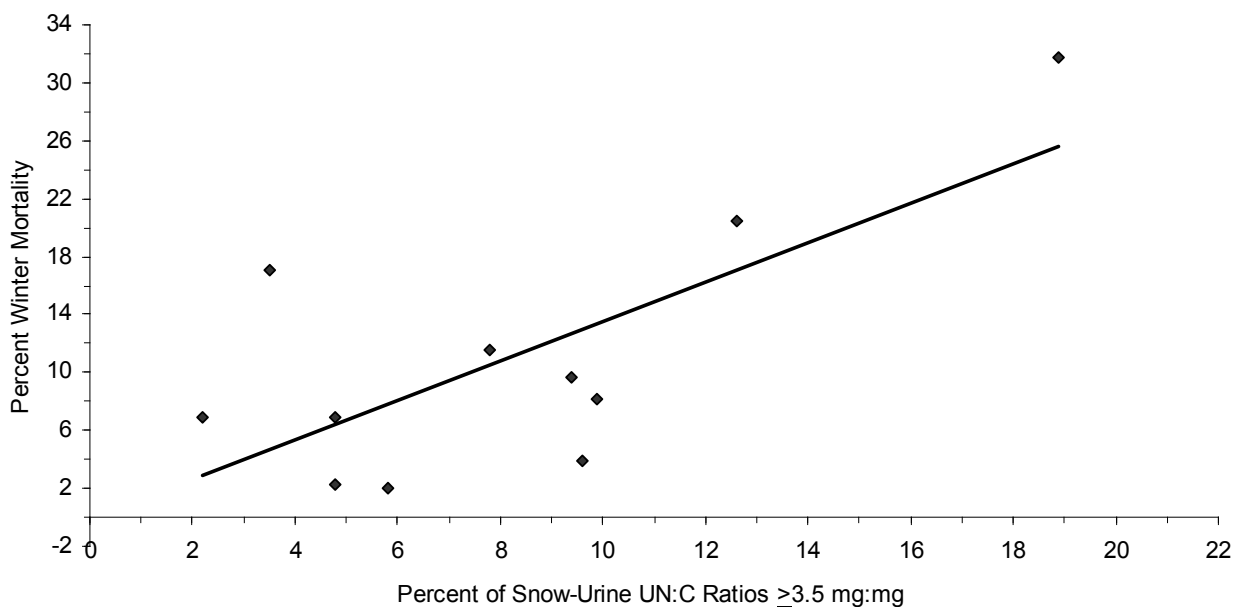


Figure 4. Relationship ( $r^2 = 0.52$ ,  $y = 3.942 + 0.381x$ ,  $P = 0.013$ ) of the annual percent of urine samples in snow (snow-urine) of white-tailed deer with urea nitrogen:creatinine (UN:C) ratios indicative of severe nutritional restriction ( $\geq 3.5$  mg:mg) to percent winter mortality, all 4 study sites pooled, north-central Minnesota, winters 1992-93 to 1998-99, 2000-01, 2001-02, 2003-04, and 2004-05.

