Forest Wildlife Populations and Research Group

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ECOLOGY AND POPULATION DYNAMICS OF BLACK BEARS IN MINNESOTA

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SUMMARY OF FINDINGS

During April 2011–March 2012, we monitored 38 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. Most of the focus of this study has been in the NW site in recent years. Hunting has been the primary source of mortality in all areas, but with a concerted effort to discourage hunters from shooting collared bears, and by clearly marking bears with large ear tags, only 1 collared bear was known to be killed by a Minnesota bear hunter in fall 2011. However, the radiocollared sample was diminished by other human-caused mortalities and radiocollar failures. Reproduction was highest in the NW study site. Data from Global Positioning System (GPS)-radiocollars indicated that males in the NW made significant use of crop fields (corn and sunflowers) from August to October. By contrast, females in this area rarely used crops, but instead spent much of their time in aspen woodlands and shrublands. Thus, the high reproductive output of females is not due to the crops, but to an abundance of natural food in the generally small woodlots that are scattered across this area (on both public and private lands). Analysis of stable isotopes in hair samples were useful in determining the reliance of individual bears on different key foods, especially distinguishing those that fed largely on corn or sunflowers, the 2 main crops consumed by bears in this area. Continuation of this work will aim to predict the extent to which bears can continue expanding westward.

INTRODUCTION

The Minnesota Department of Natural Resources (MNDNR) initiated research on black bears in 1981, spurred by concerns of low population size and over-harvest. This occurred commensurate with a management program designed to restrict the harvest via a quota on bear hunting licenses. For the first 10 years, the bear study was limited to the Chippewa National Forest (CNF), near the geographic center of the Minnesota bear range (Figure 1). Later, we started satellite bear projects in other study sites with different habitat conditions. 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).

The CNF is one of the most heavily hunted areas of the state, with large, easily-accessible tracts of public (national, state, and county) 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 cornfields 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 in the other sites.

In 2007 we initiated work in a fourth 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 crop fields, like corn and sunflowers, that bears consume), (2) most of the land, including various small woodlots, is privately-owned, with some

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larger blocks of forest contained within MDNR Wildlife Management Areas (WMAs) and a National Wildlife Refuge (NWR); (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. Quantify temporal and spatial variation in cub production and survival;
2. Quantify rates and sources of mortality;
3. Assess body condition indices across sites and years (not covered in this report);
4. Evaluate habitat use (especially 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 with their radiocollared mother. We trapped bears this year only in the NW study site, using barrel traps baited with raw bacon. Traps were checked daily, mainly by local volunteer residents. We anesthetized captured bears with Telazol and fitted them with GPS-collars, programmed to collect locations at designated intervals (varying from 6 hr to 20 min, depending on time of year). We used both GPS “pods” (Telemetry Solutions, Concord, CA) that were bolted onto standard VHF collars, and also GPS-Iridium collars (Vectronic Aerospace, Berlin, Germany). The latter collars uploaded location data to an Iridium satellite, which was then transmitted to us daily by email. The location data stored in the pods were retrievable only by physically connecting the pod to a computer; typically, we took pods off bears when they were denning.

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. We measured lengths and girths, body weight, and biolelectrical impedance (to calculate percent body fat), and took blood and hair samples. We changed or refit the collar, and attached a first collar on some yearlings. All collared bears were given brightly-colored, cattle-size ear tags (7x6 cm; Dalton Ltd., UK) that would be plainly visible to hunters. Additionally, collaborators from the University of Minnesota (Dr. Paul Iaizzo) and Medtronic, Inc. (Dr. Tim Laske) measured heart condition with a 12-lead EKG and ultrasound on a select sample of bears, and implanted (subcutaneously) a miniature heart monitoring device (developed for humans: Reveal®, Medtronic, Inc, Minneapolis, MN) to 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.

We did not monitor survival of bears during the summer. Mortalities, though, were reported to us when bears were shot as a nuisance, hit by a car, or killed by a hunter. Prior to the hunting season (1 September–mid-October), hunters were mailed a letter requesting that they not shoot collared bears with large ear tags.

