

ECOLOGY AND POPULATION DYNAMICS OF BLACK BEARS IN MINNESOTA

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SUMMARY OF FINDINGS

During April 2009–March 2010, we monitored 45 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 fringe), and a site at the northwestern (NW) edge of the range. Hunting was the primary source (79%) 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 fringes of the bear range, at the NW study site followed by Camp Ripley, due largely to an abundance of oaks and hazelnut in these areas, as well as agricultural crops consumed by bears in the late summer–fall. Data from Global Positioning System (GPS)-radiocollars indicated that males in the NW made significant use of cropfields (corn, oats, sunflowers) from August–October (>30% of locations). Females in this area rarely used crops, but instead spent much of their time in shrublands. Continuation of this work will aim to explain this sex-related disparity in habitat use and predict whether further expansion of the bear range is possible.

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 (Figure 1). 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 (Figure 1).

These study sites differ enormously. The CNF is one of the most heavily hunted areas of the state, with large public (national, state, and county), heavily-roaded forests dominated by aspen (*Populus tremuloides*, *P. grandidentata*) of varying ages. Camp Ripley is unhunted, 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 unhunted, 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 site at the northwestern edge of the Minnesota bear range (henceforth NW; Figure 1). This area differs from the other 3 areas in a number of respects: (1) it is largely agricultural (including some cropfields, like corn, oats, and sunflowers, that bears consume; however, edible crops compose only ~2% of the landscape), (2) most of the land, including various small woodlots, is privately-owned, with some larger blocks of forest contained within MNDNR Wildlife Management Areas and a National Wildlife Refuge; (3) the bear range in this area appears to be expanding and bear numbers have been increasing, whereas most other parts of the bear range are stable or declining in bear numbers; and (4) 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 (including crop use) for bears in an agricultural fringe area; and
5. Predict range expansion of bears in northwestern Minnesota (not covered in this report).

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, and implanted (subcutaneously) a miniature heart monitoring device (developed for humans) that will record heart rate, body temperature, and activity throughout the year. 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 typically reported collared bears that they killed (but see Results).

We plotted GPS locations downloaded from collars on bears in the NW study site. We used a Geographic Information System (GIS) overlay to categorize the covertypes of GPS locations, and then grouped these into 4 broad categories. We calculated percent use of these types by month for each bear, and then obtained monthly averages among bears of each sex.

We conducted food sampling in various woodlands in the NW study site, representing all the principle forest types in that area. 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. We also collected and counted fruits from bushes with various ratings to eventually convert these to biomass estimates.

RESULTS AND DISCUSSION

Radiocollaring and Monitoring

Since 1981 we have handled >800 individual bears and radiocollared >500. As of April 2009, the start of the current year's work, we were monitoring 38 collared bears: 4 in the CNF, 11 at Camp Ripley, 3 in VNP, and 20 in the NW. We captured 7 more bears in the NW study site during May–July (3 males, 4 females), and collared them, 4 with GPS-collars. We also collared 11 bears during the winter months: 9 yearlings (4 that had been orphaned), 1 adult female found in a den, and 1 previously-collared female that had dispersed from Camp Ripley to CNF.

Most GPS collars used this year were “pods” (Telemetry Solutions, Concord, CA) that were bolted onto normal VHF collars; thus, if they failed prematurely (as we experienced to a high degree with another manufacture's GPS collars last year), we would not lose track of the bear. In fact, all of them did fail prematurely, so virtually no GPS data were obtained during September–October, the main period when bears consume crops. Therefore, a major objective of this study (to discern degree of crop use as part of habitat selection) could not be accomplished this year.

Mortality

Legal hunting has been the predominant cause of mortality among radiocollared bears from all study sites; 79% 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 and colorful eartags so hunters could more easily see them. However, the mortality rate on collared bears has remained high even though some hunters reported avoiding them.

This year (September–October 2008), hunters killed 3 collared NW bears, and we surmised that 2 others were likely killed by hunters based on the condition and location of collars that we found. Two other NW bears and 1 CNF bear were found dead during the winter. We could not ascertain the cause of death, but in 2 of these cases we suspected that the bears were shot and lost by hunters. Three other collared bears were lost between late August and denning. Possibly these travelled beyond our search area (likely for 1 of them), or their signals could not be heard during winter because they were in deep, excavated dens. However, potentially as many as 2 of them were also shot by hunters. Thus, 5–8 collared bears in the NW were killed or possibly killed by hunters, or 22–35% of the radioed sample. One other NW bear was found dead during the summer, due to unknown causes. The number of deaths of NW bears due to unknown causes has been disproportionately high compared to the other study sites (Table 1).

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).

Reproduction

We visited 10 dens of females with cubs during March, 2010. In most of the state, more births occur during odd-numbered years, due to somewhat synchronous reproduction and a 2-year reproductive cycle (Garshelis and Noyce 2008). However, among collared bears in the NW, 6 had cubs and only 2 had yearlings (litters born in 2009), suggesting a different pattern than the remainder of the state.

Bears in the NW also seem to have a high reproductive rate, possibly the highest among our 4 study sites (Table 2). Litter sizes appeared to be highest in the NW (Tables 2–6), although this was influenced by a few large litters (4 or 5 cubs) and a small sample size. Among females 4 years or older, more than half were accompanied by cubs each year in the NW. With a 2-year reproductive cycle, the maximum proportion with cubs should be 0.5, but sampling variation could lead to a higher value (Table 2). The reproductive rate (cubs/female 4+ years old), which combines litter size, litter frequency, and age of first reproduction into one parameter, was higher in the NW than at Camp Ripley, which in turn was higher than the CNF and VNP (Table 2). The high reproduction in the NW was likely due to abundant foods: despite a very fragmented landscape, oaks, hazelnuts (*Corylus americana*, *C. rostrata*), and agricultural crops are plentiful.

Average sex ratio of cubs shortly after birth was slightly, but consistently male-biased (pooled average across all areas = 52% male, $n = 626$ 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 was 21% for all areas pooled, but differences were observed among areas (range of means = 18–28%), with apparently the poorest survival in VNP (Tables 3–6). Across all areas, the mortality rate of male cubs was significantly (1.6x) higher than that of females ($\chi^2 = 6.7$, $P = 0.001$), however, the predominant cause of cub mortality in Minnesota is not known.

Habitat Use of NW Bears

The landscape in the NW study site is about 20% forested. Both males and females in this region used forested lands to a high degree (40–60%) during May–July (Figure 2). Beginning in August, males made heavy use of croplands. All of the GPS-collared males used some agricultural crops (corn, oats, sunflowers), although the extent of use varied considerably by individual. In a few cases, bears used cornfields in spring and early summer that were not harvested the previous fall (Figure 3). Male use of croplands increased through October (40% use), after which they began to den. We have not yet learned why females rarely used croplands (Figure 2), but we expect it was related to avoidance of males. Instead, females made more use of shrubby areas. We have been walking into sites of bear locations to identify the attraction of these shrublands, but have no definitive results to report, as yet. High use of shrublands or wetlands in November (Figures 2, 3) represent den sites.

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 NW study site, although for the past 2 years our data collection there has been limited by faulty GPS collars. We are hopeful that these issues have been solved for the coming year. In addition to gaining information from radiocollars, we have been and will continue to interview farmers to collect additional data on bear use of crops. This will yield an historical perspective on crop use, and provide insights into specific varieties of corn and sunflowers used by bears. Moreover, in the coming year we plan to obtain hair samples from hunter-killed bears in the NW for stable isotope analysis to ascertain the relative importance of corn in the diet, for males and females living in different parts of the

study area. Ultimately we aim to create a habitat suitability map and thereby predict how far the bear population is likely to expand in this part of the state.

ACKNOWLEDGMENTS

We thank the collaborators in this study: graduate student Mark Ditmer in the NW, Brian Dirks at Camp Ripley, Dr. Paul Iaizzo at the University of Minnesota, and Dr. Tim Laske at Medtronic, Inc. Morgan Elfelt assisted with fieldwork in the NW.

LITERATURE CITED

Garshelis, D.L., and K.V. Noyce. 2008. Seeing the world through the nose of a bear — Diversity of foods fosters behavioral and demographic stability. Pages 139–163 in T. Fulbright and D. Hewitt, editors, *Frontiers in Wildlife Science: Linking Ecological Theory and Management Applications*. CRC Press, Boca Raton, FL.

Table 1. Causes of mortality of radiocollared black bears ≥ 1 year old from the Chippewa National Forest (CNF), Camp Ripley, Voyageurs National Park (VNP), and northwestern (NW) Minnesota, 1981–2009. 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	7	254
Likely shot by hunter ^a	8	1	0	2	11
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	4	2	0	3	9
Total deaths	283	27	21	13	344

^a Lost track of during the hunting season, or collar seemingly removed by a hunter.

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 (1997–2010), CNF (1981–2010), Camp Ripley (1991–2010), NW (2008–2010) ($n = 4+$ year-old female-years of observation).

Age of female	VNP ($n = 62$)			CNF ($n = 409$)			Camp Ripley ($n = 55$)			NW ($n = 23$)		
	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	0.55	2.0	0.27	0.84	2.3	0.37	1.00	2.2	0.46	1.25	2.5	0.50
7–25 yrs	1.15	2.6	0.44	1.34	2.8	0.48	1.48	2.7	0.56	2.00	3.3	0.60
4–25 yrs	0.92	2.5	0.37	1.15	2.6	0.44	1.24	2.4	0.51	1.65	2.9	0.57

Table 3. Black bear cubs examined in dens of radiocollared mothers in or near the Chippewa National Forest during March, 1982–2010. 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 ^a
1982	4	12	3.0	67%	25%
1983	7	17	2.4	65%	15%
1984	6	16	2.7	80%	0%
1985	9	22	2.4	38%	31%
1986	11	27	2.5	48%	17%
1987	5	15	3.0	40%	8%
1988	15	37	2.5	65%	10%
1989	9	22	2.4	59%	0%
1990	10	23	2.3	52%	20%
1991	8	20	2.5	45%	25%
1992	10	25	2.5	48%	25%
1993	9	23	2.6	57%	19%
1994	7	17	2.4	41%	29%
1995	13	38	2.9	47%	14%
1996	5	12	2.4	25%	25%
1997	9	27	3.0	48%	23%
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%	33%
2010	1	4	4.0	100%	—
Overall	175	459	2.6	53%	18%

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

Table 4. Black bear cubs examined in dens of radiocollared mothers in Camp Ripley Military Reserve during March, 1992–2010.

Year	Litters checked	No. of cubs	Mean cubs/litter	% Male cubs	Mortality after 1 yr ^a
1992	1	3	3.0	67%	0%
1993	3	7	2.3	57%	43%
1994	1	1	1.0	100%	—
1995	1	2	2.0	50%	0%
1996	0	0	—	—	—
1997	1	3	3.0	100%	33%
1998	0	0	—	—	—
1999	2	5	2.5	60%	20%
2000	1	2	2.0	0%	0%
2001	1	3	3.0	0%	33%
2002	0	0	—	—	—
2003	3	8	2.7	63%	33%
2004	1	2	2.0	50%	—
2005	3	6	2.0	33%	33%
2006	2	5	2.5	60%	—
2007	3	7	2.3	43%	0%
2008	2	5	2.5	60%	0%
2009	3	7	2.3	29%	29%
2010	2	4	2.0	100%	—
Overall	30	70	2.3	53%	21%

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

Table 5. Black bear cubs examined in dens of radiocollared mothers in Voyageurs National Park during March, 1999–2010. All adult collared females were killed by hunters in fall 2007, so there were no reproductive data for 2008–2009.

Year	Litters checked	No. of cubs	Mean cubs/litter	% Male cubs	Mortality after 1 yr ^a
1999	5	8	1.6	63%	20%
2000	2	5	2.5	60%	80%
2001	3	4	1.3	50%	75%
2002	0	0	—	—	—
2003	5	13	2.6	54%	8%
2004	0	0	—	—	—
2005	5	13	2.6	46%	20%
2006	1	2	2.0	50%	0%
2007	3	9	3.0	44%	—
2008	0				
2009	0				
2010	1	2	2.0	50%	
Overall	25	56	2.2	52%	28%

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

Table 6. Black bear cubs examined in dens of radiocollared mothers in Northwestern Minnesota during March, 2007–2010.

Year	Litters checked	No. of cubs	Mean cubs/litter	% Male cubs	Mortality after 1 yr ^a
2007	2	6	3.0	33%	100% ^b
2008	5	15	3.0	67%	22%
2009	1	3	3.0	33%	33%
2010	6	17	2.8	41%	
Overall	14	41	2.9	47%	25%^c

^a Cubs that were absent from their mother's den as yearlings were considered dead.

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

^c Excludes the total loss of the single 5-cub litter (which was not within the designated study area).

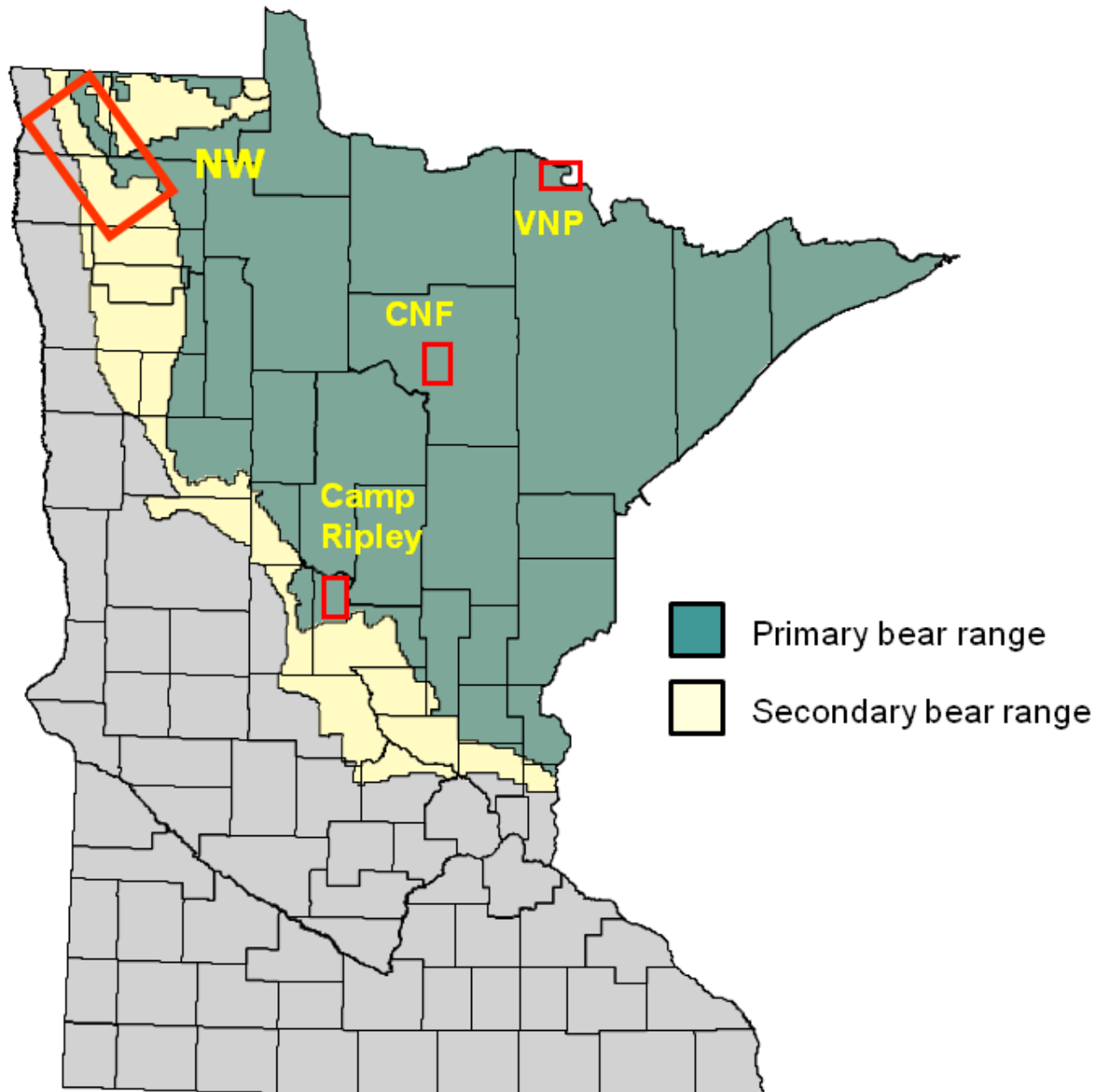


Figure 1. Location of 4 study sites within Minnesota's bear range: CNF (Chippewa National Forest, central bear range; 1981–2010); VNP (Voyageurs National Park, northern fringe of range; 1997–2010); Camp Ripley Military Reserve (near southern edge of range; 1991–2010); NW (northwestern fringe of range; 2007–2010).

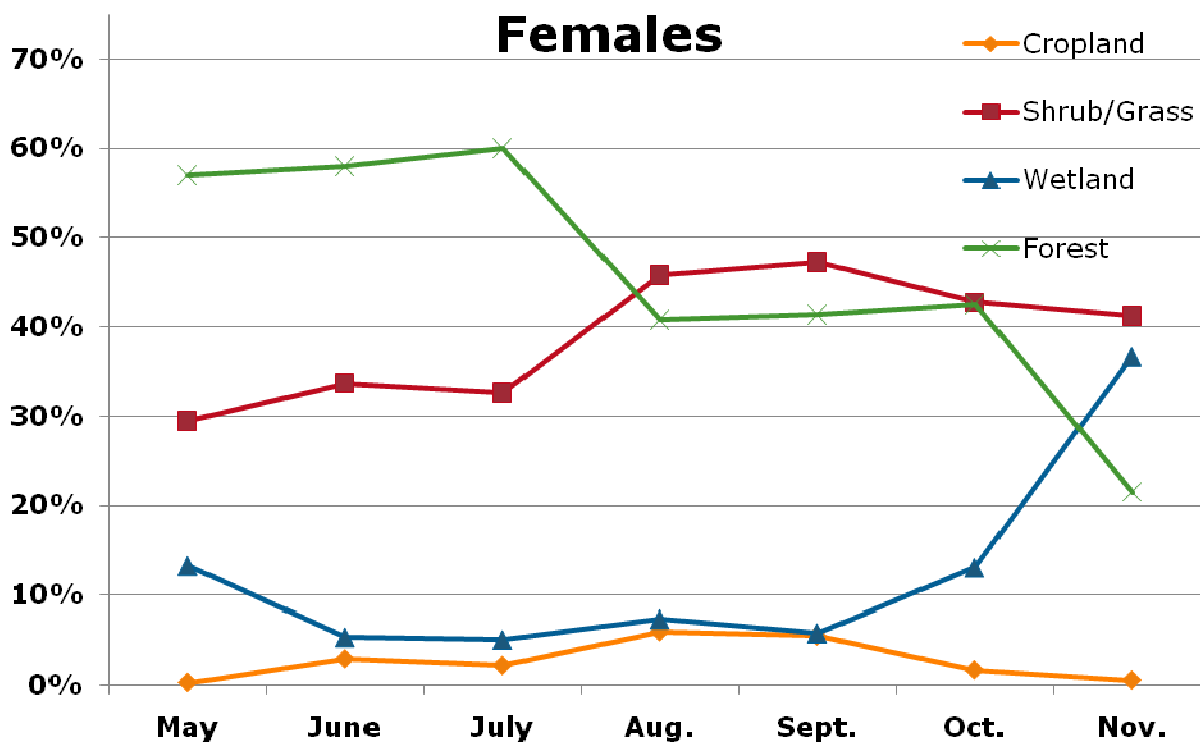
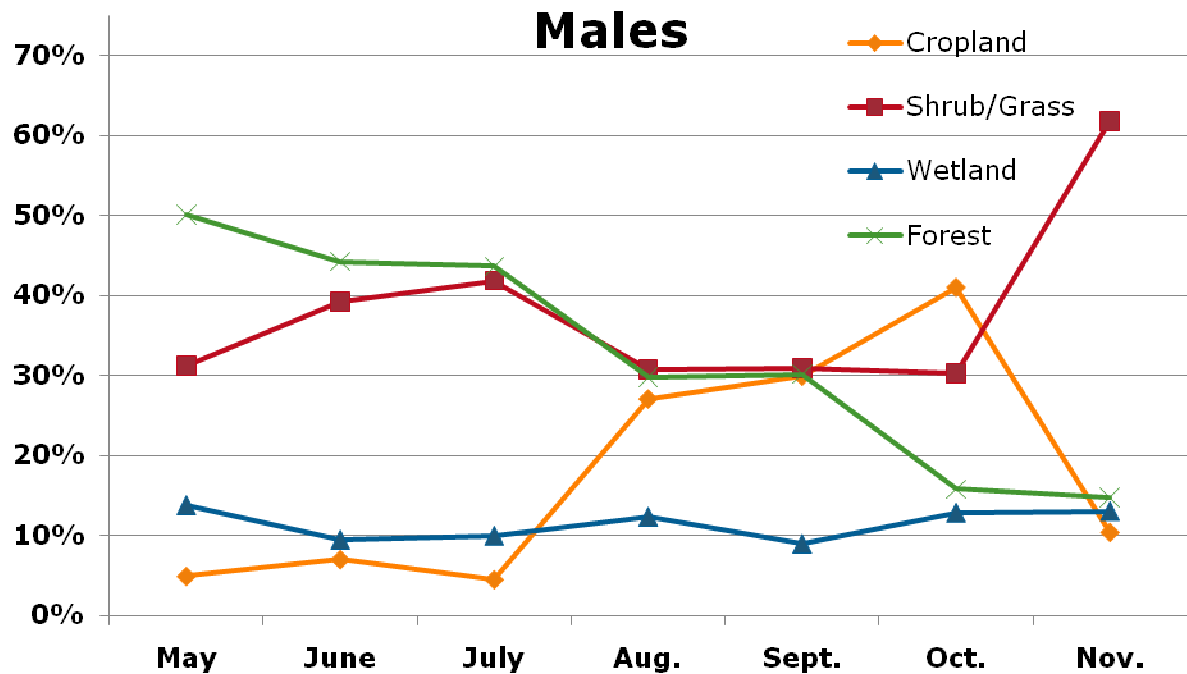


Figure 2. Trends in habitat use of black bears in Northwestern Minnesota, based on locations from GPS-radiocollars.