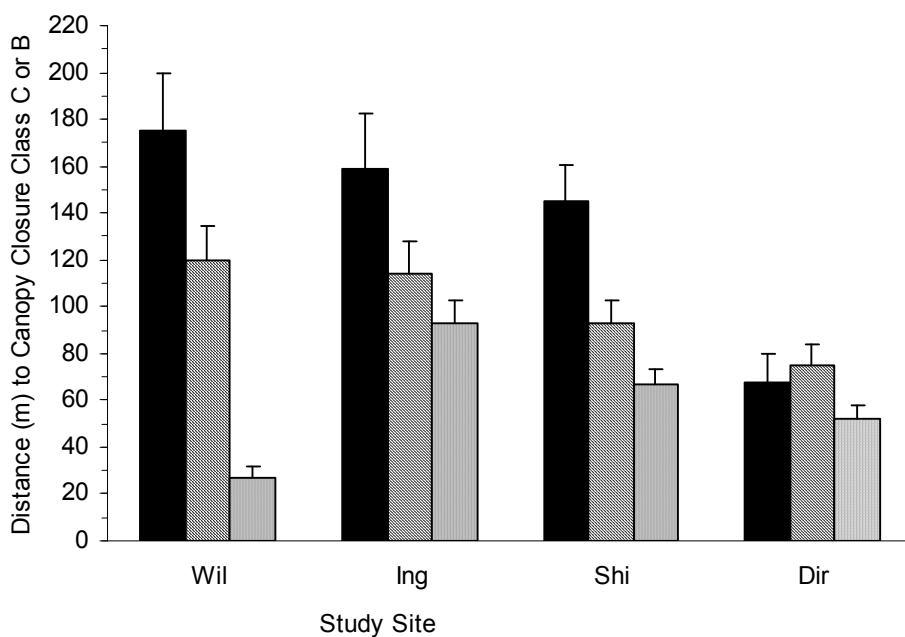
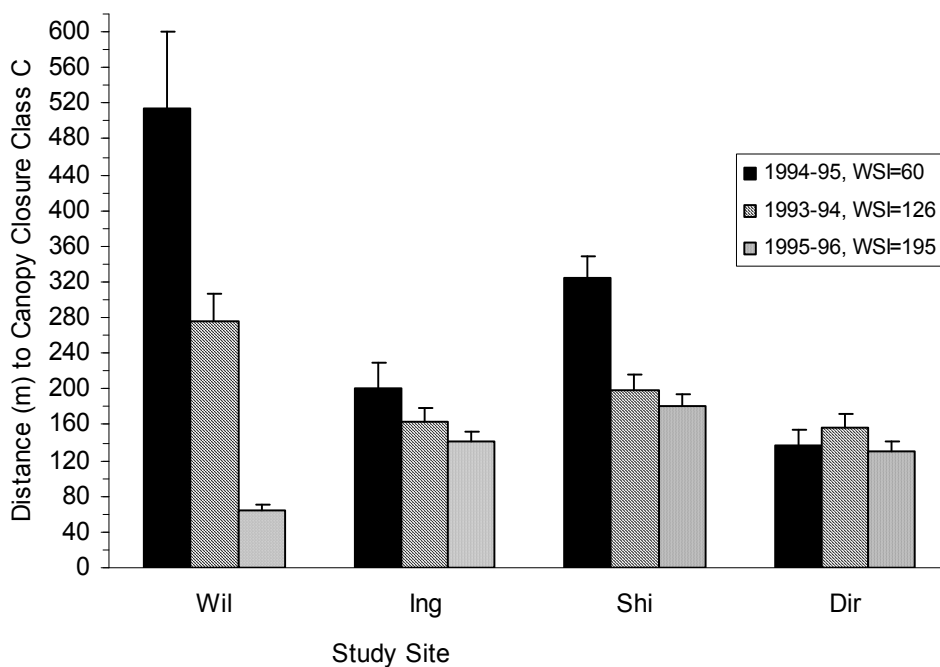


Figure 5. Mean nearest distance of radiocollared, female white-tailed deer to conifer stands classified as canopy closure (cc) Class C (top,  $\geq 70\%$  cc) and to those classified as canopy closure Class B ( $40\% < x < 70\%$  cc) or C (bottom) on the Willow Lake (Wil), Inguadona Lake (Ing), Shingle Mill Lake (Shi), and Dirty Nose Lake (Dir) study sites, north-central Minnesota, winters 1993-94, 1994-95, and 1995-96 (winter severity indexes [WSI, see text for definition] appear within the legend).