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, including types of crop fields.

Hair samples from collared bears were analyzed for stable isotopes of carbon (C) and nitrogen (N) (Colorado Plateau Stable Isotope Laboratory, Northern Arizona University, Flagstaff, AZ) to estimate the relative contribution of different types of foods, especially crops in the bears' diets. We also obtained bear hair samples from successful hunters in the NW study site for stable isotope analysis. We collected various types of bear foods from the NW study site, including herbaceous vegetation, fleshy fruits, nuts, ants, deer, corn, soybeans, and sunflowers, and obtained their isotopic signatures for C and N (Department of Geology and Geophysics, University of Minnesota, Minneapolis, MN). We used the Stable Isotope Analysis package in Program R (SIAR) to solve mixing models for the isotopic data within a Bayesian framework, and thereby generated distributions for the probabilities that different individual bears consumed and assimilated given proportions of certain types of foods.

RESULTS AND DISCUSSION

Radiocollaring and Monitoring

Since 1981 we have handled >800 individual bears and radiocollared >500. As of April 2011, the start of the current year's work, we were monitoring 30 radiocollared bears: 4 in the CNF, 7 at Camp Ripley, 3 in VNP, and 16 in the NW. We captured and collared 8 new bears in the NW study site during May−August (4 males, 4 females). All were caught near the western edge of the bear range. We also collared 3 female yearling bears and 1 adult bear in dens (1 bear in each of the 4 study sites) during March, 2012 (Table 1).

Mortality

Legal hunting has been the dominant cause of mortality among radiocollared bears from all study sites: since 1981, 77% of mortalities that we observed were due, or likely due to hunting (Table 2). 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 shifted to reproduction and habitat use rather than mortality, beginning in 2001 we sought to protect the remaining sample of bears by asking hunters not to shoot them and marking them with easily-visible ear tags (Figure 2).

This year, we confirmed only 1 collared bear that was killed by a Minnesota hunter (NW study site). Three other NW collared bears disappeared during the fall. One of these, though, was photographed by a trail camera the next spring (Figure 2); the photograph showed that the bear still wore the radiocollar, which had evidently failed. Two GPS-Iridium collars also stopped functioning, although we later located one of these bears in a den site. That bear had died in mid-October just after it had entered the den, based on the record from the implanted heart monitor, suggesting that it had been shot outside the den at the end of the bear hunting season. However, the carcass was too frozen to verify the cause of death.

Other collared bears in the NW were lost from our monitored sample for a variety of reasons (Table 1), including 1 that was legally killed in Manitoba during the spring bear hunt there, 2 that were killed as nuisances (1 at a beehive, 1 at a residence), and 1 struck by a car on a highway. In all, although we added 9 collared bears to the NW sample, we lost 13, so we had 4 less bears in our monitoring sample in April 2012 (3 males, 9 females) than the previous year (Table 1).

None of the collared bears in the other 3 study sites died. One of these, a CNF female that was collared as a 7-year-old in 1981, lived through 2011. She reused a former den, and emerged in mid-March 2012, as a 38-year-old, the oldest known wild black bear in North America (ever). We used a trail camera to document the date of her den emergence, which was especially early due to unusually warm weather in March (Figure 3).

Reproduction

We visited 7 dens of females with litters containing 17 cubs during March, 2012. Since 1982, we have checked 258 litters with 661 cubs (2.6 cubs/litter), of which 52% were male (Tables 3–6). Only VNP showed no indication of a male-biased sex ratio at birth (Table 5). Mortality of cubs during their first year of life averaged 21%, with mortality of male cubs (26%) exceeding that of females (16%; $\chi^2 = 7.2$, $P < 0.01$). The timing and causes of cub mortality are unknown.

Reproductive rates were highest in the NW study area, and lowest in VNP (Figure 4). The reproductive rate (cubs/female 4+ years old) combines litter size, litter frequency, and age of first reproduction into a single parameter. Reproductive rate was higher for 7+ year-old bears than 4–6 year-old bears because many bears in this younger age group either had not yet reproduced or had their first litter, which tended to be smaller. Regional variation in reproductive rates of older bears relates to effects of food supply on litter size and litter interval. Litter size averaged ≥3.0 cubs only for 7+ year-olds in the NW.