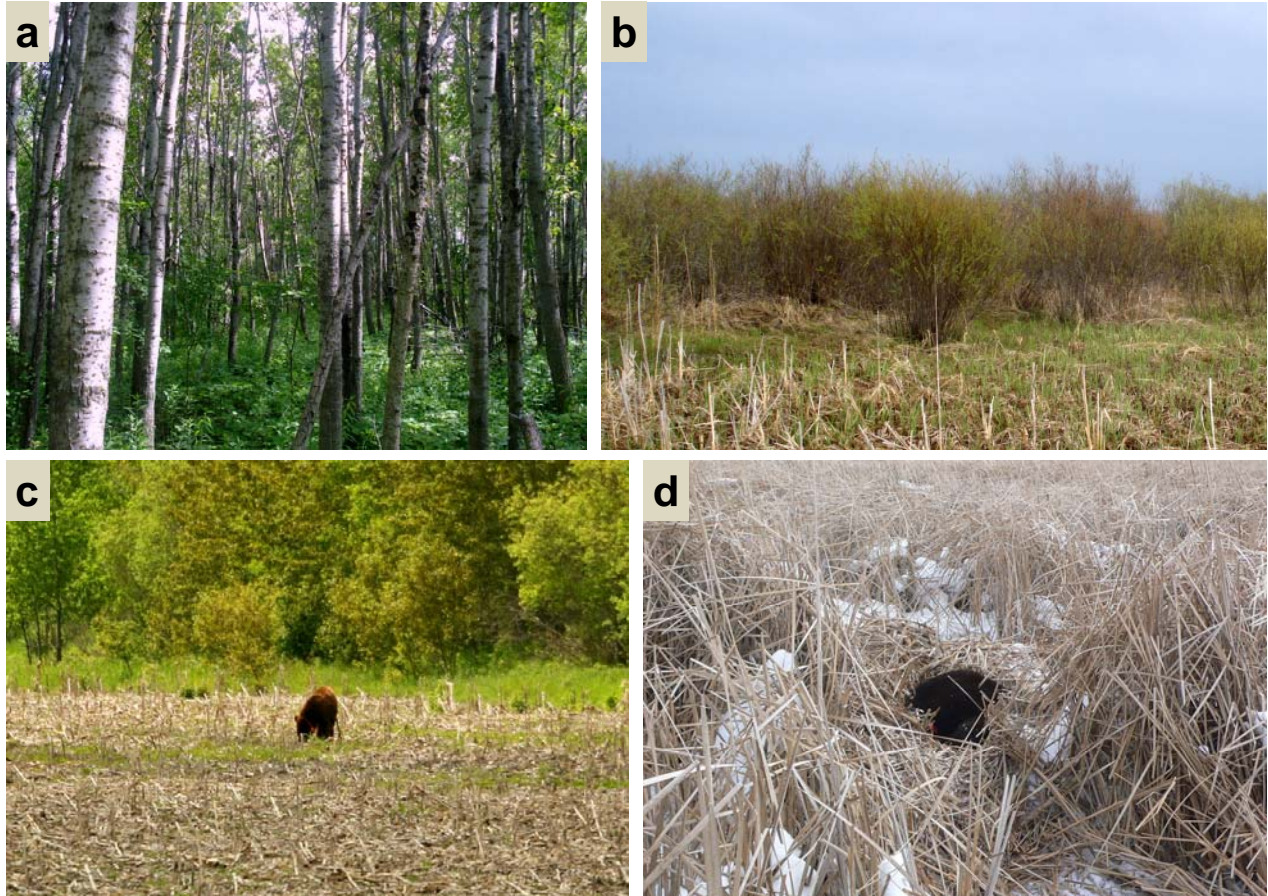


Figure 3. Examples of the main habitat types used by bears in the NW study site. (a) aspen forest; (b) lowland shrub, (c) cornfield (in this case, an unplowed field from the previous year, with a bear foraging on remnant cobs in June), (d) wetland, cattail swamp (used by denning bears). Photos: (a,b,c) M. Ditmer and M. Elfelt; (d) D. Garshelis.

REPRODUCTIVE ECOLOGY OF FISHER AND MARTEN IN MINNESOTA

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SUMMARY OF FINDINGS

As part of a larger project on *Martes* ecology in Minnesota, we began monitoring reproductive success of radiocollared fisher (*Martes pennanti*) and marten (*Martes americana*) during spring 2009. Including the pilot year of the study, we have captured 86 martens (44F, 42M) and 45 fishers (25F, 20M). A total of 28 female martens and 21 female fishers have been available for monitoring during the kit-rearing season. However, age information is not yet available for all animals, and this year's den and litter searches are ongoing. To date, we have confirmed presence of kits for 10 female martens, 8 of which we have obtained litter counts (average minimum litter size = 3.4). In addition, we have confirmed litters for 14 female fishers, all of which we obtained litter counts (average litter = 2.7). Initial data suggests that pregnancy rates and litter sizes are smaller for 2 year old fishers compared to older adults. Of 13 marten natal or maternal dens we have located, 54% have been in tree cavities, while 46% have been underground. All of the natal or maternal dens we have located for fisher prior to June 1 (n=16) have been in tree cavities, primarily large-diameter aspen. One fisher maternal den located in late June was in a hollow log on the ground. Fisher kits appear to be born during the last 2 weeks of March, while marten parturition appears to be centered on the last 2 weeks of April.

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 areas as far west as Roseau County. Limited information suggests they occurred as far south as Crow Wing County and as far southwest as Polk County. As a result of unregulated harvest, martens 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 martens 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, including southeast Minnesota (Swanson et al. 1945, Balser and Longley 1966). The impacts of unregulated 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 lower, though perhaps increasing, in the Red River Valley (western Minnesota) and the highly fragmented transitional forests in central Minnesota. Peak harvest levels have been near 4,000 and 3,500 for marten and fisher, respectively. However, due to apparent multi-year population declines for both species, harvest seasons were reduced from 16 days to 9 days for the past 3 seasons, with harvests averaging 2,000 and 1,400 for marten and fisher, respectively.

While both species appear to have naturally re-colonized a significant portion of their historic range, Minnesota-specific information on reproductive ecology is limited to carcass

(corpora lutea, placental scar) data collected from harvested animals primarily from 1985-90 (Kuehn 1989, Minnesota DNR unpublished data). Reproductive data is also available from other geographic areas, but questions remain on the accuracy of various methods to assess reproduction, and the amount of spatial and temporal variation in reproductive parameters. Minnesota-specific data on structures and sites used by fisher for natal and maternal dens is also lacking.

Martes pregnancy rate and litter size data are generally quantified from 1 of 4 methods: counts of corpora lutea (CL) in ovaries; counts of blastocysts (BC) in uteri; placental scar (PS) counts; or direct observation of litter size (Gilbert 1987). Assuming both species are induced ovulators (but see Cherepak and Connor 1992, Frost et al. 1997), CL counts should accurately reflect copulation and ovulation rates, but all CL persist even if only 1 ovum is fertilized. Blastocyst counts reflect the number of fertilized ova, but not all BC may implant in the uterus and develop, and BC are often destroyed in poorly preserved carcasses. Hence, these 2 measures may not only overestimate litter size for parous females, but may also overestimate parturition rate (i.e., females may ovulate, 1 or more ova become fertilized, yet they fail to ultimately den and give birth). Placental scars, formed last in the reproductive process, would seem the most reliable carcass-based estimate of parturition rate and litter size. However, several authors (Gilbert 1987, Payne 1982, Strickland and Douglas 1987) have suggested that PS may not always persist long enough in mustelids to be detected during the harvest season when carcasses are easily collected, and PS can persist even if fetuses are resorbed (Conaway 1955). Nevertheless, PS have been reliably used in the past (e.g., Coulter 1966, Crowley et al. 1990), though others have noted that reliable results may only be obtainable when doing microscopic analysis of fresh and properly preserved/prepared uteri.

In spite of these concerns, average litter size estimates from reproductive organs do not appear to be substantially biased. Strickland and Douglas (1987), summarizing data from 136 captive marten litters, computed average litter size of 2.9 for marten. This is within the range of average litter sizes reported from ovary or uterine analysis (~ 2.5 – 3.5; Strickland et al. 1982, Strickland and Douglas 1987, Flynn and Schumacher 1995, 2009, Aune and Schladweiler 1997, MN DNR unpublished data). For fisher, the same appears to be true, with an average litter size of 2.8 from 60 captive fisher litters (reviewed in Strickland and Douglas 1987) and 19 wild litters (York 1996), which compares favorably to estimates based on reproductive organs (2.7 – 3.9 (CL), 2.7 – 3.2 (BC), and 2.5 – 2.9 (PC); review in Powell 1993).

Of greater concern is the possibility that ovary, and to lesser degree uterine analyses might consistently overestimate parturition rate, thereby also underestimating annual variability in parturition rates. Various indications of pregnancy may be detected, though not all of those females may den and produce kits in spring. This might occur, for example, if ova are not fertilized following copulation or females experience nutritional stress during the period of embryonic diapause (Arthur and Krohn 1991). Overall, CL counts have generally yielded ovulation rates for fisher of $\geq 95\%$ (Shea et al. 1985, Douglas and Strickland 1987, Paragi 1990, Crowley et al. 1990, MN DNR unpublished data), while more 'direct' estimates of average parturition rate from radio-marked animals have been lower (46-75%; Crowley et al. 1990; Arthur and Krohn 1991; Paragi 1990; Paragi et al. 1994, York 1996, Truex et al. 1998, Higley and Mathews 2009), and often highly variable. Conversely, Kuehn (1989) did not detect changes in pregnancy rate (from CL analysis) during a 64% decline in snowshoe hare indices in Minnesota.

For marten, several largely ovarian-based estimates of annual pregnancy rate have often been in the range of 80-90% (Archibald and Jessup 1984, Strickland and Douglas 1987, Aune and Schladweiler 1997, Flynn and Schumacher 1994, Fortin and Cantin 2004, MN DNR unpublished data). However, like for fisher, several marten studies have documented (also based largely on CL counts) lower or more variable pregnancy rates (Thompson and Colgan 1987, Aune and Schladweiler 1997, Strickland and Douglas 1987, Flynn and Schumacher 2009), perhaps a result of fluctuations in prey abundance (Hawley and Newby 1957, Weckwerth and Hawley 1962, Strickland 1981, Strickland and Douglas 1987, Thompson and Colgan 1987, Fryxell et al. 1999, Flynn and Schumacher 2009). We are aware of direct field-based estimates

of parturition rate from radio-marked marten in only one state (Maine). Pooling samples across 4 years, the proportion of lactating adult females was 75, 81, and 92% for their 3 different study areas (Phillips 1994, Payer 1999), similar to much of the CL-based pregnancy studies.

Understanding reproductive ecology of these species also necessitates gathering information on natal and maternal den structures and selection of den sites. Natal dens are the structures where kits are born, whereas maternal dens are sites used subsequently by the female with her dependent young. Although data is absent for Minnesota, nearly all reported fisher natal dens have been in cavities of large-diameter trees or snags (Leonard 1986, Paragi et al. 1996, Powell et al. 1997, Truex et al. 1998). In northern studies, the majority of fisher natal dens have been in large diameter aspens (*Populus* spp), and females may use up to 3 or more different maternal dens (Powell et al. 2003, Higley and Mathews 2009). Marten natal and maternal 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). Though not further discussed here, the literature is also voluminous with documentation of the importance of tree cavities, large downed logs, and other forest 'structure' for fisher and marten resting sites (see Powell et al. 2003 for a review). Given the continuing pressure to maximize fiber production from forests (i.e., short forest rotation, biomass harvesting, etc), the forest structural attributes critical to fisher and marten could become limiting in the future, if not already. Hence, acquiring Minnesota-specific information is critical to better inform forest management activities.

As part of a larger project on *Martes* (Erb et al. 2009), we began efforts to better describe the reproductive ecology of fisher and marten in Minnesota, specifically: 1) denning chronology; 2) structures used for natal and maternal dens; 3) vegetative characteristics in the area surrounding natal and maternal dens; 4) field-based estimates of pregnancy rate, litter size, and where possible, kit survival; and 5) the influence of age, food habits, prey fluctuations, home range habitat quality, and winter severity on reproductive success. After initial evaluation of field methods during the pilot year of the study, spring 2009 marked the beginning of full-scale research activities. Herein we present basic information on field methods, though we only report preliminary findings related to items 2 and 4. We defer a more complete evaluation of results until additional data is collected or additional analysis is completed.

STUDY AREA

Marten research is focused on 1 study area located in northeastern Minnesota (Figure 1; Area 1), though an occasional marten is captured and radiocollared in Area 2 (Figure 1). Area 1 (~ 700 km²) is composed of approximately 69% mixed coniferous-deciduous 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 < 1% deciduous forest. Area 1 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 basic summaries, we do not discuss other aspects of that project in this report. Area 2 (1075 km²), 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 typically baited with deer (*Odocoileus virginianus*) or beaver (*Castor canadensis*) meat, and we placed commercial lure 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). Fishers 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. Martens were ear-tagged with a monel #1 tag (National Band and Tag Co., Newport, KY) in each ear.

During processing, we placed animals on chemical hand warmers or heating pads connected to a power inverter and 12 volt battery. Portable shelters and 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 assist with determining which females would likely produce kits, blood samples were drawn when possible to measure serum progesterone level in each animal (Frost et al. 1997). All blood samples were sent to the University of Minnesota Veterinary Diagnostics Lab for progesterone analysis. Antibiotics were administered subcutaneously to all animals prior to release.

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). Since the pilot year, we have primarily deployed ATS M1940 (~ 43 g) or Sirtrack TVC-162 collars (~ 45 g) on fisher. The majority of martens have been fitted with Holohil MI-2 collars (~ 31 g). We retrofitted each collar with a temperature data logger, in part to assist with determination of exact parturition date.

We primarily used ground tracking to locate den sites, but also deployed remotely-activated cameras (Reconyx PC-85 or RC-55, Reconyx, Inc, Holmen, WI) at suspected sites to monitor female activity. However, we considered a female to have given birth only if kits were confirmed via sound or video/camera, or if other reliable evidence (e.g., obvious lactation, placental scars, or kit bite marks on collar) was obtained when an animal was subsequently handled as a mortality or recapture. Litter size was ascertained via visual confirmation in most cases, though we also utilized placental scar counts on any females that died during summer or fall, and for which other methods failed to produce a count. To confirm or count kits at dens located in tree cavities, we used an MVC2120-WP color video camera (Micro Video Products, Bobcaygeon, Ontario), attached to a telescoping pole if necessary, and connected to a laptop computer. Underground dens were examined when possible using the same video probe attached to a flexible rod. Dens were only examined when the radio-marked female was not present. If video inspection equipment did not work at a particular den structure, we deployed remote cameras in an effort to obtain pictures of kits when they emerged or were moved by the female (Jones et al. 1997).

When a natal or maternal den was confirmed, we recorded den location (above/on/below-ground) as well as various location-specific details (e.g., tree species, log/tree diameter, burrow entrance attributes, etc). We note that since birth is never observed, and kits may be moved to new dens within days following birth, distinguishing natal dens from maternal dens can rarely be done with certainty. Hence, while we report our best assessment of den

type, our focus is ultimately on determining whether initial dens (be they natal or maternal) used early in the kit-rearing period (e.g., prior to June 1) are structurally different than dens used as kits get larger and more mobile. Hence, we organize our tabular reporting on the date at which the den was first documented to be in use.

We will also be collecting more detailed information on vegetative characteristics of the site surrounding each den structure, with a goal of not only developing a biologically meaningful den site selection model, but also to do so using methods and metrics that will be 'transferable' to long-term habitat monitoring over large areas using existing forest sampling data (e.g., see Zielinski et al. 2006). Following the United States Forest Service's Forest Inventory and Analysis (FIA) protocol, we will quantify vegetative characteristics in a 1-acre (120' radius) area surrounding the den structure by sampling in 4 circular subplots, each being 0.04-acre (24-ft radius) in size. One subplot will be centered on the den structure, with the other 3 subplots centered 120 feet from the den at 360°, 120°, and 240°. Within each subplot, 3 24' coarse woody debris sampling transects are established, originating from the subplot center, and oriented at 30°, 150°, and 270°. Deviating from FIA protocol, we also establish 3 (not 1, as with FIA) 0.003-acre (6.8 ft radius) circular micro-plots for estimating sapling density, each micro-plot situated at the end of the 3 coarse woody debris sampling transects. Details of vegetation sampling methods within each subplot will be outlined in subsequent years as results become available. Herein, we simply note that we will collect quantitative data on: 1) mean DBH and basal area of live trees, overall and by species; 2) % overhead (angular) canopy; 3) sapling density; 4) understory cover density; 5) density and volume of snags and stumps; and 6) volume of coarse woody debris; 7) distance to improved road; and 8) distance to water. Canopy structure will also be categorized based on number and distribution of canopy layers.