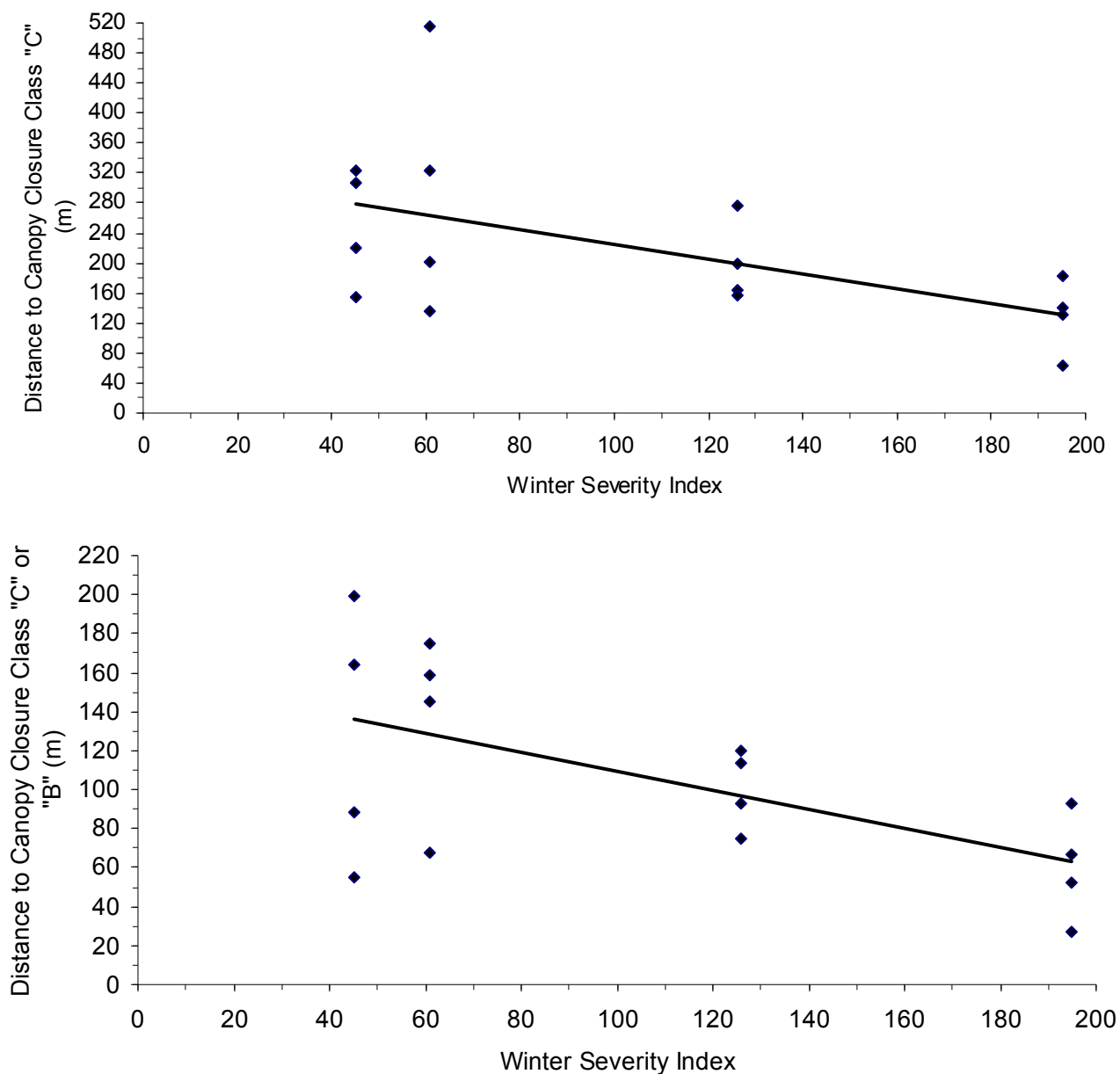


Figure 6. Relationship of the annual maximum winter severity index (see text for definition) to the mean nearest distance of radiocollared, female white-tailed deer to conifer stands classified as canopy closure (cc) Class C (top,  $\geq 70\%$  cc;  $r^2 = 0.30$ ,  $y = 323 - 0.978x$ ,  $P = 0.028$ ) and to those classified as canopy closure Class B ( $40\% < x \leq 70\%$  cc) or C (bottom,  $r^2 = 0.35$ ,  $y = 158 - 0.488x$ ,  $P = 0.015$ ), all 4 study sites pooled, north-central Minnesota, winters 1993-94, 1994-95, 1995-96, and 2001-02.