Habitat and Crop Use by NW Bears

During spring and summer, both male and female GPS-collared bears in the NW study site most frequently used forests dominated by aspen; on average, they spent 35–50% of their time in aspen forests, whereas this forest type comprised only 13% of the regional area, in a patchy mosaic (e.g., small WMAs, Agassiz NWR, and private lands). Lowland shrub was the second-most used habitat type (20–30% frequency of use) during these seasons. In fall, females continued to use primarily these same 2 habitat types; surprisingly, they used oak forests (primarily bur oak [*Q. macrocarpa*], which comprised ~2% of the landscape) <10% of the time. Also, few females used agricultural crops (Figure 5). Their high reproductive rate (Figure 4), which should be reflective of high food availability, appeared to be due more to an abundance of hazelnuts (*Corylus americana, C. cornuta*) and dogwood berries (mainly *Cornus racemosa* and *C. sericea*) than acorns or crops.

Males, in contrast, were frequently found in croplands during fall, on average spending nearly 30% of their time there in September; however, individual variation in crop use among males was large (Figure 5). Although about half the landscape was comprised of agricultural crops, the crops that bears consumed (mainly corn and sunflowers) represented a small areal coverage (corn 2%, sunflowers 3% of the total cropland area).

Key bear foods separated into 5 distinct groups through stable isotope analysis: natural vegetation (herbaceous, berries, and nuts), ants, deer, corn, and sunflowers (Figure 6). We were surprised to be able to distinguish sunflowers based on their uniquely enriched N^{15}. When isotopic signatures of whole bear hairs (representing the assimilated diet over the course of the year) were plotted against these key foods, they expectedly clustered toward natural vegetation (Figure 6), and indicated that this type of food comprised 70% (67–73% 95% Credible Interval) of male and 81% (78–84%) of female annual diets. We caution that these and all other stable isotope results are preliminary, pending further analysis.

We encountered 2 problems in this analysis: (1) Most hunter-killed bears were taken early in the hunting season (~70% during the first week, September 1–7), and these animals had little time to consume and incorporate crop signatures into their hair (corn was available only about 2 weeks before the start of the hunting season). (2) The use of whole hairs, from either hunter-killed bears or collared bears, made it difficult to identify use of sunflowers, which was distinguishable only along the N axis, and could have been confounded with deer and ants, which are consumed by bears earlier in the year (Figure 6). Therefore, we conducted some experimental analyses of just the proximal end of hairs (most recent growth) obtained from a few denning bears, representing just the fall diet.

Stable isotope results obtained from the section of hair that grew during the fall showed a clearer separation in individual diets and more robust evidence of crop use by some individuals (Figure 7). Each of the 6 samples that we tested was from a GPS-collared bear for which we could match fall habitat use with fall stable isotope results. Especially distinctive were an adult female (#4021) that appeared to never used crop fields and so was assumed to feed predominantly on natural vegetation, an adult male (#4020) that spent most of the fall (every night from 13 August–24 September, and again 30 September–6 October) in a sunflower field, and another adult male (#4055) that spent most of the fall feeding in a cornfield. Diets of these bears estimated from the mean of posterior distributions from the stable isotope results were: 79% (30–100%) natural vegetation for bear 4021; 37% (10–67%) sunflowers for bear 4020; and 80% corn (73–87%) for bear 4055. Notably, bear #4055 was the heaviest bear handled during this study (554 lbs = 251 kg in late-December) and nearly half his mass was fat (49%). He lost only 7% of this mass over the winter. He denned in a small woodlot directly adjacent to the cornfield in which he fed all fall (Figure 8).

FUTURE DIRECTIONS

We plan to continue monitoring bears on these 4 study sites, although sample sizes have been greatly diminished by the loss of collared bears due to various causes. We will continue to collect GPS-collar data in the NW study site, and we are now matching the location data to heart rate data from their implanted monitors. We will compare calories of food available in different patches of habitat (derived from previous assessments of biomass of bear foods) to the energy expended by bears as they travel across the landscape. We will use this comparison to deduce the minimum amount of forest needed to support a bear (in terms of energy, not protection from humans), and thereby predict the extent to which the population can continue to expand toward the western border of the state.