To better understand any observed fluctuations in reproductive parameters, we are also collecting data on factors that may influence reproductive success, including winter severity and prey fluctuations. In each study area, a temperature monitor was placed in each of 6 cover types. Each sensor records temperature every 30 minutes, and was placed on the north-facing side of a tree situated along a transect that we used for recording cover-type specific snow information. In addition to monitoring temperature, at each of 3 locations along a transect, and repeated once within each 10-day interval (1 Dec – 1 Apr), we recorded snow depth and 2 measures of snow compaction. Two snow compaction tools were constructed using PVC pipe, one each with an end-cap similar in diameter to a typical marten and fisher track in the snow. Each pipe length was then adjusted to ensure the pipe-specific load (g/cm^2) was similar to marten and fisher foot-load measures (females) reported by Krohn et al. (2004). Depth of snow compaction was recorded by dropping each load tool from 1 in. above snow level and measuring compaction depth.

Prey sampling transects have also been established in both study areas. Prey sampling is being conducted primarily to document between-area differences in prey abundance, annual within-area fluctuations in prey, and ultimately to assess whether fisher or marten habitat use, diet, survival, or reproductive success is correlated with prey dynamics. Prey-sampling transects ($n \approx 125$ in each study area) consist of 10 sampling locations (2 parallel lines of 5 stations) spaced 20m apart, with transects distributed in 6 cover types throughout each study area. Transects are generally oriented perpendicular to roads or trails, with the first plot 30m off the trail. In spring, we count snowshoe hare (*Lepus americanus*) pellets in a 1- m^2 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 (Aarhus-Ward 2009). 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 (e.g., 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 86 martens (44F, 42M) and 45 fishers (25F, 20M) have been captured. Herein we provide a basic summary of data collected to date on den structures, pregnancy status, and litter size. Because tooth aging has not yet been completed for all animals, and some yet-to-be-aged females may be only 1 year of age (i.e., not capable of producing kits), we present results only for animals known to be ≥ 2 years of age during spring den visits, or those of unknown age but for which we have confirmed parturition at the time of this writing (i.e., until age is known, we do not include animals that we have confirmed to be nulliparous).

Treating females that were alive during multiple parturition periods as independent units, and excluding females known to be 1 year of age during the parturition period, a total of 28 female martens have been available for monitoring during the kit-rearing season. However, at the time of this writing, we have only confirmed age and reproductive status for 6 females, and have confirmed litters (but are awaiting age data) from 5 additional females (Table 1). Two additional females for which we are awaiting age results were confirmed to be nulliparous, and we were unable to confirm birth status for 2 females in 2009. Of the remaining 13 females, den monitoring efforts are ongoing, and while we suspect many do not have kits, we also expect age data to confirm many are 1 year of age. Because it has been more difficult to inspect marten natal dens with video equipment, we have had to rely more on remote cameras to obtain litter information when kits are moved by the female, or when they are older and more mobile. Hence, many estimates of marten litter size are reported as minimums. Acknowledging this, average size of 8 litters confirmed to date is 3.4. Based on initial data, it appears marten kits are typically born in mid- to late-April. Given the timing of our marten capture (blood-drawing) operations (i.e., mid-Dec. through early Feb.), preliminary results indicate that marten progesterone levels have not sufficiently elevated in pregnant animals at that time to allow us to confirm mid-winter pregnancy status.

A total of 13 marten natal or maternal dens have been located to date (Table 2). We have not confirmed sufficient numbers of dens used later in the kit rearing process (after 1 June) to evaluate whether the type of den structures used changes as kits get older. Based on 11 marten natal/maternal dens confirmed prior to June 1 of each year, 64% have been in tree cavities, while 36% have been in underground tunnels (Table 2). The only 2 maternal dens we have confirmed after 1 June have been in underground burrows (Table 2).

Similar to marten, we treat female fishers that were alive during multiple parturition periods as independent units. Excluding individuals known to be 1 year of age during the parturition period, a total of 21 female fishers have been available for monitoring during the kit-rearing season. At the time of this writing, we have confirmed age and reproductive status for 15 females, and have confirmed litters (but are awaiting age data) from 3 additional females (Table 3). The remaining 3 females were confirmed to be nulliparous, and we are awaiting age results. We have obtained litter data for 14 fisher litters, with an average litter size of 2.7. Sample sizes are small, but there is some indication that average litter size for 2 year olds is lower than older females (2.5 versus 2.9). There is also some indication that birth rates are lower for 2 year olds compared to older females (67% versus 78%), a difference that would be further magnified were it not for the 2 (apparently) 'failed' reproductive seasons by the same 7+ year old female (i.e., F09-354; Table 3). Based on data collected to date, it appears fisher kits are typically born in mid- to late-March, or ~ 1 month earlier than marten kits. Perhaps owing to earlier parturition, as well as apparently longer active gestation (Powell et al. 2003), it does appear that the fisher progesterone levels are sufficiently elevated in pregnant females at the time of our winter capture operations (i.e., mid-Dec. through mid-March) to allow accurate assessment of mid-winter pregnancy status using hormone profiles developed in Maine (Frost et al. 1999).

A total of 17 fisher natal or maternal dens have been located to date (Table 4). We have not confirmed sufficient numbers of dens used later in the kit rearing process (after 1 June) to

evaluate whether the type of den structures they use changes as kits get older. Based on 16 fisher natal/maternal dens confirmed prior to 1 June of each year, 100% have been in tree cavities, primarily large-diameter aspen. Pooling all tree species, average DBH for natal and maternal den trees is ~ 22 inches. The only fisher maternal den confirmed after 1 June was in a large diameter hollow log on the ground (Table 4).

ACKNOWLEDGEMENTS

We thank volunteer Richard Nelles for his dedicated assistance with trapping and den monitoring efforts. We also acknowledge Carolin Humpal for her ongoing assistance with various aspects of the project, including trapping, tooth aging, and prey sampling. We also thank pilots Al Buchert and Don Murray for aerial telemetry efforts, and are grateful to staff from the U.S. Forest Service for their logistical assistance on various aspects of the project.

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Table 1. Parturition status and litter size for radiocollared female marten¹ in Minnesota.

ID	Year	Age	Litter	Litter size
M09-280	2010	2	Yes	3
M09-264	2009	3	No	
M08-140	2008	9	Yes	
M09-286	2009	9	Yes	>=3
M08-140	2009	10	Yes	>=2
M09-286	2010	10	Yes	4
M09-247	2009		Yes	4
M09-262	2009		Yes	
M09-254	2010		Yes	>=3
M09-262	2010		Yes	4
M09-237	2010		Yes	4

¹ Excludes unknown-aged nulliparous females, and all 1 year olds.

Table 2. Natal and maternal den structures used by radiocollared female marten in Minnesota.

ID	Year	Confirmed	Den type	Den structure	Den details
M09-254	2010	4/19	natal	Tree cavity	15.9" dbh live red maple
M09-237	2010	4/19	natal	Tree cavity	16.8" dbh live tamarack
M08-140	2009	4/21	natal	underground burrow	rock-laden soil
M09-280	2010	4/28	natal	underground burrow	rock-laden soil
M08-140	2008	4/30	natal	underground burrow	rock-laden soil
M09-286	2010	5/7	natal	Tree cavity	21.5" dbh live cedar
M09-262	2010	5/10	natal	Tree cavity	18.8" dbh live cedar
M09-286	2009	5/19	natal	Tree cavity	16.1" dbh live cedar
M09-286	2010	5/19	maternal	Tree cavity	live cedar; no DBH yet
M09-286	2009	5/22	maternal	Tree cavity	20.9" dbh live cedar
M09-254	2010	5/26	maternal	underground burrow	rock-laden soil
M08-140	2009	7/6	maternal	underground burrow	base of snag, rocky soil
M09-286	2009	7/9	maternal	underground burrow	along roots; base of cedar

Table 3. Parturition status and litter size for radiocollared female fisher¹ in Minnesota.

ID	Year	Age	Litter	Litter size
F08-375	2008	2	Yes	>=2
F09-360	2009	2	Yes	2
F08-304	2009	2	Yes	2
F08-077	2009	2	Yes	4
F09-362	2009	2	No	
F09-364	2009	2	No	
F09-394	2009	3	Yes	3
F08-375	2009	3	Yes	3
F08-353	2009	3	Yes	3
F09-380	2009	4	Yes	3
F09-394	2010	4	Yes	2
F08-353	2010	4	Yes	3
F09-354	2009	7	No?	
F09-354	2010	8	No?	
F09-370	2009	11	Yes	3
F10-328	2010		Yes	2
F09-461	2010		Yes	3
F10-507	2010		Yes	3

[†] Excludes unknown-aged nulliparous females, and all 1 year olds.

Table 4. Natal and maternal den structures used by radio-collared female fishers in Minnesota.

ID	Year	Confirmed	Den type	Den structure	Den details
F08-353	2010	3/24	natal	Tree cavity	15.1" dbh live aspen
F10-507	2010	3/26	natal	Tree cavity	25.6" dbh live oak
F09-394	2010	3/26	natal	Tree cavity	24.9" dbh live aspen
F09-360	2009	4/8	natal	Tree cavity	15.3" dbh aspen snag
F08-353	2009	4/8	natal	Tree cavity	23.2" dbh live aspen
F08-375	2009	4/9	natal	Tree cavity	21.9" dbh w. pine snag
F09-394	2010	4/9	maternal	Tree cavity	22.1" dbh live aspen
F09-461	2010	4/11	natal	Tree cavity	18.3" dbh live oak
F10-507	2010	4/13	maternal	Tree cavity	22.1" dbh aspen snag
F09-380	2009	4/14	natal	Tree cavity	23.6" dbh aspen snag
F09-370	2009	4/15	natal	Tree cavity	23.5" dbh aspen snag
F09-394	2009	4/18	natal	Tree cavity	21.5" dbh live aspen
F09-394	2010	4/20	maternal	Tree cavity	26.1" dbh live aspen
F08-353	2010	4/22	maternal	Tree cavity	24.3" dbh aspen snag
F09-461	2010	5/18	maternal	Tree cavity	22.3" dbh live aspen
F09-360	2009	5/29	maternal	Tree cavity	19.1" dbh live oak
F08-375	2008	6/25	maternal	Hollow log	15.7" diam. sugar maple

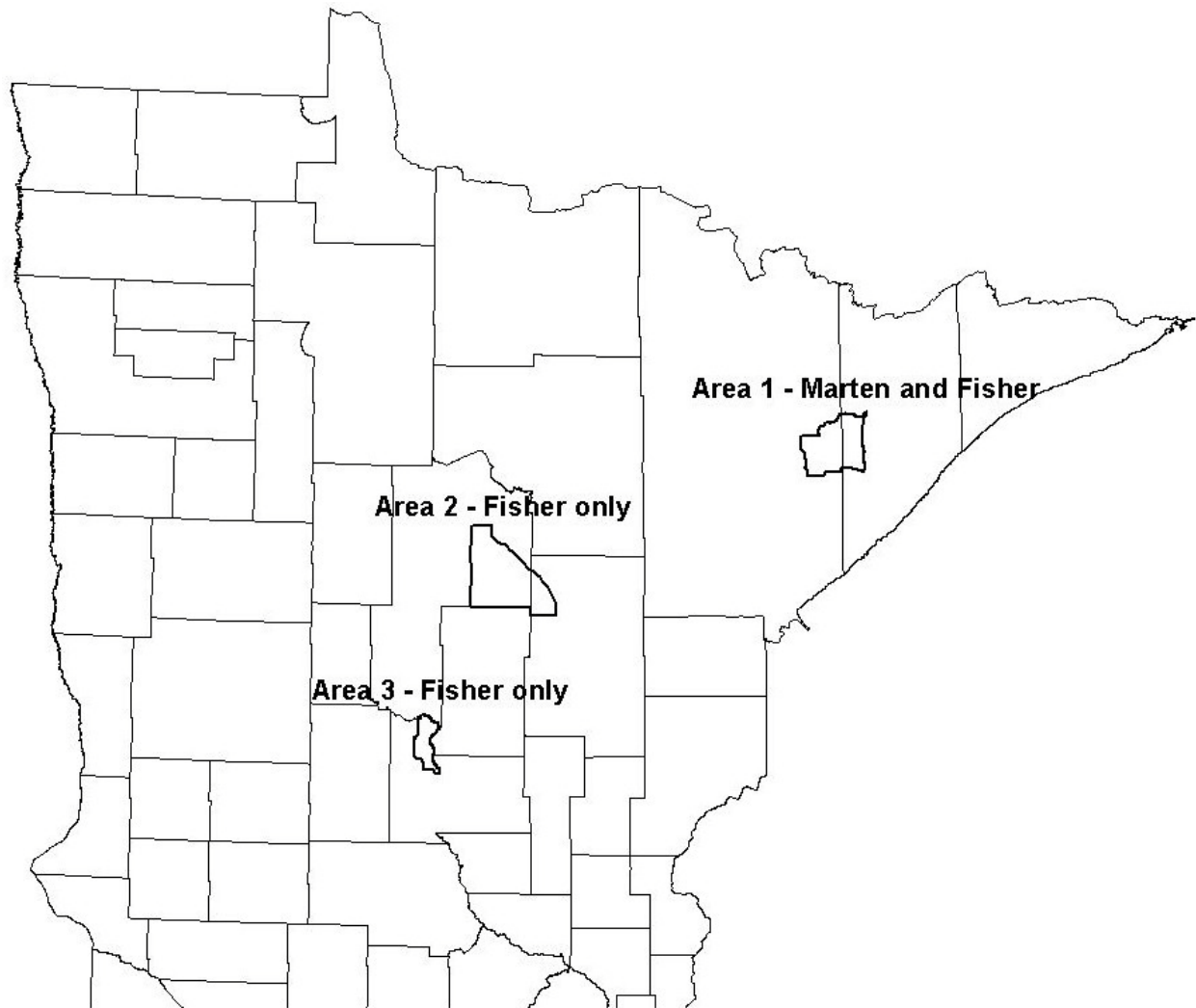


Figure 1. Fisher and marten study areas in Minnesota, 2008-2010.

SURVIVAL AND CAUSES OF MORTALITY FOR FISHER AND MARTEN IN MINNESOTA

John Erb, Barry Sampson, and Pam Coy

SUMMARY OF FINDINGS

As part of a larger project on *Martes* ecology in Minnesota, we began monitoring survival of radiocollared fisher (*Martes pennanti*) and marten (*Martes americana*) during winter 2007-08. Including the pilot year of the study, a total of 86 martens (44F, 42M) and 45 fishers (25F, 20M) have been captured. Of the 86 martens radiocollared, 39 are still actively monitored (19F, 20M), radio-contact was lost on 12 (9 slipped collars, 3 missing), and 35 deaths have occurred. Of the 35 known deaths (17F, 18M), most have been from regulated fur trapping ($n=10$; 9M, 1F) and predation ($n=20$; 13F, 7M). Of the 20 predation events, 14 marten were killed by mammalian predators, while 6 were taken by raptors, most during late winter and spring. To date, predation mortality of marten has been noticeably female-biased ($\sim 2:1$). Conversely, trapping harvest of marten is significantly male-biased. The combination of male-biased harvest mortality and female-biased non-harvest mortality may produce offsetting effects on the population sex ratio. Of the 45 fishers captured, 42 were radiocollared, of which 14 are still being monitored (8F, 6M), radio contact was lost on 14 (10 belting hardware failures, 3 missing, 1 collar removed) and 14 deaths (8F, 6M) have occurred (1 struck by a vehicle, 1 accidentally trapped out of season, 2 legally trapped, 2 died from unknown but apparently natural causes, and 8 (6F, 2M) were killed (1 possibly scavenged) by other predators). Although sample size is small, all predation mortality of fishers took place from March – May. Five of the 8 predation deaths, all females, were by mammalian predators, with the remaining 3 by raptors. Of greatest significance, all 6 of the female fishers killed by predators were adults, and 5 of the 6 were killed while they still had dependent young in natal dens, indirectly resulting in the death of their 14 kits. We suspect that energetic demands faced by adult female fishers with kits (i.e., lactation, and shortly after the energetically demanding winter) force them to increase their activity in search of food. In addition, activity likely increases as a result of breeding activity in the weeks following parturition, and all the increased activity occurs at a time when concealment cover is diminished (i.e., before 'green-up'), thereby exposing them to increased predation risk. It remains unclear whether the fisher mortality pattern we have observed to date is consistent with past dynamics, and if not, whether the underlying explanation is related to short-term (e.g., periodic fluctuations in prey) or long-term (e.g., deteriorating habitat quality) changes affecting fisher energetics/activity, or a result of changes in the predator community. What is clear from initial results is that for both species, predation has been the dominant source of mortality.

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 areas as far west as Roseau County. Limited information suggests they occurred as far south as Crow Wing County and as far southwest as Polk County. As a result of unregulated 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, including southeast Minnesota (Swanson et al. 1945, Balser and Longley 1966). The impacts of unregulated 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 lower, though perhaps increasing, in the Red River Valley (western Minnesota) and the highly fragmented transitional forests in central Minnesota. Peak harvest levels have been near 4,000 and 3,500 for marten and fisher, respectively. However, due to apparent multi-year population declines for both species, harvest seasons were reduced from 16 days to 9 days for the past 3 seasons, with harvests averaging 2,000 and 1,400 for marten and fisher, respectively.

While both species appear to have naturally re-colonized a significant portion of their historic range, Minnesota-specific information on survival and causes of mortality is limited. Except for harvest data, 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 specific to marten, now nearly 30 years old, and based on a very limited sample size. Gathering cause-specific mortality information can be useful for informing population models, detecting unknown mortality agents, and guiding management remedies to any population declines of concern.

Krohn et al. (1994) estimated 11% annual non-harvest mortality for adult fisher in Maine, while York (1996) estimated 19% and 7% annual non-harvest mortality (incl. 4% poaching mortality on males) for adult male and female fisher, respectively, in Massachusetts. Excluding the first 4-5 months of life, juvenile non-harvest mortality rates have been estimated to be 28% in Maine (Krohn et al. 1994), and 0% (females) and 23% (males) in Massachusetts (York 1996). While mortality may be higher in the first months of life than the rest of the year, if we assume a similar non-harvest mortality rate during the first 4-5 months of life, we calculate that annual non-harvest mortality for juveniles would be ~ 56% in Maine. Combining minimum summer survival estimates for kits with telemetry estimates of survival the rest of the year, York (1996) estimated ~ 67% (males) and 22% (females) annual non-harvest mortality for juveniles in Massachusetts. Kelly (1977, in Paragi et al. 1994) reportedly estimated 18% annual mortality of juveniles and 44% annual mortality for adult fisher in New Hampshire. More recently, Koen et al. (2007) estimated annual mortality rate (including harvest mortality) of fishers in Ontario to be 55-67% for males, and 29-37% for females. While non-harvest mortality of adult fishers is often presumed to be 'low', it has not always proven to be the case. Furthermore, there is limited data on which to assess the amount of geographic or temporal variation in non-harvest mortality of fisher.

Marten are more susceptible to natural mortality, primarily via predation. Survival data is available from Maine (Hodgman et al. 1994, 1997), Ontario (Thompson 1994), Oregon (Bull and Heater 2001), British Columbia (Poole et al. 2004), Alaska (Flynn and Schumacher 1995, 2009), Quebec (Potvin and Breton 1997), and Newfoundland (Fredrickson 1990). While we do not summarize details of these studies here, a couple conclusions are worthwhile. First, when comparing across studies, annual adult non-harvest mortality rates varied from ~ 0.07 – 0.48. Juvenile data was rarely separated, but a few studies pooled ages, and mortality rates also fell within the above interval. While this variability may be attributable to both sampling and biological variability, the wide range suggests that it is risky to assume results from any area are applicable elsewhere. Secondly, at least 1 study (Maine; Hodgman et al. 1997) has documented significantly higher natural mortality for females compared to males, and others researchers have postulated this to be common given the typical male-biased harvest, 50:50 sex ratio at birth, and often balanced adult sex ratio (Strickland et al. 1982, Strickland and Douglas 1987). Due to male-biased harvest and our *assumed* sex-related equality in non-harvest mortality, our marten population model currently projects a very female-biased population, contradicting our preliminary capture results and suggesting that our model inputs may overestimate female survival, underestimate male survival, or incorrectly assume a 50:50 birth sex ratio.