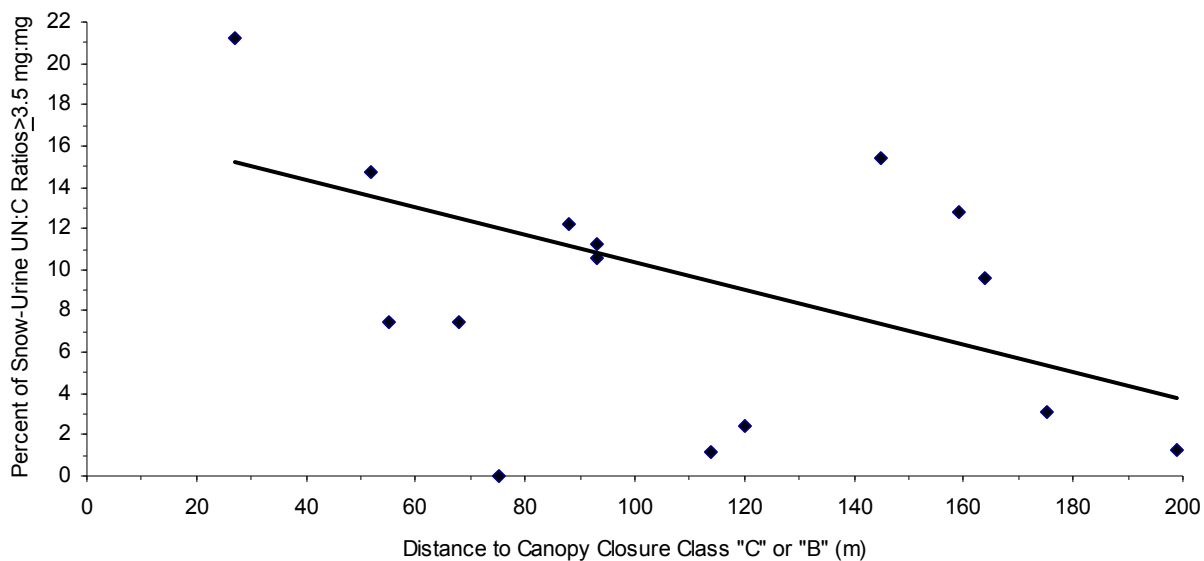


Figure 7. Relationship ( $r^2 = 0.19$ ,  $y = 134 - 2.790x$ ,  $P = 0.095$ ) of the mean nearest distance of radiocollared, female white-tailed deer to conifer stands classified as canopy closure (cc) Class C ( $\geq 70\%$  cc) or B ( $40\% < x \leq 70\%$  cc) to the percent of urine samples in snow (snow-urine) of white-tailed deer with urea nitrogen:creatinine (UN:C) ratios indicative of severe nutritional restriction ( $\geq 3.5$  mg:mg), all 4 study sites pooled, north-central Minnesota, winters 1993-94, 1994-95, 1995-96, and 2001-02.



## UNDERSTANDING DIEL WINTER MOVEMENTS OF NORTHERN WHITE-TAILED DEER<sup>1</sup>

Christopher O. Kochanny<sup>2</sup> and Glenn D. DelGiudice

### ABSTRACT

Northern deer (*Odocoileus* spp.) have evolved physiological, behavioral, and morphological adaptations for survival during winter. Among them is voluntary restriction in movement, which contributes to energy conservation and limits the impact of negative energy balance. To better understand the diel movements of white-tailed deer in winter, we deployed 14 global positioning system (GPS) collars on adult ( $\geq 1.5$ -year old) female deer during February–March 1999 and 2000. Collars collected 1 location per hour. A total of 10,329 ( $n = 11$ ) 1-h movement segments were used to calculate hourly diel movement distances and rates. Deer were relatively active 24 h/day with mean daily movements of  $2.9 \pm 0.13$  (SE) km and no difference ( $P = 0.45$ ) between mean hourly diurnal (0600–1759 h,  $153 \pm 7$  m) and nocturnal (1800–0559 h,  $142 \pm 12$  m) distances moved. Mean total daily diurnal and nocturnal distances moved ( $1.7 \pm 0.1$  vs.  $1.4 \pm 0.1$  km) were different ( $P = 0.02$ ). We observed no relations of ambient temperature or snow depth to mean movements during these 2 mild winters with minimal snow cover (mean weekly snow depths of 8–15 cm). A 70-kg doe requires an estimated 0.049 kcal/ m of energy to travel in  $\leq 18$  cm of snow and 0.110 kcal/ m (or 2.25 times more energy) to travel in 40 cm of snow. For GPS-collared deer in our study making estimated daily minimum movements of 2.9 km, the daily energy cost for travel alone in shallow snow ( $\leq 18$  cm) was 142 kcal, but would have been an estimated 319 kcal during a more severe winter with 40 cm of snow. Over an entire winter (1 Dec–31 Mar), the difference in cumulative energetic impact would have amounted to an estimated 21,417 kcal. The relatively low cost of movement during winters 1999 and 2000 would largely explain the relatively high activity of deer diurnally and nocturnally. Despite reasonably similar diurnal and nocturnal movements, deer may show temporal variation in the use of space and winter habitat, particularly relative to the nocturnal thermal benefits of conifer cover. Knowledge of the relation of winter cover on the landscape to the winter diel movements of deer is essential to a fuller understanding of their activity patterns and habitat requirements.

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## COMPARING GLOBAL POSITIONING SYSTEM AND VERY HIGH FREQUENCY TELEMETRY HOME RANGES OF WHITE-TAILED DEER<sup>1</sup>

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### ABSTRACT

Use of Global Positioning System (GPS) collars on free-ranging ungulates overcomes many limitations of conventional very high frequency (VHF) telemetry and offers a practical means of studying space use and home range estimation. To better understand winter home ranges of white-tailed deer (*Odocoileus virginianus*), we evaluated GPS collar performance, and compared GPS- and VHF-derived diurnal home ranges (for the same animals) and GPS-derived home range estimates for diurnal and nocturnal locations. Overall, the mean fix success rate of our GPS collars was 85% (range = 14–99%). Kernel density estimates of home range (using the 95% probability contour) derived from GPS and VHF locations were generally similar, as were GPS-derived diurnal and nocturnal home ranges. Overlap indices between GPS and VHF utilization distributions (UDs) ranged from 0.49 to 0.78 for the Volume of Intersection (VI) index and from 0.67 to 0.94 for Bhattacharyya's Affinity (BA); overlap indices for GPS-diurnal and nocturnal UD's ranged from 0.29 to 0.81 for VI and from 0.56 to 0.94 for BA. Despite similarities of home ranges estimated from GPS versus VHF locations and GPS-diurnal versus nocturnal locations, our data also indicate that differences may have important implications for studies focused on deer use of space, habitat, and resources at a finer scale.

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