ACKNOWLEDGMENTS

We thank the collaborators in this study: Brian Dirks, who conducted the fieldwork and provided all materials for the work at Camp Ripley; Dr. Paul Iaizzo at the University of Minnesota, and Dr. Tim Laske at Medtronic, Inc., who assisted with fieldwork and provided the implantable heart monitors and GPS-Iridium radiocollars. Doug Page and Tom Enright assisted greatly with fieldwork in the NW. We also thank the many landowners who allowed us to set traps on their property and checked them daily. Foremost among them were Randy Knott and Brent Hemly. Agassiz NWR kindly provided use of their bunkhouse during the winter fieldwork.

Table 1. Fates of radiocollared black bears in 4 study sites (Chippewa National Forest, Camp Ripley, Voyageurs National Park, and northwestern Minnesota), April 2011−March 2012.

	CNF	Camp Ripley	VNP	NW
Collared sample April 2011	4	7	3	16
Killed spring hunt Manitoba				1
Caught and collared summer 2011				8
Killed as nuisance				2
Killed in vehicle collision				1
Killed by Minnesota hunter				1
Dropped collar[a]				2
Failed radiocollar				2
Lost contact[b]				2
Died in den[c]				1
Collar removed in den by investigators				1
Yearlings collared in den	1		1	1
Adult female collared in den		1		
Collared sample April 2012	**5**	**8**	**4**	**12**

[a] Due to premature failure of breakaway link.
[b] Due to radiocollar failure, unreported kill, or long-distance movement.
[c] Likely shot before denning.

Table 2. Causes of mortality of radiocollared black bears ≥1 year old in 4 Minnesota study sites, 1981–2011. 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	223	11	15	12	261
Likely shot by hunter[a]	8	1	0	4	13
Shot as nuisance	22	2	1	3	28
Vehicle collision	12	8	1	2	23
Other human-caused death	9	1	0	0	10
Natural mortality	7	3	4	0	14
Died from unknown causes	4	2	0	3	9
Total deaths	**285**	**28**	**21**	**24**	**358**

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

Table 3. Black bear cubs examined in dens of radiocollared mothers in or near the Chippewa National Forest during March, 1982–2012. High hunting mortality of radiocollared bears severely reduced the sample size in recent years.

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%	50%
2011	1	4	4.0	25%	50%
2012	1	3	3.0	67%	
Overall	**177**	**466**	**2.6**	**52%**	**19%**

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

Table 4. Black bear cubs examined in dens in or near Camp Ripley Military Reserve during March, 1992–2012.

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	75%	25%
2011	3	8	2.7	50%	25%
2012	1	2	2.0	100%	
Overall	**34**	**80**	**2.4**	**53%**	**22%**

[a] Blanks indicate no cubs were born to collared females or collared mothers with cubs died before the subsequent den visit to assess cub survival.

Table 5. Black bear cubs examined in dens in Voyageurs National Park during March, 1999–2012. All adult collared females were killed by hunters in fall 2007, so no reproductive data were obtained during 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		—	—	—
2003	5	13	2.6	54%	8%
2004	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%	0%
2011	1	2	2.0	0%	0%
2012	1	2	2.0	0%	
Overall	**27**	**60**	**2.2**	**48%**	**26%**

[a] Blanks indicate no cub mortality data because no cubs were born to collared females.

Table 6. Black bear cubs examined in dens in northwestern Minnesota during March, 2007–2012.

Year	Litters checked	No. of cubs	Mean cubs/litter	% Male cubs	Mortality after 1 yr
2007	2	6	3.0	33%	100%
2008	5	15	3.0	67%	22%
2009	1	3	3.0	33%	33%
2010	6	17	2.8	41%	13%
2011	2	4	2.0	75%	25%
2012	4	10	2.5	60%	
Overall	**20**	**55**	**2.8**	**52%**	**34%**[a]

[a] Excludes the total loss of a 5-cub litter in 2007 (which was not within the designated study area).