As part of a larger project on *Martes* ecology in Minnesota (Erb et al. 2009), we began monitoring survival and causes of mortality for fisher and marten. After initial evaluation of field methods during the pilot year of the study, winter 2008-09 marked the beginning of full-scale research activities. While details are not further discussed here, we are also collecting data on various potential correlates to survival (e.g., prey dynamics, winter severity, diet, habitat use, activity patterns, and body condition). Herein we present basic information on field methods, and descriptive information regarding number of captures and number and causes of deaths. We defer a more comprehensive and statistically-oriented analysis until a later time.

STUDY AREA

Marten research is focused on 1 study area located in northeastern Minnesota (Figure 1; Area 1), though an occasional marten is captured and radiocollared in Area 2 (Figure 1). Area 1 (~ 700 km²) is composed of approximately 69% mixed coniferous-deciduous 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 < 1% deciduous forest. Area 1 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 basic summaries, we do not discuss other aspects of that project in this report. Area 2 (1075 km²), 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 typically 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). Fishers 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. Martens were ear-tagged with a monel #1 tag (National Band and Tag Co., Newport, KY) in each ear.

During processing, we placed animals on either chemical hand warmers or heating pads connected to a power inverter and 12 volt battery. Portable shelters and 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). 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). Since the pilot year, we have primarily deployed ATS M1940 (~ 43 g) or Sirtrack TVC-162 collars (~ 45 g) on fisher. The majority of martens in both years have been fitted with Holohil MI-2 collars (~ 31 g). While not discussed in detail here, we retrofitted each collar with a temperature data logger, in part to allow for determination of exact time of death.

All radio-locations, except for some taken during the den-monitoring period, will be obtained from fixed-wing aircraft at approximately weekly intervals. When a radiocollar emits a mortality signal, we usually investigate and recover the animal or collar within 1-2 days. To determine cause of mortality, we use a combination of field investigation and animal necropsy. Starting in the second year of the project, we also began collecting forensic samples (hair by wound, wound swabs) from all animals exhibiting signs of being predated, particularly if a mammalian predator is suspected. Forensic samples are submitted to the University of California-Davis Veterinary Genetics Laboratory. If non-predation natural causes are suspected after initial analysis (i.e., no visible trauma), the animal is submitted to the University of Minnesota's Veterinary Pathology Lab for a full pathological exam.

RESULTS AND DISCUSSION

Including the pilot year of the study, a total of 86 martens (44F, 42M) and 45 fishers (25F, 20M) have been captured. Tooth aging has not yet been completed for all animals, and herein we do not report any formal survival estimates. Instead, we provide a simple overview of the fate of collared animals.

Of the 86 martens radiocollared, 39 (19F, 20M) are actively being monitored, 9 individuals (6F, 3M) were able to subsequently slip their collars, and 3 are missing. In addition, we have confirmed 35 (17F, 18M) mortalities, 3 from capture/collar related complications, 1 from starvation (intestinal disorder), 1 from unknown natural causes, 10 (1F, 9M) from regulated fur trapping, and 20 (13F, 7M) from predation. Although we have confirmed predation mortality in most months of the year (Figure 2), it is concentrated in December and late-winter through spring (Feb – May), with little predation mortality in January or summer through fall. We note, however, that all 4 predation mortalities that occurred in December took place within 2 weeks of capture, and therefore may be censored from the final dataset. Of the 20 predation events, 14 marten were killed by mammalian predators, while 6 were taken by raptors. Forensic (DNA) analysis of samples collected from dead marten (mammalian predation only) is incomplete. To date, DNA analysis has confirmed bobcat predation in 2 cases, with a third death, based on sign in the snow, also attributed to bobcat.

Predation mortality on marten has been noticeably female-biased (~ 2:1). Conversely, and within the context of Minnesota's harvest season structure, trapping harvest of marten is significantly male-biased. Within the biological year for marten (~ 1 May – 30 Apr), the male-biased harvest mortality occurs prior to the female-biased non-harvest mortality. While we suspect that the birth sex ratio is balanced, data is lacking and there is some indication from our results that birth sex ratios (or early juvenile survival) could favor males – i.e., shortly after a very male-biased harvest, our more intensive live-trapping efforts have yielded balanced, not female-biased, sex-ratios. If the population sex ratio is in fact reasonably balanced starting post-harvest (early winter), the subsequent female-bias we have observed in number of predated marten may be due to differential vulnerability, not differential abundance. Regardless, the combination of male-biased harvest mortality and female-biased non-harvest mortality may produce offsetting effects on the population sex ratio.

Of the 45 fishers radiocollared, 14 are still being monitored (8F, 6M), 3 are missing, 10 shed their collars due to belting design failures, and 1 collar was removed at the time of

recapture due to neck abrasion. In addition, 3 juvenile males were ear-tagged only. Of the 14 known deaths (8F, 6M), 1 was struck by a vehicle, 1 was accidentally trapped out of season, 2 were legally trapped, 2 died from unknown but apparently natural causes, and 8 (6F, 2M) were killed by other predators (scavenging by an eagle can't be ruled out in 1 case).

Although sample size is small, all predation mortality of fishers took place from March – May (Figure 3), and very rarely was any portion of a dead fisher consumed. Five of the 8 predation deaths, all females, were by mammalian predators. In one case, bobcat was confirmed via trail camera placed at the site a fisher was cached. We are awaiting forensic results for several other cases. Bald eagles are suspected in 2 of the 3 raptor predation events, both of male fisher, though as noted above we can't rule out scavenging in 1 case (only the radiocollar was retrieved directly underneath an active eagle nest). The third raptor predation involved a female fisher, likely attacked by an owl or hawk.

Of greatest significance, all 6 of the female fishers killed by other predators were adults, and 5 of the 6 were killed while they still had dependent young in natal dens, indirectly resulting in the death of their 14 kits. We suspect that energetic demands faced by adult females with kits (i.e., lactation, and shortly after the energetically demanding winter) force them to increase their activity in search of food, and preliminary data from temperature data loggers on radiocollars suggests this to be the case. In addition, activity likely increases as a result of breeding activity in the weeks following parturition, and all the increased activity occurs at a time when concealment cover is diminished (i.e., before 'green-up'), thereby exposing them to increased predation risk. Regardless of the explanation, and acknowledging the limited sample size, it seems unlikely that the high level of predation on nursing females is sustainable, which may partially explain the recent decline in fisher abundance. However, the correlates to the timing of predation mortality that we have mentioned are not new challenges for adult female fisher, and the population appears to have been in decline only for the last ~ 6 years, suggesting that other factors may be 'altering the system'. It remains unclear whether the fisher mortality pattern we have observed to date is consistent with past dynamics, and if not, whether the underlying explanation is related to comparatively short- (e.g., periodic fluctuations in prey) or long-term (e.g., deteriorating habitat quality) changes affecting fisher energetics/activity, or relatively rapid changes in the predator community (e.g., the increased bobcat population).

ACKNOWLEDGEMENTS

We thank volunteer Richard Nelles for his dedicated assistance with trapping and other project activities. We also acknowledge Carolin Humpal for her ongoing assistance with various aspects of the project, including trapping, tooth aging, and prey sampling. We also thank pilots Al Buchert and Don Murray for aerial telemetry efforts, and are grateful to staff from the U.S. Forest Service for their logistical assistance on various aspects of the project.

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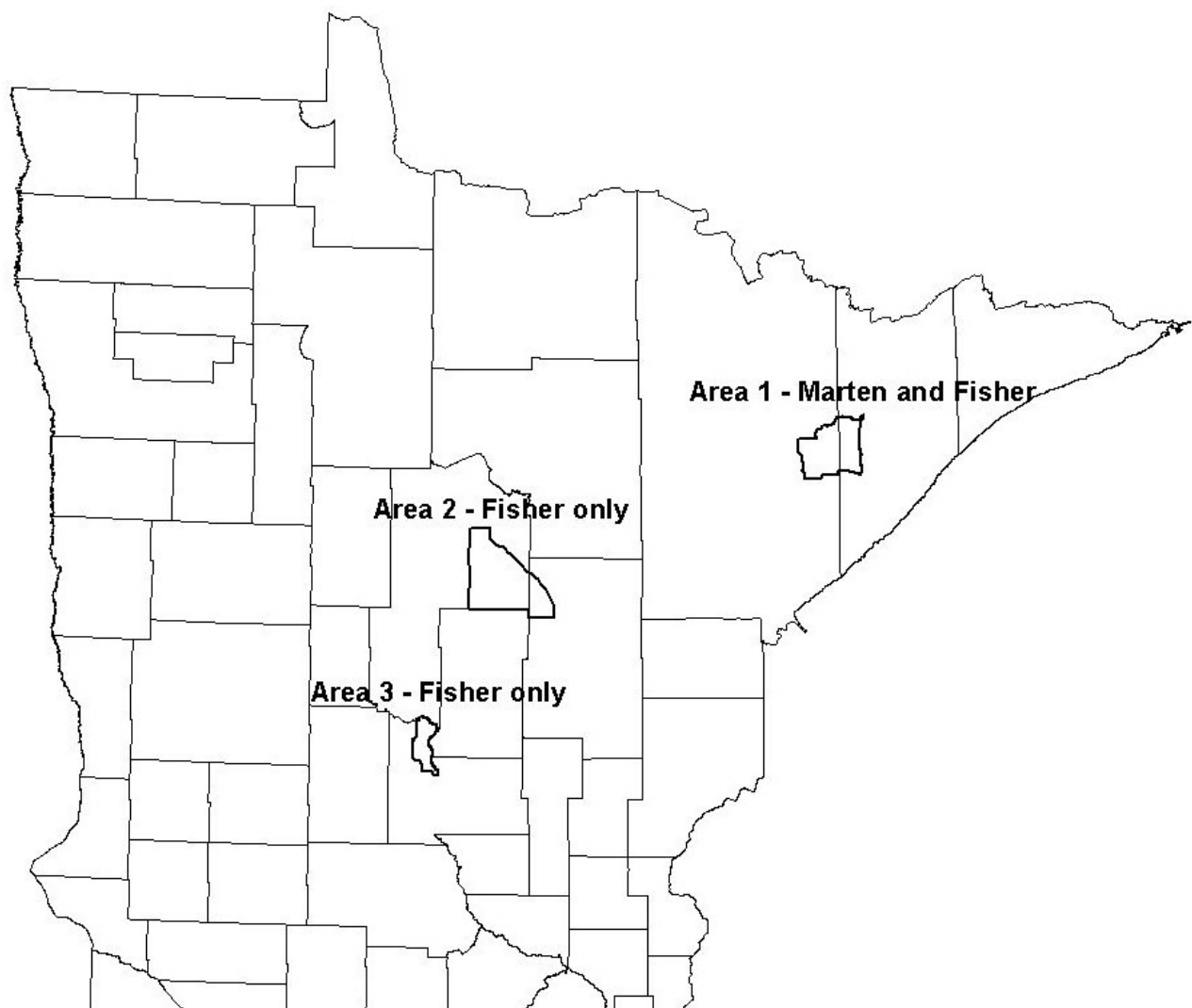


Figure 1. Fisher and marten study areas in Minnesota 2008-2010.

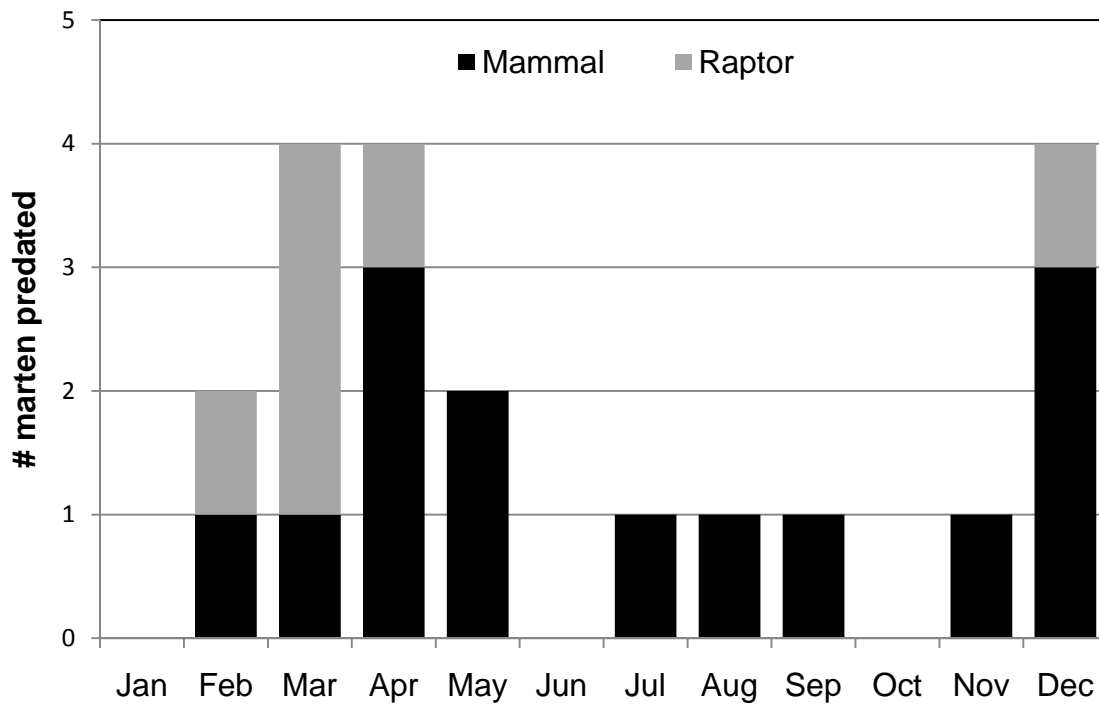


Figure 2. Seasonal timing of marten deaths attributable to predation in northeast Minnesota, 2007-2009.

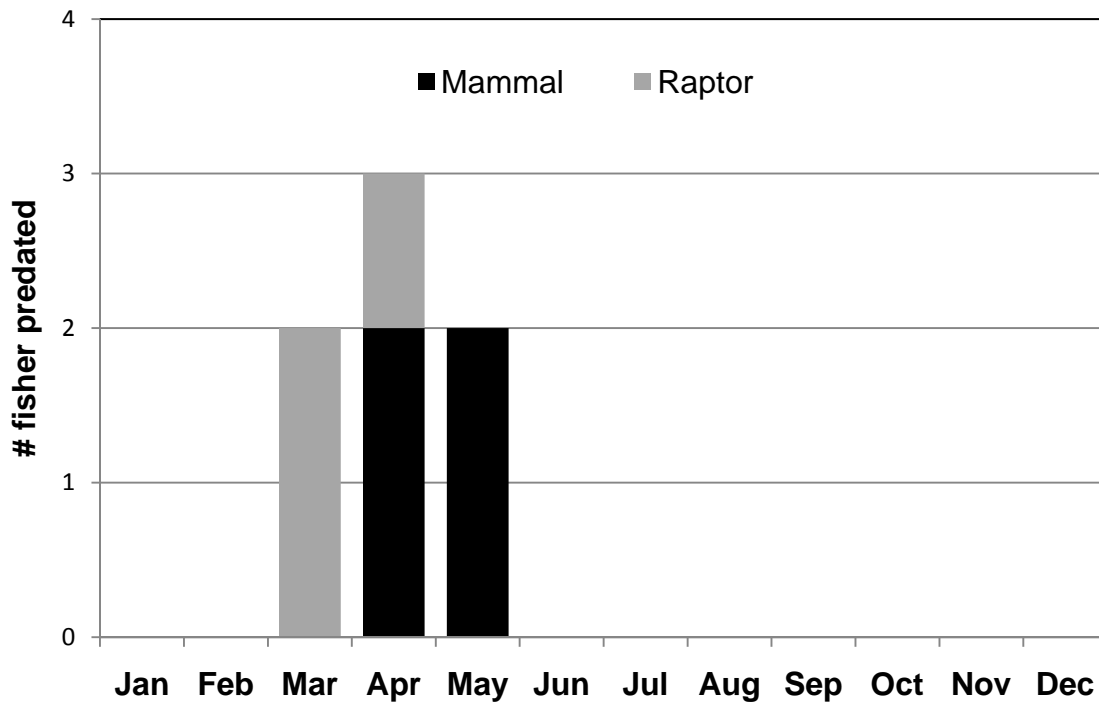


Figure 3. Seasonal timing of fisher deaths attributable to predation in north-central Minnesota, 2007-2009.

MOOSE POPULATION DYNAMICS IN NORTHEASTERN MINNESOTA

Mark S. Lenarz, Michael W. Schrage¹, Andrew J. Edwards², and Michael Nelson³

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 2010, 105 collared moose (48 adult males and 57 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 2 manuscripts are published, 1 manuscript is in press, and 2 other manuscripts are in preparation.

INTRODUCTION

Moose formerly occurred throughout much of the forested zone of northern Minnesota. Today they 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. 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 150 moose in southern Lake County and southwestern Cook County between 2002 and 2008, 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. 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 a survey the following year 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 2010, 105 collared moose (48 adult males and 57 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 2 moose. Moose that died within 2 weeks of capture (6) were designated as capture mortality. Hunters killed 17 moose, 2 were poached, and 11 were killed in collisions with vehicles (cars, trucks, or trains). The remaining mortality (69) was considered to be non-anthropogenic and causes included wolf predation (8), bacterial meningitis (1), or unknown (60).

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 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 18% for males (0 to 40%, SE = 5, $n = 7$) and 21% for females (5 to 30%, SE = 3, $n = 7$; 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 (21%; 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 1997), 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 2009, 197 radiocollared adult females gave birth to a minimum of 167 calves (96 singles, 34 twins, and 1 set of triplets; M. W. Schrage, Fond du Lac Resources Management Division, unpublished). The annual ratio of calves: radiocollared females ranged from 0.53 to 0.95 ($\bar{x} = 0.82$, SE = 0.06, $n = 6$). These estimates were biased low because in 4 of 6 years, radiocollared females not accompanied by calves during the MJ survey were subsequently observed to be accompanied by a single calf (4 in 2004, 4 in 2005, 1 in 2007, 4 in 2008). 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 K carrying capacity, respectively.

During the past year, 2 manuscripts discussing the results of this research have been prepared for publication. The first, entitled "Living on the edge: Viability of moose in northeastern Minnesota" will be published in the July 2010 issue of the Journal of Wildlife Management. A second manuscript, entitled "Winter body condition of moose (*Alces alces*) in a declining population in northeastern Minnesota" was accepted by the Journal of Wildlife Diseases and is in press. Two additional manuscripts are in preparation. One will discuss the development of the sightability model used in our aerial moose survey to correct for visibility bias. A second paper will evaluate the use of cover types for thermal refuge using compositional analysis and Euclidian distance analysis.

ACKNOWLEDGMENTS

We thank Al Buchert from the DNR's Enforcement Division for his piloting skills throughout the project.

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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 ¹	7% (25) ²	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)
2008	0% (7)	21% (38)	16% (45)
Mean	18%	21%	20%

¹ Period: 1 June – 31 May.

² Sample size as of 31 May

LIVING ON THE EDGE: VIABILITY OF MOOSE IN NORTHEASTERN MINNESOTA¹

Mark S. Lenarz, John Fieberg, Michael W. Schrage², and Andrew J. Edwards³

ABSTRACT

North temperate 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 Mountain cordillera. We used a series of matrix models to evaluate the demographic implications of estimated survival and reproduction schedules for a moose population in northeastern Minnesota, USA, between 2002 and 2008. We used data from a telemetry study to calculate adult survival rates and estimated calf survival and fertility of adult females using results of helicopter surveys. Estimated age- and year-specific survival rates showed a sinusoidal temporal pattern during our study and were lower for younger and old aged animals. Estimates of annual adult survival (when assumed to be constant for ages >1.7 yr old) 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. Point estimates for the finite rate of increase (λ) from yearly matrices ranged from 0.67 to 0.98 during our 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. Even if heat stress is not responsible for current levels of survival, continuation of this growth rate will ultimately result in a northward shift of the southern edge of moose distribution. Population growth rate, and its uncertainty, was most sensitive to changes in estimated adult survival rates. The relative importance of adult survival to population viability has important implications for harvest of large herbivores and the collection of information on wildlife fertility.

1. Abstract of paper in press in Journal of Wildlife Management.

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WINTER BODY CONDITION OF MOOSE (*ALCES ALCES*) IN A DECLINING POPULATION IN NORTHEASTERN MINNESOTA¹

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ABSTRACT

Assessments of the condition of moose (*Alces alces*) 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 northeastern Minnesota by body condition scoring (BCS_F, scale of 0-10), and 67 of these by ultrasonographic measurements of rump fat (Maxfat), which was used to estimate ingesta-free body fat (IFBF) in all but 2 of these females. Scores of the BCS_F were related ($r^2 = 0.34$, $P < 0.0001$) to Maxfat. Body condition scores were not affected by sex X capture-year, capture-year, or age-at-capture, but the mean body condition score of males (6.5 ± 0.2 [SE], $n = 40$) was less ($P \leq 0.009$) than that of females (7.4 ± 0.2 , $n = 39$). Overall, Maxfat ranged from 0 to 4.6 and 0.3 to 2.8 cm in females and males, respectively, and was unaffected by age-at-capture. There was a sex X capture-year effect ($P = 0.021$) on Maxfat; mean values were stable for males during winters 2003 to 2005, but in females were lowest during 2003, consistent with lowest pregnancy rates and lowest winter and spring survival compared to 2004 and 2005. Based on estimates of % IFBF, late winter-early spring survival in 2003 of at least 6.1% of the collared animals assessed by Maxfat, 11.8% of the adult females specifically may have been seriously challenged directly by poor condition. Data from this study provide reference values, and assessments of body condition of moose will be an essential component of the additional comprehensive research needed to more closely examine and better understand relations of seasonal heat stress, nutrition, body condition, habitat use, and performance of this important remaining viable, but declining population. We will concentrate on improving the reliability of the BCS_F to extend the range of IFBF estimation (once rump fat is depleted) using an index that combines BCS_F scores and Maxfat measurements.

¹Journal of Wildlife Diseases (2010) Accepted.

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HABITAT SELECTION BY MALE RUFFED GROUSE AT MULTIPLE SPATIAL SCALES

Meadow J. Kouffeld¹, Michael A. Larson, and R. J. Gutiérrez¹

SUMMARY OF FINDINGS

We conducted the first of two field seasons during 2009. We located 742 drumming structures, and 454 of those structures were within 200 m of a transect. We sampled vegetation characteristics at 434 used drumming structures and 434 nearby unused structures. We will complete the second field season during 2010 before analyzing the data.

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 and/or quantity of ruffed grouse habitat in Minnesota.

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

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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 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 km²; 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 these 2 adjacent sites to comprise our study area. The study area is in portions of Red Lake Wildlife Management Area and adjacent Beltrami Island State Forest. We did not include Red Lake Band Tribal Lands in our study.

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 that was nearest to one of 30 randomly located points in the aspen study site and 30 randomly located points in the conifer study site. We determined randomly the direction of each transect from that point along the road or trail and also 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 were 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 were detected during a survey, the exact location of each one was determined by approaching it and identifying the log or other structure on which it was standing to drum, often indicated by the presence of fresh droppings. Universal Transverse Mercator (UTM) coordinates were taken using a hand held global positioning system (GPS) unit at drumming structure and the drumming structure's location was confirmed by approaching 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 measured characteristics of ruffed grouse habitat at 3 spatial scales. The smallest scale was the area immediately surrounding drumming locations identified during surveys. Characteristics at this scale were measured in the field. The same variables were measured at an unused but potential drumming structure (e.g., log or stump with no signs of use by grouse) nearest a randomly selected point within 85 m of each used drumming structure. 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 ensured that selected unused locations were within the home range, whereas the 146-m radius of the home range would not have. This information was collected for all used drumming structures that fell within 200 m of the transect line.

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 roadside 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 that may be related to ruffed grouse habitat quality. We may randomly select a subsample of roadside 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 roadside 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 roadside transect. In most cases, however, much uncertainty exists about the locations of the points because the locations 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).

RESULTS

We conducted the first of two field seasons during 2009. We located 742 drumming structures, and 454 of those structures were within 200 m of a transect. We sampled vegetation characteristics at 434 used drumming structures and 434 nearby unused structures. We will complete the second field season during 2010 before analyzing the data.

ACKNOWLEDGEMENTS

We appreciate reviews and comments on our proposal from Wes Bailey, John Fieberg, and Mark Lenarz. We thank Wes Bailey for GIS assistance during the selection of the study area; Ryan Anderson, David Grunzel, Lyn Snoddy, Ben Stenberg, and Ed Zlonis for collecting data; and Ted Dick, Jeff Dittrich, Scott Laudenslager, Gretchen Mehmehl, Lorelle Berkeley, and Jayson Hansen for their assistance with logistics and field work.

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HABITAT SELECTION OF SPRUCE GROUSE AT MULTIPLE SPATIAL SCALES IN NORTHWEST MINNESOTA

Michael A. Larson

SUMMARY OF FINDINGS

This study was proposed during spring 2010. We will evaluate some of the field methods during summer 2010 and intend to have full field seasons of data collection during spring and summer of 2011 and 2012.

INTRODUCTION

The spruce grouse (*Falciennnis canadensis canadensis*) is listed as a Species of Greatest Conservation Need (SGCN) by the Minnesota Department of Natural Resources (MNDNR 2006), which cited its dependence on a potentially vulnerable habitat type and a lack of population trend data. It is also on the Regional Forester's Sensitive Species list for the Chippewa National Forest (Gregg et al. 2004:22). Projected climate change could have dramatic effects on the extent and composition of forests in Minnesota (Frelich and Reich 2009), and boreal coniferous forests in Minnesota are projected to experience a moderate level of climate stress relative to other areas in the United States (Joyce et al. 2008:11). Due to the unknown or tenuous status of spruce grouse along the southern edge of their range and the existence of several threats to the viability of their populations, there is interest in learning more about their status and ecology. The Association of Fish and Wildlife Agencies (Williamson et al. 2008) recommended developing formal surveys for monitoring population change and conducting research on the impacts of habitat change and hunting on spruce grouse.

Previous studies of spruce grouse habitat focused on their associations with certain forest cover types and traditional metrics of forest structure (e.g., tree density and height). They did not address important questions that are relevant to how we currently manage forests. For example, we do not know whether the density and species of residual trees are important, what size and shape of forest stands are best, what proportions of different cover types in a landscape are best, and what the importance is to spruce grouse of different native plant community types. Furthermore, all three of the previous studies of spruce grouse in Minnesota were conducted in rather unique study areas (i.e., either entirely black spruce lowlands or primarily peatlands), so it is difficult to apply their results broadly (Anderson 1973, Haas 1974, Pietz and Tester 1979).

This study will provide information about how to improve forest management for spruce grouse. The habitat selection information learned during this study also will be beneficial for assessing the vulnerability of spruce grouse to changes in forests that are anticipated due to climate change. Additionally, the surveys conducted for this study will provide an empirical basis for designing a spring survey that could be used to monitor the status of spruce grouse populations throughout northern Minnesota every 1–5 years, for which there is increasing interest.

OBJECTIVES

1. To determine which habitat characteristics are most highly correlated with the presence of displaying male spruce grouse during spring in Minnesota; and
2. To determine which habitat characteristics are most highly correlated with the presence of female spruce grouse with broods during summer in Minnesota.

STUDY AREA

We will conduct the study in the Red Lake Wildlife Management Area and adjacent portions of the Beltrami Island State Forest, which are in Lake of the Woods, Beltrami, and Roseau counties in northwestern Minnesota.

METHODS

We will conduct repeated surveys at a random sample of points, stratified by important categories of cover types. During spring the surveys will focus on males, whose flutter-flight displays are detectable from up to 100 m away (Keppie 1992). We will survey for females and broods during summer using a recorded chick distress call (Healy et al. 1980, Bouta 1991, Ross and Johnson 2008). With survey data from this design we will compare points that were and were not occupied, using attributes measured at several spatial scales.

ACKNOWLEDGEMENTS

The initial motivation, ideas, and funding for this project came from Gretchen Mehmehl. She, Scott Laudenslager, and Ted Dick contributed greatly to the development of the project, and they will oversee the field work. Maggie Anderson and Gregg Knutsen from Agassiz National Wildlife Refuge also participated in the initiation of the project and will be valuable collaborators.

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LANDSCAPE CHARACTERISTICS ASSOCIATED WITH DANCING GROUNDS OF SHARP-TAILED GROUSE

Michael A. Larson and J. Wesley Bailey

SUMMARY OF FINDINGS

We are developing a habitat model to describe the landscape characteristics associated with dancing grounds of sharp-tailed grouse (*Tympanuchus phasianellus campestris*) across their range in Minnesota. Our analyses are not complete, so the results are only preliminary and are subject to revision.

INTRODUCTION

Sharp-tailed grouse in Minnesota occur in open landscapes of “grass, brush, savanna, and boreal peatland,” which “are sometimes associated with small grain and livestock farming” (Berg 1997:1, 4). Although sharp-tailed grouse habitat was widely distributed in Minnesota during the early- and mid-1900s, the range of sharp-tailed grouse is now limited to areas in the northwest and east central portions of the state (Figure 1). The succession and conversion of their habitat to unsuitable cover types coincided with a dramatic decline in estimates of annual harvest by hunters from 120,000 sharp-tailed grouse in 1952 to 4,000 in 1965 (Landwehr 1984). Since 1980 the average number of grouse per dancing ground during spring has fluctuated between 7 and 13 and has had a slightly positive trend (Larson 2009), whereas harvest has had a noticeably negative trend ending with harvests of 6,000–16,000 birds/year during the last decade (Dexter 2009).

To benefit sharp-tailed grouse and other wildlife, the Minnesota Department of Natural Resources’ (MNDNR) Section of Wildlife has emphasized the management and restoration of targeted open lands within the forested part of the state. These efforts include designating priority open landscapes within the Subsection Forest Resources Management Plan process and spending more money on openland/brushland management than any other habitat improvement activity in the forested regions of the state. Identifying landscapes to target with openland management, however, is challenging.

Although Solberg (1999) attempted to identify priority areas for sharp-tailed grouse management using maps and landscape characteristics, Hanowski et al. (2000) were the first to quantify the habitat characteristics of dancing grounds at the landscape scale. Both studies focused on sharp-tailed grouse range in east central Minnesota and provided valuable information. We were interested in quantifying variations in landscape characteristics associated with dancing grounds across their full geographic range in Minnesota. Our goal was to develop a spatially explicit habitat model for identifying priority areas for sharp-tailed grouse management, including habitat improvement, land acquisition, population monitoring, and potential reintroduction.

OBJECTIVES

1. To determine which landscape characteristics are most highly correlated with the presence of dancing grounds of sharp-tailed grouse in Minnesota.
2. To map variations in the quality of habitat for sharp-tailed grouse dancing grounds throughout their range in Minnesota.

STUDY AREA

We defined the study area as occurring within both of 2 different boundaries for describing the geographic extent of sharp-tailed grouse range in Minnesota (Figure 1). One boundary encompassed the subsections of Minnesota’s Ecological Classification System (ECS,

following Cleland et al. 1997) where dancing grounds were observed during 1991–1993. The sample of dancing ground locations that we used is described and justified in the METHODS section below. The other boundary was the 85% kernel density estimate around observed dancing grounds. We selected the 85% kernel boundary because it encompassed 21% less area than the 95% kernel boundary and excluded only 1% of the used sites. The 80% kernel boundary encompassed 32% less area than the 95% kernel boundary, but we thought it excluded too many used sites (5%).

METHODS

We investigated habitat selection of sharp-tailed grouse for dancing grounds in Minnesota by comparing the attributes of a sample of locations known to have been used as dancing grounds (i.e., used sites) and an independent sample of locations that were representative of areas available for use as dancing grounds (i.e., available sites).

Use-Availability Data

Used sites were detected during annual surveys conducted by the MNDNR during spring of each year (see Larson 2008 for survey methods). Although the spatial sampling design of the survey was haphazard, the spatial extent of the survey covered the known range of the species in Minnesota, and we think the probability of detecting an existing dancing ground in a given year was >0.3 (M. A. Larson, unpublished data). The sample of used sites consisted of locations where a dancing ground was observed at least once during 1991–1993 because that was the time interval during which the land cover imagery was captured (see Landscape Data below). Each used site was included in the set of data only once, and locations were precise to the quarter-section of the Public Land Survey.

We selected the sample of available sites from the spatial extent defined in the STUDY AREA section above. The only other constraint we applied for the area from which available sites were randomly selected was that the forest and non-habitat cover types (defined below) were excluded. The definition of the study area, or spatial extent, is important for use-availability comparisons. Using a more restrictive study area (e.g., within a limited-distance buffer of known dancing grounds) would lead to inferences focusing on specific characteristics of patches of open cover types (e.g., area, edge density). Using a broader extent for the study area (e.g., all of northern Minnesota) likely would lead to inferences emphasizing the importance of open lands in general. We sought a balance between those extremes.

Landscape Data

We created for the study area a Geographic Information System (GIS) data layer consisting of cover types relevant to sharp-tailed grouse habitat. We started with level 4 classes of land use/land cover from the Minnesota Gap Analysis Project (MN-GAP, MNDNR 2001) and reclassified them to the following 8 cover types: cropland, disturbed grass (grassland and prairie cover types on non-public lands), undisturbed grass (grassland and prairie cover types on public lands), sedge meadow, shrub (lowland deciduous shrub), bog (lowland evergreen shrub, stagnant black spruce, and stagnant tamarack), forest (all other MN-GAP level 4 forest classes, including upland shrub, which is primarily post-harvest regeneration), and non-habitat (all other MN-GAP level 4 classes).

Then we superimposed (i.e., replaced the MN-GAP data with) data from better sources for 3 of the cover types. Using the National Wetlands Inventory (NWI, Cowardin et al. 1979, Minnesota Land Management Information Center 2007) we selected scrub-shrub (broad-leaved deciduous and deciduous) and persistent emergent types that occurred within flooded, saturated, and seasonally flooded NWI water regime modifiers. We added the NWI scrub-shrub areas to our shrub cover type and the persistent emergent areas to our sedge meadow cover type, regardless of what the MN-GAP classification was. Then we added areas with

herbaceous vegetation cover practices from the 1997 Conservation Reserve Program (CRP, Minnesota Natural Resources Conservation Service 2010) to our undisturbed grass cover type, regardless of what the MN-GAP or NWI classifications were.

Our land cover layer is a raster (ESRI) grid in UTM zone 15 (NAD 83) with a cell size of 30 m x 30 m. We used ArcGIS 9.3.1 to calculate landscape metrics for areas within 4 different buffer distances of each used and available point (i.e., 400 m, 800 m, 1,600 m, and 3,200 m). We considered a total of 19 variables for inclusion in our models (Table 1). To preclude potential computational problems caused by large values we normalized the values of all covariates (i.e., $[x_i - \bar{x}]/SD[x]$) before fitting the models.

Model Set

Correlations between values from different spatial scales for the same variable were very high for most variables, so we decided to use only the 800-m scale for our *a priori* models. That spatial scale was similar to those at which Hanowski et al. (2000) found that characteristics differed most between active and inactive leks (i.e., 500 and 1,000 m). We also considered Simpson's Evenness Index but its values were highly correlated with values of Simpson's Diversity Index, so we retained only the latter because it accounted for the number of cover types as well as the evenness among the area of the different cover types (McGarigal et al. 2002).

We used different combinations of the variables to define 73 *a priori* models. Including an intercept term, 30, 10, 9, 10, 2, 4, 2, 3, and 2 of the models had 3, 4, 5, 6, 7, 8, 9, 10, and 11 parameters, respectively. Several of the models were formulated to be similar to the best models of Hanowski et al. (2000) and Niemuth and Boyce (2004). This is a relatively large set of *a priori* models because there are relatively few previous studies and there is still much uncertainty about the importance of different landscape characteristics.

Model Fitting

The most appropriate way to analyze and interpret data from a use-availability study design is still debated in the literature (Keating and Cherry 2004, Johnson et al. 2006). We found the approach advocated by Lele and Keim (2006), which is a form of logistic regression, to be the most appealing because it addressed potential concerns about logistic regression that were raised by Keating and Cherry (2004), and the concept of weighted distributions upon which it is based is more intuitive than alternative approaches to the analysis. We fit our models using scripts for programs R and WinBUGS provided by S. Lele, which were based on a data cloning method described by Lele (2009). These analysis methods are potentially sensitive to initial values specified by the user, so to estimate initial values we fit the models using standard logistic regression and then using the script for program R from Lele and Keim (2006), which is not as robust as the data cloning method used in the script based on Lele (2009). We used AIC values to rank the *a priori* models based on how well they fit the data.

RESULTS AND DISCUSSION

We used 1,245 randomly selected available sites and 249 used sites in our analyses. Our sample of used sites excluded 3 of the 252 dancing grounds observed at least once during 1991–1993 because they were outside the 85% kernel boundary (Figure 1). We have generated initial values for all models, but we have not yet fit all models using the data cloning method. Both methods used to generate initial values resulted in the same AIC rankings for the best 5 models, which had 9–19 parameters. Looking at the best model with a given number of parameters for models with 3–8 parameters ($n = 6$ models), the distance to nearest lek variable occurred in all of them and the area of the shrub cover type occurred in 4 of them.

Results are preliminary and are subject to revision based on continuing work on this project. When our results are complete we will compare them to those of Hanowski et al. (2000)

and Niemuth and Boyce (2004), who have developed similar models of landscape characteristics associated with the dancing grounds of sharp-tailed grouse.

ACKNOWLEDGEMENTS

We greatly appreciate the motivation and persistence of Jodie Provost in getting this project initiated and the substantial GIS support provided by Bob Wright. We also thank the members of the technical committee that helped us develop the set of models and decide which land use/land cover data to use. They were Chris Balzer, Diane Granfors (USFWS), Cynthia Osmundson, Donovan Pietruszewski, Tim Pharis, Jodie Provost, and Bob Wright. Others who participated in the development phase included Gregg Knutsen (USFWS), Dave Pauly, and Chris Scharenbroich. John Fieberg provided valuable discussion and advice regarding the analysis and interpretation of use-availability data, but he is not responsible for any errors or inappropriate inferences we may have made.

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Table 1. Variables considered in models for distinguishing sites used and available for dancing grounds of sharp-tailed grouse in Minnesota during 1991–1993.

Number	Name	Description
1	GRSU	Area in the undisturbed grass cover type
2	GRSD	Area in the disturbed grass cover type
3	SEDG	Area in the sedge meadow cover type
4	OPEN	Area in the undisturbed grass, disturbed grass, and sedge meadow cover types
5	CROP	Area in the crop cover type
6	SHRB	Area in the shrub cover type
7	BOG	Area in the bog cover type
8	FRST	Area in the forest cover type
9	SIMP	Simpson's Diversity Index ^a
10	DILK	Distance to nearest known lek, or dancing ground
11	DIGR	Distance to nearest patch of disturbed grass patch
12	DIFR	Distance to nearest patch of forest
13	DIRD	Distance to nearest road
14	RDDN	Road density
15	EDBS	Distance of edge between the bog and shrub cover types
16	EDBO	Distance of edge between the bog and open cover types
17	EDOF	Distance of edge between the open and forest cover types
18	PAFO	Number of patches in the forest cover type
19	PASH	Number of patches in the shrub cover type

^a McGarigal et al. (2002).

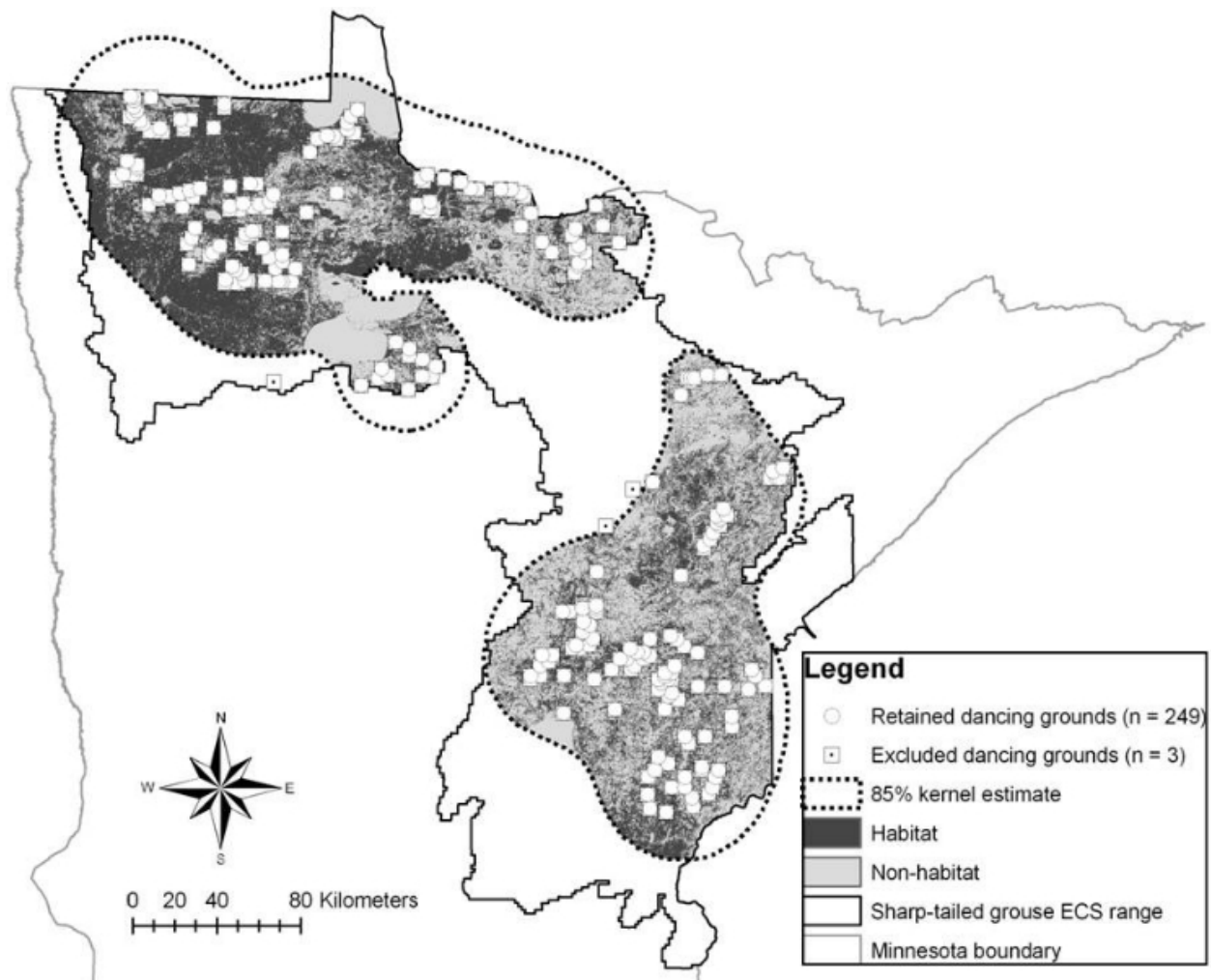


Figure 1. Map of the spatial extent of the habitat selection model for sharp-tailed grouse in northern Minnesota (shaded areas). The extent was defined as occurring within (1) occupied ECS subsections and (2) the 85% kernel estimate of space use, based upon the locations of dancing grounds that were documented during 1991–1993.

MODELING CONNECTIVITY OF SHARP-TAILED GROUSE DANCING GROUNDS TO AID IN OPEN LANDSCAPE MANAGEMENT

J. Wesley Bailey

SUMMARY OF FINDINGS

This is the first attempt to identify pathways or connections among sharp-tailed grouse (*Tympanuchus phasianellus campestris*) dancing grounds across the Minnesota range. I used Circuitscape software, which uses algorithms based on circuit theory, to model connectivity of sharp-tailed grouse dancing grounds. Raster datasets consisting of landcover converted to conductance and resistance layers are being further developed as are improvements to habitat patches. Analyses of these data are in progress; therefore, definitive results are unavailable. However, initial modeling suggests connectivity varies among dancing grounds but is greatest among clusters of dancing grounds in northwest Minnesota. In east-central Minnesota, particularly in Aitkin and Carlton Counties, individual dancing grounds occur in highly connected clusters, but connectivity among individual clusters appears limited. Data analyses will include investigating how connectivity may affect dancing ground persistence and I will evaluate connectivity differences among high versus low count dancing grounds.

INTRODUCTION

To date, open-brushland management funds are allocated to Minnesota Department of Natural Resources (MNDNR) wildlife work areas based on receipt and approval of management project proposals submitted by Area offices. However, there is some uncertainty whether the current brushland project proposal process is effective for sharp-tailed grouse management because sharp-tailed grouse dancing grounds are often the nexus of this management, yet individual projects may not have the intended desired impact because efforts may be spatially disjunct and target dancing grounds with a varying number of birds and proximity to core sharp-tailed grouse complexes. Priority open-landscapes have been identified in MNDNR forest resource management plans and Area offices do prioritize open-brushland management within these landscapes. However, these efforts could be improved or further justified by spatially modeling connectivity which would help identify multiple pathways or corridors linking open-brushland habitats with dancing grounds.

OBJECTIVES

1. To identify range wide and local pathways or connections among sharp-tailed grouse dancing grounds to aid in prioritizing open-brushland management.

STUDY AREA

I modeled connectivity among dancing grounds identified in MNDNR's 2009 annual sharp-tailed grouse survey across the range of sharp-tailed grouse in Minnesota (Figure 1), but because of computational limitations I split the range into 3 regions: northwest, central, and south. Northwest region included parts if not all of the following counties: Beltrami, Kittson, Lake of the Woods, Marshall, and Roseau. Central region included parts if not all of the following counties: Beltrami, Clearwater, Itasca, Koochiching, and St. Louis. Southern region included parts if not all of the following counties: Aitkin, Carlton, Kanabec, Pine, and St. Louis.

METHODS

I used Circuitscape (version 3.5.1), an open source program that uses circuit theory and is compatible with ArcGIS (Environmental Systems Research Institute, Inc., Redlands, CA, USA), to model habitat connectivity across heterogeneous landscapes (McRae and Shah 2009). Landscapes, in the form of raster datasets, are represented as conductive surfaces, with low resistance values assigned to habitats most conducive to movement, and high resistances assigned to poor dispersal habitat or movement barriers (McRae and Shah 2009). Circuitscape is simply modeling a random walk from the source or point of current injection (i.e., dancing grounds) until a target patch is encountered. Movement probabilities are determined by the conductance or resistance values assigned to each cell. At any given cell, the conductances of the adjacent cells are directly proportional to the probability an animal will move from the cell into one of the adjacent cells (McRae and Shah 2009). Animals are more likely to move into a cell with a higher conductance value. Users supply Circuitscape with a raster habitat map, which is either coded in resistances (with higher values denoting greater resistance to movement) or conductance (higher values indicate greater ease of movement). For this analysis, I coded all land cover with conductance values (Table 1, Figure 1) such that less permeable land cover (i.e., forest) received a low value (e.g., 1); in contrast, highly permeable land cover favored by sharp-tailed grouse (e.g. grass) received the highest value of 100. Habitat patches (Figure 2), or collections of cells serve as the input of current injected into the landscape (McRae and Shah 2009). I defined habitat patches as the area of suitable cover types within a 3.2 km buffer around dancing grounds; lands within this buffer should support annual habitat needs for sharp-tailed grouse (Connelly et al. 1998). However, more work is needed to better refine habitat patches to take into account patch sizes of suitable cover types and areas of suitable habitat that likely would not be used for a variety of reasons (e.g., habitat spurs, long and narrow but not much value). Output from Circuitscape consists of a raster of current flow; areas with greater connectivity have higher current flow values. Because habitat patches serve as the source of current, current flow is maximized at the source and spreads out across the landscape resulting in connective pathways to other dancing grounds or dead ends because of habitat barriers. In addition to finding “pinch points” (i.e., the least cost path), Circuitscape complements least-cost approaches by identifying all possible pathways (i.e., connections) across the landscape (McRae and Shah 2009).

Circuitscape offers four connectivity modeling modes: pairwise, one-to-all, all-to-one, and advanced. I used “all-to-one” which grounds one focal node (i.e., habitat patch) at a time with others are activated. I used focal regions as focal nodes (i.e., habitat patches comprised of suitable cover types within a 3.2 km buffer around a dancing ground). I specified the input habitat raster as conductance and used “connect 4 neighbors” cell connection scheme and calculated average conductance. I output current maps and imported them into ArcGIS.

I used 2009 dancing ground locations identified from annual survey data to develop habitat patches. Habitat patches consist of land cover data derived from several data sources developed for a sharp-tailed grouse dancing ground prediction model (Larson and Bailey, this volume). I used Spatial Analyst to reclassify the land cover layer into conductance values (Table 1). I buffered each dancing ground by 3.2 km and clipped the conductance raster with this buffer. To retain land cover classes most conducive to movement, I reclassified conductance values within the 3.2 km buffer of 100 and 85 to 1 and 2, respectively. Although this reduced the number of land cover classes sharp-tailed grouse are known to use, doing so retained land cover classes that best facilitate movement and connectivity.

RESULTS

Initial modeling suggests connectivity varies among dancing grounds but is greatest among clusters of dancing grounds in northwest Minnesota. In east-central Minnesota, particularly in Aitkin and Carlton Counties, individual dancing grounds occur in highly connected clusters, but connectivity among individual clusters appears limited. Data analyses will include

investigating how connectivity may affect dancing ground persistence and I will evaluate connectivity differences among high versus low count dancing grounds.

ACKNOWLEDGMENTS

B. McRae (National Center for Ecological Analysis and Synthesis, Santa Barbara, California and The Nature Conservancy, Washington) and users of the Circuitscape listserve provided consultations on Circuitscape use and output interpretation. Members of the MNDNR Grouse Committee and Sharp-tailed Grouse Model Team developed the landcover layer used in these analyses.

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- McRae, B. H., and V. B. Shah. 2009. Circuitscape User Guide. Online. The University of California, Santa Barbara. Available at: <http://www.circuitscape.org>. Last accessed: 26 April 2010.

Table 1. Landcover classes used to model sharp-tailed grouse dancing ground connectivity and associated conductance values in Minnesota, 2010.

Land cover	Conductance value
Non-habitat	9999
Cropland	50
Disturbed grass	100
Undisturbed grass	100
Sedge meadow	100
Lowland shrub	85
Bog	75
Forest	1

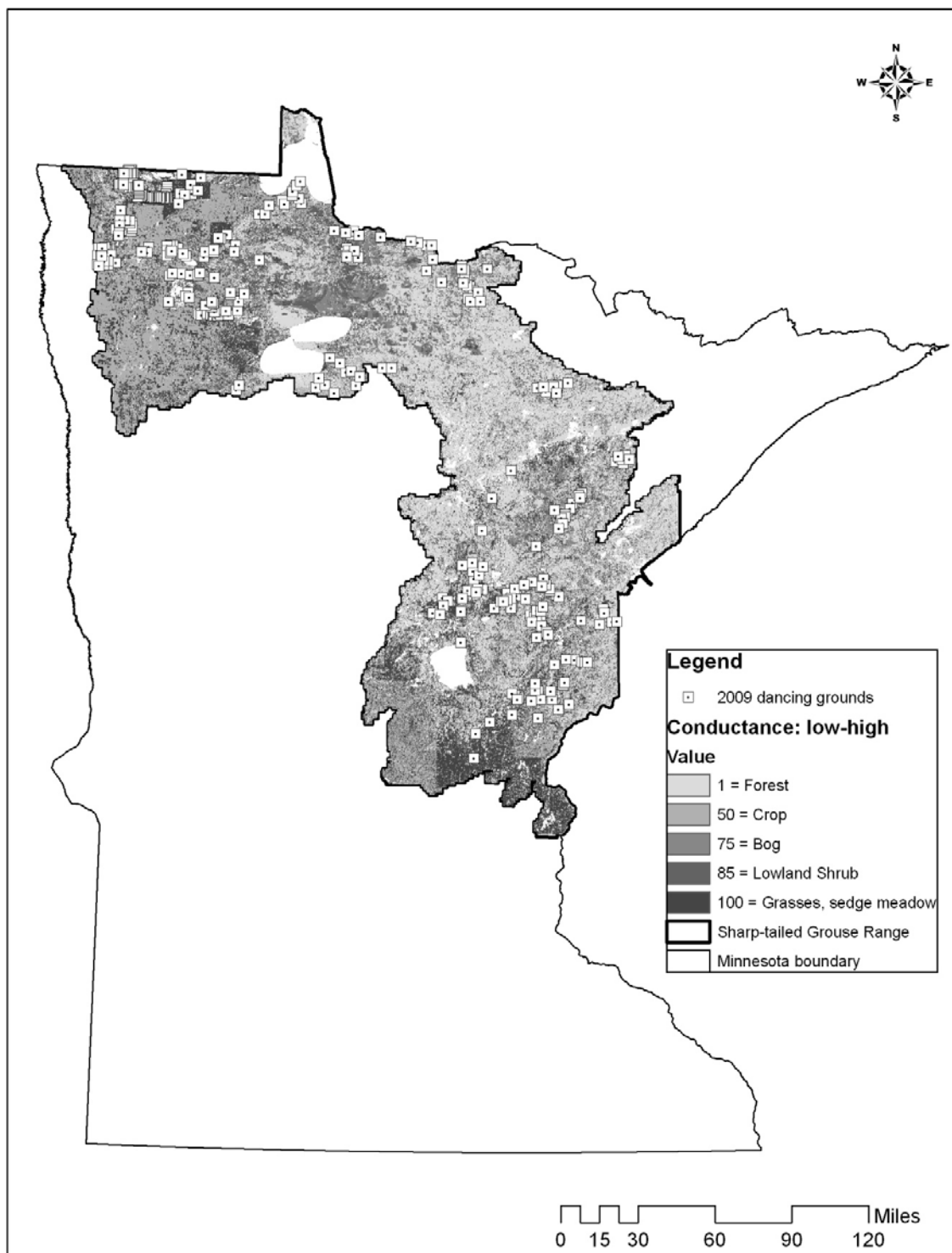


Figure 1. Sharp-tailed grouse dancing ground locations and landcover converted to conductance values to model connectivity in Minnesota, 2010.

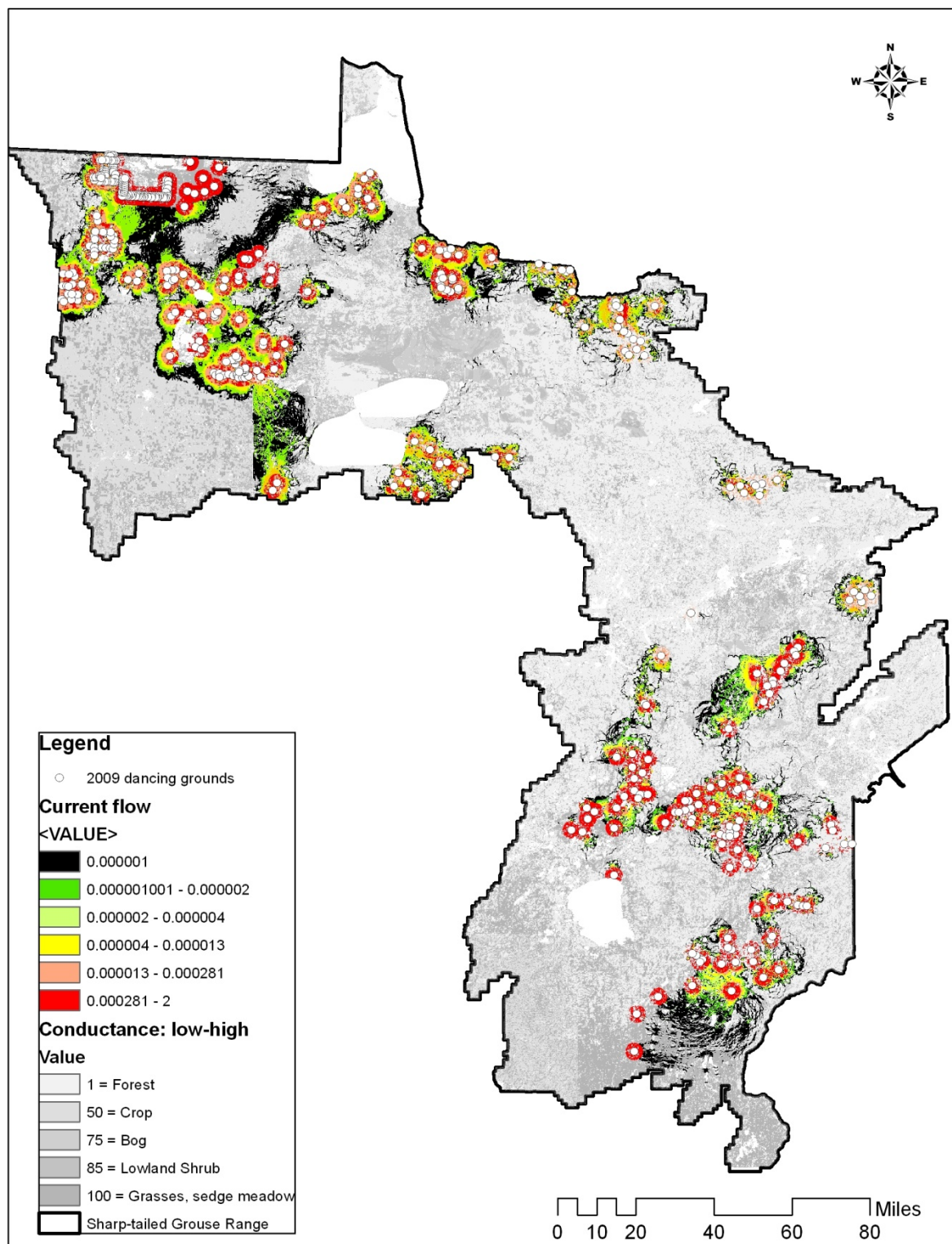


Figure 2. Preliminary output from Circuitscape modeling connectivity as current (low to high) flowing out from dancing grounds across the conductive landscape in Minnesota, 2010.

INCREASING OUR UNDERSTANDING OF THE EFFECTS OF WINTER SEVERITY AND CONIFER COVER ON WINTER DISTRIBUTION, MOVEMENTS, AND SURVIVAL OF FEMALE WHITE-TAILED DEER IN NORTH-CENTRAL MINNESOTA

Glenn D. DelGiudice, Barry A. Sampson, and John Fieberg

SUMMARY OF FINDINGS

The goal of this long-term (1991-2005) investigation was to assess the value of conifer stands as winter thermal cover/snow shelter for white-tailed deer (*Odocoileus virginianus*) at the *population level*. The variation in winter weather during this study period provided a valuable broader context for data examination, interpretation, and understanding than would have been possible in a typical short-term study. Over the course of this 15-year study period, we radiocollared and monitored a total of 452 female deer, including 43 female newborn fawns. On the Inguadona, Shingle Mill, and Dirty Nose study sites, we located radiocollared deer significantly ($P \leq 0.05$) closer to dense conifer cover during severe winters than during winters of mild-average conditions. At the Willow site, where dense conifer cover was most available (almost 25% of the site), a similar difference, albeit insignificant ($P > 0.05$), was apparent. Radiocollared deer also were more likely to be in dense conifer cover as a function of snow depth rather than of ambient temperature. The increasing trend of deer using dense conifer cover as depth of snow cover increased was strongest at Willow and Shingle Mill where conifer stands were most available; the trend was weakest at Dirty Nose where this cover type was least prevalent. At Willow, the probability of deer being in dense conifer cover was greater than 0.5 when depth of snow cover approached 100 cm. Overall, relative to the number of plant species, their diet was highly diverse; however, beaked hazel, mountain maple, and red-osier dogwood accounted for 81.9 and 89.3% of their diet during mild-average and severe winters, respectively. Most typically, mountain maple and red-osier dogwood were selected (proportion of overall use was >overall proportional availability) for by deer, whereas, beaked hazel, although co-dominant in their diet, was used in proportion to availability. The category “other species” consisted of about 24 browse species, and on average accounted for 28.8-35.4% and 17.7-33.8% of their diet during mild-average winters and severe winters, respectively. However, use of the “other species” category decreased ($P \leq 0.05$) by 48% and 42% during severe winters compared to mild-average winters on the Willow and Dirty Nose sites, respectively, suggesting that the diversity of their diet decreased during severe winters. We noted significant ($P \leq 0.05$) differences between mean UN:C ratios during mild-average versus severe winters on all 4 sites. From the perspective of the deer’s physiological response to winter conditions, we would consider WSIs of 124-126 to be reflective of conditions less than severe. Serious nutritional restriction was most common (indicated by UN:C ratios of 18-20% of snow-urine samples) during severe winters at the Willow and Shingle Mill sites where dense conifer cover was most available and where deer were most likely to be using this cover. In our ongoing, more in-depth data analyses we will examine the individual and interactive effects of specific components of winter conditions, conifer availability, timber harvesting activities, and stand regeneration on habitat use, food habits, nutritional status, and survival.

INTRODUCTION

The goal of this long-term investigation was to assess the ecological value of conifer stands as winter thermal cover or snow shelter for white-tailed deer at the *population level*. This study was prompted directly by an increasing need of the Minnesota Department of Natural Resources’ (MNDNR) wildlife managers for information regarding the habitat requirements of white-tailed deer in the forest zone of the state. Expanding our understanding of their habitat requirements and ecology during all seasons in relatively complex ecosystems impacted frequently by significant natural and human-related forces is critical to effective population management. It is also essential to the wildlife manager’s ability to provide meaningful input to

coordinated long-term forest management strategies and the short-term activities that immediately and dramatically alter deer habitat. Both white-tailed deer and the forests of the Great Lakes region are highly regarded for their recreational value and have notable positive impacts on local and state economies.

Because winter is the most nutritionally challenging season for northern deer, the season when most natural mortality of adults (1.0 year old) occurs and when nutritional restriction of the season may impose the greatest overall negative impact on population performance, focus on winter habitat requirements is often considered paramount. For northern deer, conifer stands specifically may play a critical role in the winter energy balance of deer, and ultimately in their survival, but when...during all winters, during winters of particularly cold ambient temperatures, deep snow cover, or both?

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, and this in part, has increased management's need for a better understanding of the value of this cover type to deer. The level of logging of all tree species collectively, and conifer stands specifically, has recently reached the estimated allowable harvest. Land management agencies and commercial landowners commonly restrict 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 anticipate 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. This need has been reinforced by increasing information about the potential effects of climate change on northern forest ecosystems in Minnesota, including a shift northward of spruce-fir forests (Iverson and Prasad 2001, Hansen et al. 2003), as well as a pronounced decline in lowland coniferous forests and the potential benefits they afford as snow shelter and thermal cover. According to MNDNR (2008), "wildlife associated with coniferous forests may be under the greatest threat of extirpation from Minnesota due to climate change."

OBJECTIVES

Expecting that environmental variation, particularly in winter weather conditions, would have biologically significant influences on various aspects of deer ecology, we knew a long-term study would enhance our ability to examine and understand these influences and the importance of conifer cover as a habitat component (DeGiudice and Riggs 1996). We hypothesized that winter severity and conifer availability affect the use of moderately dense (40-69% canopy closure [Class B]) and dense $\geq 70\%$ canopy closure [Class C]) conifer stands on winter range 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 and the population performance (survival and reproduction) of deer. Relative to varying winter severities, the objectives of the comprehensive approach of this study have been to:

1. Monitor deer movements (i.e., migration) between seasonal ranges and on winter ranges by very high frequency (VHF) radio-telemetry and Global Positioning System (GPS) collars to assess spatial distribution;
2. Determine habitat composition of winter range study sites and deer use of conifer cover types;
3. Monitor winter food habits;
4. 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);
5. Monitor age-specific survival, cause-specific mortality, and reproduction; and
6. 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) included 4 winter range study sites (Willow, Inguadona, Shingle Mill, and Dirty Nose), located in the Grand Rapids-Remer-Longville area of north-central Minnesota; they range from 13 to 23.6 km² (5-9.1 mi²) in area (Table 1). Conifer stands on the sites primarily included balsam fir (*Abies balsamea*), northern white cedar (*Thuja occidentalis*), black spruce (*Picea mariana*), and jack and red pine (*Pinus banksiana* and *P. resinosa*). Common browse species were beaked hazel (*Corylus cornuta*), mountain maple (*Acer spicatum*), sugar maple (*A. saccharum*), red-osier dogwood (*Cornus stolonifera*), and ironwood (*Ostrya virginiana*). 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. We applied an experimental treatment (timber harvest) 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]) on the Inguadona and Shingle Mill sites midway through the study; limited, unplanned decreases of conifer cover occurred on all 4 sites over the 15-year period (Figure 1). Mean area of conifer canopy closure classes A, B, and C differed markedly among the 4 sites (Table 1). During the 15-year study, availability of dense conifer cover was greatest on the Willow (23.2%) and Shingle Mill (16.5%) sites (Figure 1). The most pronounced reduction (percentage) in dense conifer cover as the study progressed occurred on the Inguadona site (Figure 1). The temporal variations in conifer cover and differences among sites are proving to be of notable value to many of our analyses.

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, ArcMap 9.3.1) for temporal and spatial analyses; (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 approximately 20 constituents, (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, Use of Conifer Cover, Nutrition, and Survival of White-Tailed Deer

Weather is one of the strongest environmental forces impacting wildlife populations. Our 15-year study period allowed us to capture a wide breadth of variation in the severity of winter weather conditions, including 2 back-to-back historically severe winters (1995-1996, 1996-1997), followed by 3 consecutive, unprecedented mild winters in more than 100 years of weather data collection (P. Boulay, Minnesota State Climate Office, personal communication), as well as many of mild to average conditions. The MNDNR's maximum winter severity index (WSI, 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 ≥ 38 cm during November -May) ranged from 42 to 195. This long-term variation in winter weather provided a valuable broader context for data

examination, interpretation, and understanding than would have been possible in a typical short-term study.

In an effort to assess the importance of dense conifer cover to deer, we employed ArcGIS (Version 9.3.1) to measure 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. On the Inguadona, Shingle Mill, and Dirty Nose sites, we located radiocollared deer significantly ($P \leq 0.05$, comparison of 95% confidence limits [$2 \times SE$]) closer to dense conifer cover (Class C) during severe winters ($WSI \geq 124$) than during winters of mild-average conditions (Table 2). At Willow, where dense conifer cover was most available (almost 25% of the site), a similar difference, albeit insignificant ($P > 0.05$), was apparent (Table 2). Importantly, using ArcGIS to generate 5,000 randomly located points annually within each study site showed that the availability and distribution of dense conifer cover did not influence the differences in the nearest distance to dense conifer cover during mild-average or severe winters, rather this appeared to be behavioral selection by deer in response to differences in winter conditions. When we examined nearest distance of radiocollared deer to moderately dense (Class B) or dense (Class C) conifer cover, mean distances were shorter than relative to Class C alone, as would be expected, but the differences between mild-average and severe winters were significant ($P \leq 0.05$) at Shingle Mill and Dirty Nose, but not at Willow and Inguadona (Table 2). Again, examination of random points indicated that “nearest distances” of deer were a result of behavioral responses rather than availability or distribution of these conifer stands. During mild-average and severe winters, mean “nearest distances” of deer to conifer cover at Willow were significantly ($P \leq 0.05$) shorter than at Inguadona, Shingle Mill, and Dirty, where they were quite similar (Table 2).

Our analyses also showed that radiocollared deer were more likely to be in dense conifer cover as a function of snow depth rather than of ambient temperature (Figures 2 and 3). The increasing trend of deer using dense conifer cover as depth of snow cover increased was strongest at Willow and Shingle Mill where conifer stands were most available; the trend was weakest at Dirty Nose where this cover type was least prevalent (Figure 2). At Willow, the probability of deer being in dense conifer cover was greater than 0.5 when depth of snow cover approached 100 cm. Daily minimum ambient temperature exhibited no consistent influence on deer use of dense conifer cover at any of the 4 sites (Figure 3). Similarly, we had previously reported that WSI and snow depth had significant negative effects on winter survival of our radiocollared deer, whereas ambient temperature exhibited no influence (DeGiudice et al. 2002, 2006). Future work is planned to enhance the rigor of our analytical approach and will include a simulation study (J. Fieberg and J. Schildcrout, Department of Biostatistics, Vanderbilt University) designed to compare regression methods for correlated binary data and provide insights into the performance of these estimators when applied to highly imbalanced data and small sample sizes, as observed in the present study. In addition to further analyses of the potential effects of minimum ambient temperature and snow depth on deer use of conifer cover, we will examine potential influences of changes in conifer availability associated with our experimental timber harvests.

Our 14-year monitoring of winter food habits of white-tailed deer on the 4 sites showed that, overall, relative to the number of plant species, their diet was highly diverse; however, beaked hazel, mountain maple, and red-osier dogwood accounted for 81.9 and 89.3% of their diet during mild-average and severe winters, respectively (Table 3). Most typically, mountain maple and red-osier dogwood were selected (proportion of overall use was >overall proportional availability) for by deer, whereas, beaked hazel, although co-dominant in their diet, was used in proportion to availability (Table 3). The category “other species” consisted of about 24 browse species, and on average accounted for 28.8-35.4% and 17.7-33.8% of their diet during mild-average winters and severe winters, respectively, on the 4 sites (Table 3). Diet diversity is critical to the deer’s ability to maintain its nutritional status during winter (Verme and Ullrey 1972). Use of the “other species” category decreased ($P \leq 0.05$) by 48% and 42% during severe winters compared to mild-average winters on the Willow and Dirty Nose sites,

respectively, suggesting that the diversity of their diet decreased during severe winters. At Willow, where deer were most likely to be in dense conifer cover during severe winters of deep snow, mean proportional use of mountain maple also declined (28%, $P \leq 0.05$), as did use of red-osier dogwood (47%), although not significantly so due to greater variability. Deer made significantly ($P \leq 0.05$) greater use (up to 71%) of beaked hazel at all sites during severe winters, except at Willow, where the increase was less pronounced. Deer typically made relatively low use of paper birch (*Betula papyrifera*) and trembling aspen (*Populus tremuloides*) at all 4 sites, but in proportion to their availabilities.

Winter nutritional restriction or deprivation of white-tailed deer and other northern ungulates 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). Overall, we documented significant ($P = 0.057$ and $P = 0.013$) relationships between maximum WSIs and percent of snow-urine samples collected during each winter with urea nitrogen:creatinine (UN:C) ratios indicative of severe nutritional restriction (UN:C ≥ 3.5 mg:mg, Figure 4) and between the latter and percent winter mortality of radiocollared deer (Figure 5). Interestingly, we noted significant ($P \leq 0.05$) differences between mean UN:C ratios during mild-average versus severe winters on all 4 sites when we included winters 1992-1993 (WSI = 124) and 1993-1994 (WSI = 126) in the mild-average category (Table 4), as opposed to including these winters in the severe winter category. Additionally, the differences in the percentage of samples collected that were indicative of severe nutritional restriction was more apparent when winters 1992-94 and 1993-94 were categorized as mild-average (Table 4). So from the perspective of the deer's physiological response to winter conditions during these 2 winters, we would consider WSIs of 124-126 to be reflective of conditions less than severe. Serious nutritional restriction was most common (indicated by UN:C ratios of 18-20% of snow-urine samples, Table 4) during severe winters at the Willow and Shingle Mill sites where dense conifer cover was most available (Figure 1) and where deer were most likely to be using this cover (Figure 2).

The preliminary findings presented herein revealed a number of biologically significant quantifiable responses to winter severity by deer with respect to their use of conifer cover, food habits, metabolic physiology and nutritional status, as well as to survival, reproduction, and migration patterns (DelGiudice et al. 2002, 2006, 2007; Fieberg et al. 2008; Carstensen et al. 2009). In our ongoing, more in-depth data analyses we will examine the individual and interactive effects of specific components of winter conditions, conifer availability, timber harvesting activities, and stand regeneration on habitat use, food habits, nutritional status, and survival.

ACKNOWLEDGMENTS

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

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Table 1. Mean area of 4 study sites and conifer canopy closure classes "A" (40%), "B" (41 -69%) and "C" (≥70%) within their boundaries, north-central Minnesota, winters 1990-1991 to 2004-2005.

Site	Mean		Area of canopy closure class					
			"A"		"B"		"C"	
	mi ²	km ²	ha	%	ha	%	ha	%
Willow	7.6	19.6	296	15.16	131	6.71	453	23.18
Dirty Nose	5.0	13.0	466	35.79	114	8.75	80	6.18
Inguadona	9.1	23.6	744	31.53	257	10.89	1,029	8.47
Shingle Mill	8.7	22.6	343	15.19	244	10.80	373	16.52

Table 2. Nearest distance of radiocollared, female white-tailed deer to conifer cover with canopy closures of at least 70% ("C") or 41-69% ("B") on 4 study sites during mild to average winters versus severe winters, north-central Minnesota, winters 1993-1994 to 2004-2005.¹

Site	Nearest distance to canopy closure class (m)				
		“C”		“C” or “B”	
Winter severity	N	Mean	SE	Mean	SE
Willow					
Mild-average	622	100	7.2	52	4.2
Severe	667	83	5.5	40	3.0
Inguadona					
Mild-average	764	243	7.3	102	5.3
Severe	822	192	6.7	95	4.9
Shingle Mill					
Mild-average	668	354	10.6	196	6.6
Severe	771	185	6.6	94	4.4
Dirty Nose					
Mild-average	550	240	8.5	116	4.9
Severe	517	168	7.7	81	4.2

¹Winters of mild-average severity (winter severity indices [WSI] ≤ 108) included winters 1994-1995, 1998-1999, 1999-2000, and 2001-2002 to 2004-2005, and severe winters (WSIs ≥ 124) included winters 1993-1994, 1995-1996, 1996-1997, and 2000-2001.

Table 3. Browse availability and use by white-tailed deer on 4 study sites during mild-average versus severe winters, north-central Minnesota, winters 1991-1992 to 2004-2005.¹

Site Species categories	Use (%)			Availability (%)		
	N	Mean	SE	N	Mean	SE
Willow						
Mild-average winters						
Mountain maple	148	50.7	2.8	181	28.4	2.2
Red-osier dogwood	56	19.8	3.6	181	2.6	0.7
Beaked hazel	160	17.9	1.7	181	24.9	1.7
Paper birch	87	2.6	0.6	181	1.7	0.3
Trembling aspen	72	3.8	1.2	181	2.1	0.5
"Other" species	180	33.9	2.3	181	40.4	1.9
Severe winters						
Mountain maple	122	36.4	2.1	126	43.9	2.7
Red-osier dogwood	27	10.5	4.1	126	1.0	0.6
Beaked hazel	115	23.8	2.1	126	27.2	2.2
Paper birch	56	2.7	0.5	126	1.7	0.3
Trembling aspen	46	4.1	1.0	126	2.0	0.7
"Other" species	125	17.7	1.6	126	24.1	1.8
Inguadona						
Mild-average winters						
Mountain maple	127	19.7	2.0	200	4.6	0.6
Red-osier dogwood	52	8.9	1.9	200	0.8	0.2
Beaked hazel	197	43.8	2.0	200	51.8	1.8
Paper birch	131	5.5	0.8	200	3.6	0.5
Trembling aspen	148	11.9	1.4	200	9.2	1.1
"Other" species	199	28.8	1.7	200	29.9	1.5
Severe winters						
Mountain maple	71	10.5	1.7	128	2.8	0.5
Red-osier dogwood	37	8.4	2.3	128	1.4	0.7
Beaked hazel	126	59.0	2.1	128	57.3	2.1
Paper birch	80	4.7	1.0	128	3.1	0.7
Trembling aspen	102	10.2	1.6	128	11.9	1.6
"Other" species	126	23.0	1.6	128	23.4	1.5
Shingle Mill						
Mild-average winters						
Mountain maple	115	38.7	2.8	152	15.8	1.7
Red-osier dogwood	56	17.0	2.8	152	2.0	0.5
Beaked hazel	140	26.8	2.1	152	32.2	2.0
Paper birch	66	1.5	0.4	152	1.3	0.3
Trembling aspen	84	6.9	1.4	152	3.7	0.8
"Other" species	152	35.4	2.1	152	45.0	2.0
Severe winters						
Mountain maple	90	23.0	2.5	125	11.3	1.5
Red-osier dogwood	21	6.8	3.0	125	0.6	0.3
Beaked hazel	117	44.9	2.6	125	41.8	2.6
Paper birch	50	3.1	0.6	125	1.6	0.3
Trembling aspen	77	7.7	1.6	125	5.6	1.1
"Other" species	124	33.8	2.3	125	39.1	2.3

Table 3. Continued.

Site Species categories	Use (%)			Availability (%)		
	N	Mean	SE	N	Mean	SE
Dirty Nose						
Mild-average winters						
Mountain maple	113	22.3	2.3	149	6.3	0.9
Red-osier dogwood	76	20.8	2.6	149	3.2	0.5
Beaked hazel	147	32.2	2.0	149	45.7	2.0
Paper birch	74	3.4	0.7	149	2.1	0.5
Trembling aspen	105	8.3	1.3	149	5.4	0.8
"Other" species	149	33.2	2.0	149	37.3	1.8
Severe winters						
Mountain maple	93	22.7	2.4	123	9.6	1.4
Red-osier dogwood	43	18.9	4.1	124	2.5	0.6
Beaked hazel	121	52.3	2.5	124	54.1	2.5
Paper birch	54	4.2	1.1	124	1.7	0.3
Trembling aspen	88	6.2	1.3	124	5.8	1.2
"Other" species	123	19.4	1.8	124	26.2	2.1

¹Winters of mild-average severity (winter severity indices [WSI] ≤ 108) included winters 1994-1995, 1998-1999, 1999-2000, and 2001-2002 to 2004-2005, and severe winters (WSIs ≥ 124) included winters 1993-1994, 1995-1996, 1996-1997, and 2000-2001.

Table 4. Mean urea nitrogen:creatinine (UN:C) ratios in urine recently voided (≤ 72 hr) in snow by white-tailed deer and percent of samples indicative of severe nutritional restriction (UN:C ≥ 3.5 mg:mg) on 4 study sites during mild-average versus severe winters, north-central Minnesota, winters 1992-1993 to 2004-2005.¹

Site	Urinary UN:C ratios					
Species categories	<i>N</i>	Mean	SE	Range	Percent of samples with UN:C ≥3.5 g:mg	
Willow						
Mild-average winters	621	2.0	0.13	0.2 - 62.0	6.28	7.31 ²
Severe winters	388	2.8	0.22	0.3 – 51.9	18.04	13.96 ²
Inguadona						
Mild-average winters	636	1.6	0.05	0.1 – 15.7	5.66	7.22 ²
Severe winters	368	2.5	0.29	0.4 – 81.9	9.24	6.75 ²
Shingle Mill						
Mild-average winters	564	2.1	0.12	0.2 – 48.9	8.16	8.45 ²
Severe winters	370	2.7	0.07	0.2 – 12.5	20.00	16.35 ²
Dirty Nose						
Mild-average winters	586	1.6	0.08	0.1 – 43.3	5.80	7.62 ²
Severe winters	368	3.1	0.44	0.4 – 132.7	11.41	8.25 ²

¹Winters of mild-average severity (winter severity indices [WSI] ≤ 126) included winters 1994-1995, 1998-1999, 1999-2000, and 2001-2002 to 2004-2005, and severe winters (WSIs ≥ 153) included winters 1993-1994, 1995-1996, 1996-1997, and 2000-2001.

²These percentages were recalculated with winters 1992-1993 and 1993-1994 included as severe winters, rather than as mild-average winters.

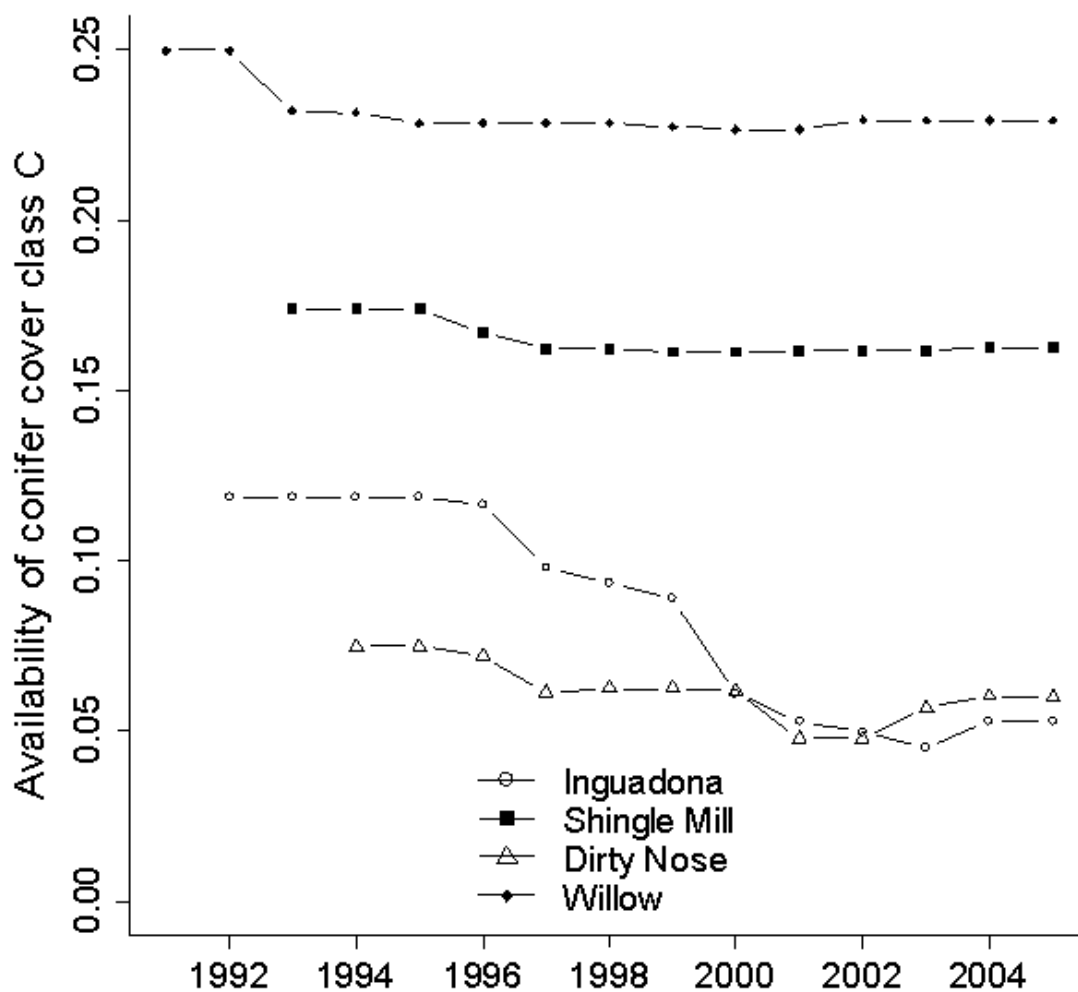


Figure 1. Changes in the availability of conifer cover with canopy closure of at least 70 percent within the 4 study sites of the white-tailed deer/winter cover study, north-central Minnesota, winters 1990-1991 to 2004-2005.

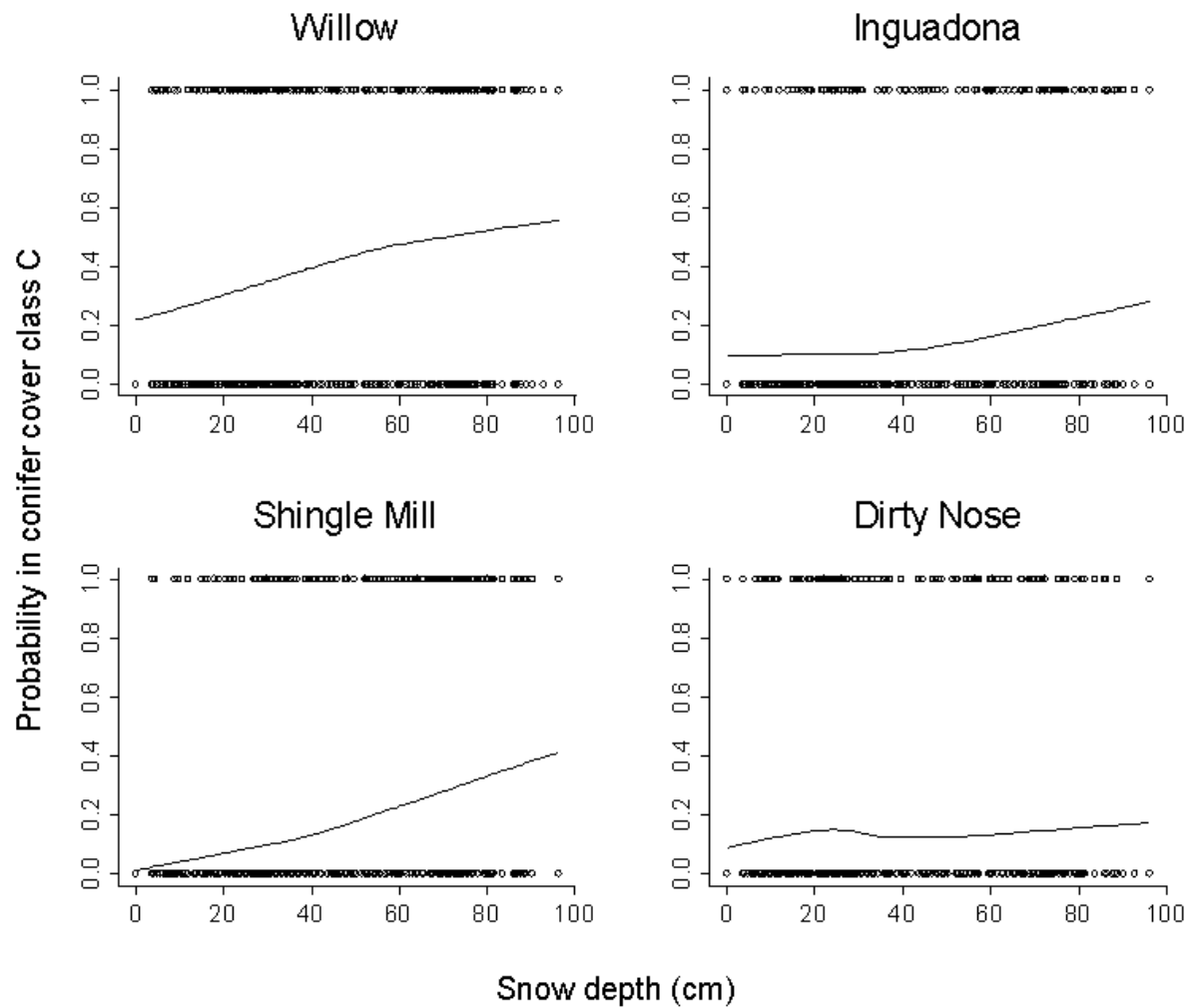


Figure 2. Probability of radiocollared deer being in conifer cover with canopy closures of at least 70% as a function of snow depth on the 4 study sites, north-central Minnesota, winters 1993-1994 to 2004-2005. (Small circles at the bottom and top of graphs represent the density of data collected.)

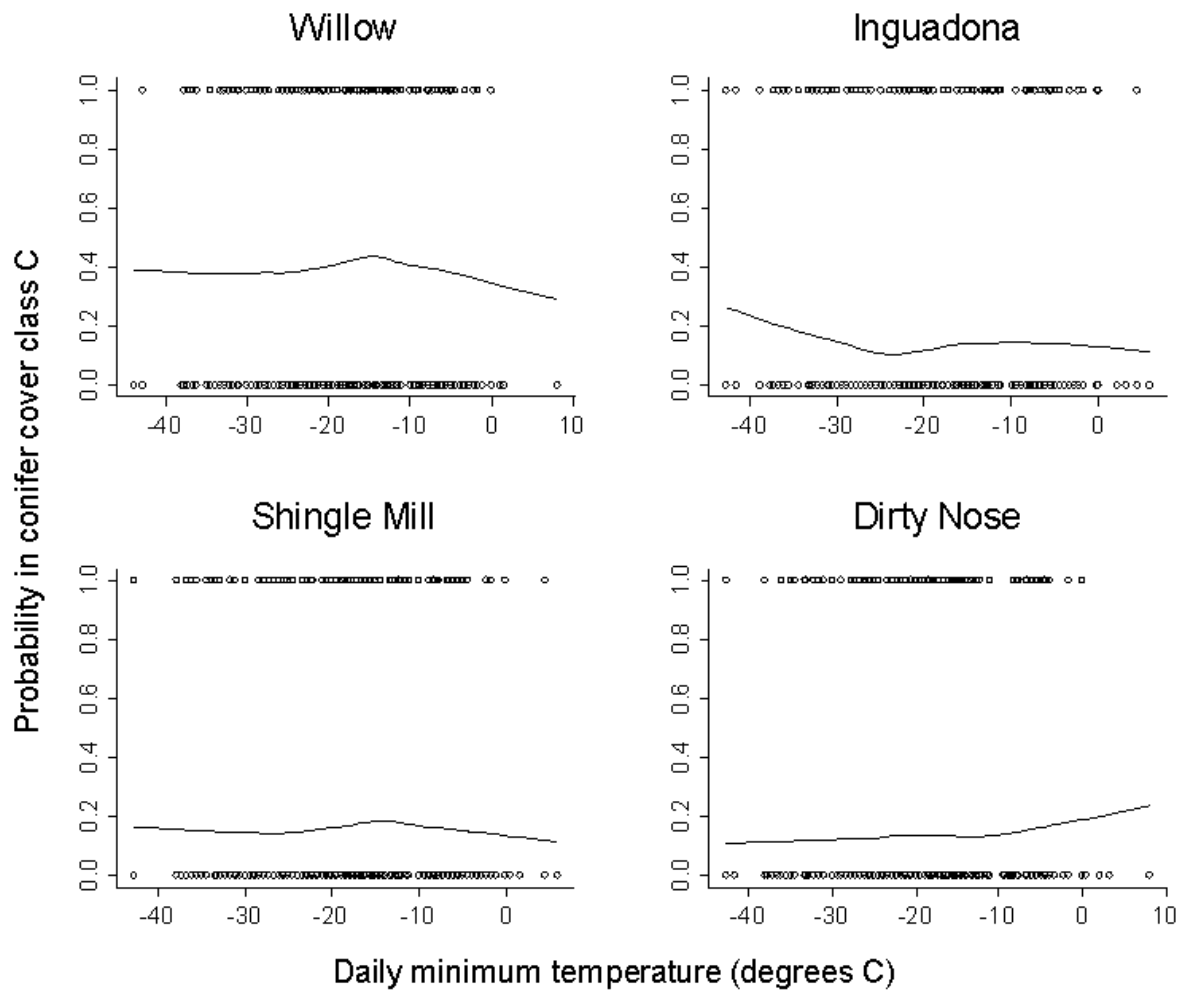


Figure 3. Probability of radiocollared deer being in conifer cover with canopy closures of at least 70% as a function of daily minimum temperature on the 4 study sites, north-central Minnesota, winters 1993-1994 to 2004-2005. (Small circles at the bottom and top of graphs represent the density of data collected.)

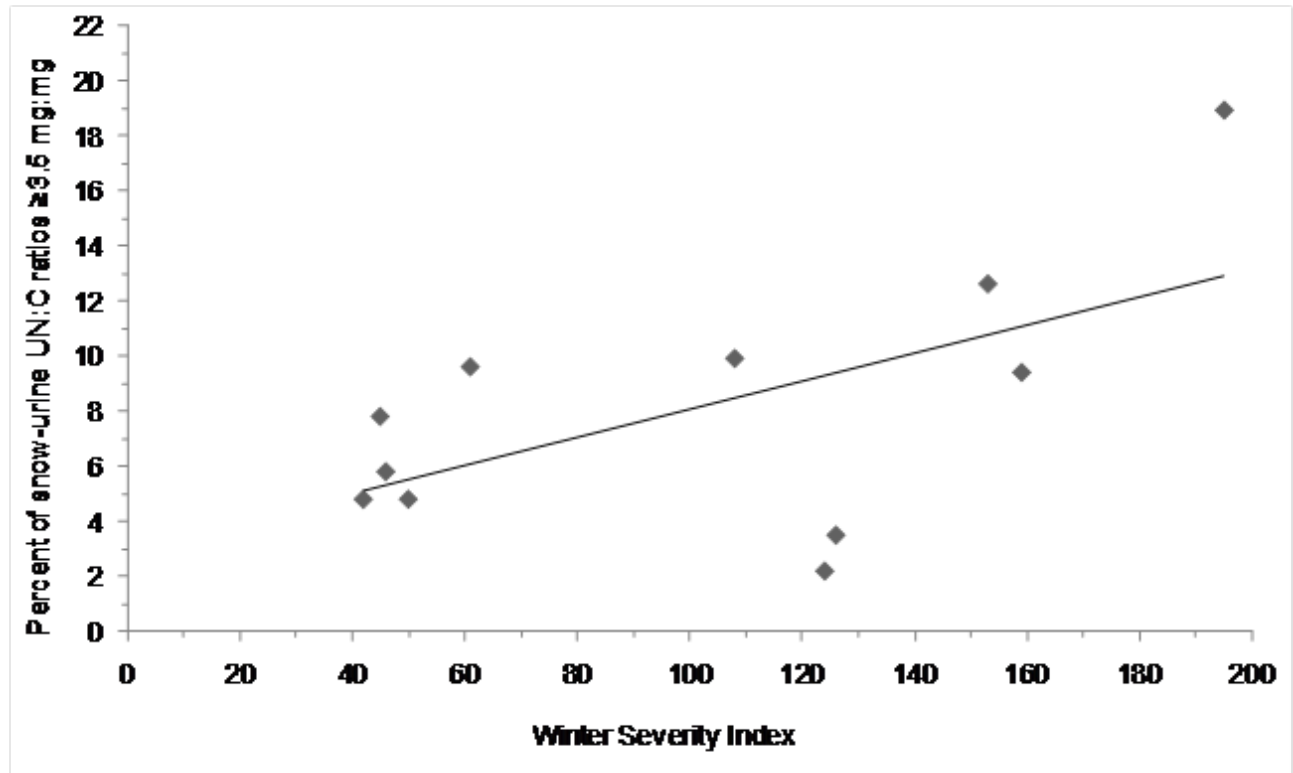


Figure 4. Relationship ($r^2 = 0.35$, $y = 2.958 + 0.051x$, $P = 0.057$) of the annual maximum winter severity index (see text for definition) to the percent of urine samples in snow (snow-urine) of white-tailed deer with urea nitrogen:creatinine (UN:C) ratios indicative of severe nutritional restriction (≥ 3.5 mg:mg), all 4 study sites (pooled), north-central Minnesota, winters 1992-1993 to 1998-1999, 2000-2001, 2001-2002, 2003-2004, and 2004-2005.

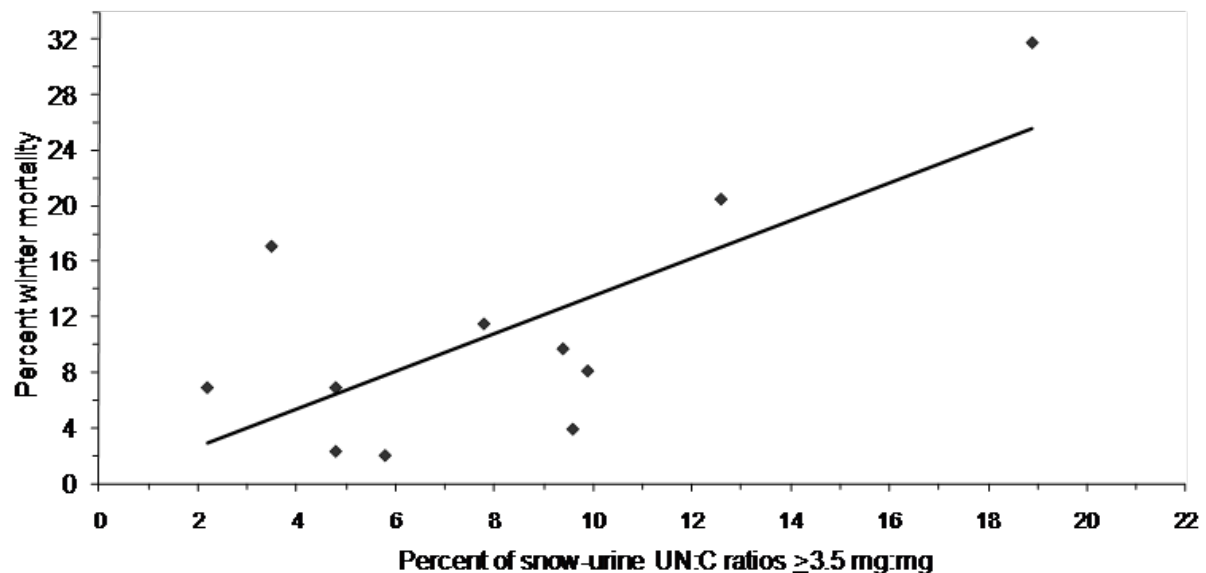


Figure 5. Relationship ($r^2 = 0.52$, $y = 3.942 + 0.381x$, $P = 0.013$) of the annual percent of urine samples in snow (snow-urine) of white-tailed deer with urea nitrogen:creatinine (UN:C) ratios indicative of severe nutritional restriction (≥ 3.5 mg:mg) to percent winter mortality, all study 4 study sites (pooled), north-central Minnesota, winters 1992-1993 to 1998-1999, 2000-2001, 2001-2002, 2003-2004, and 2004-2005.