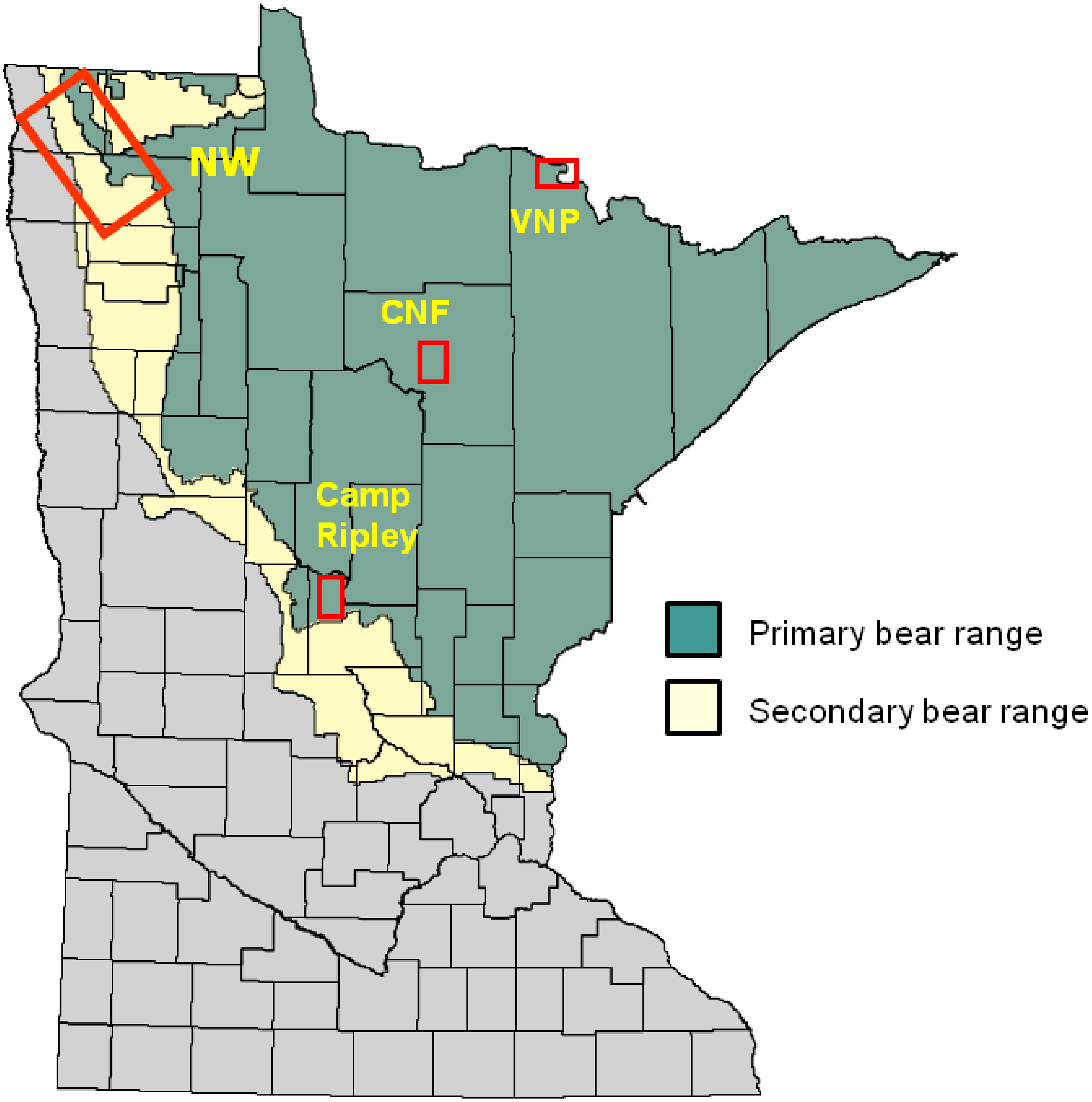

Figure 1. Location of 4 study sites within Minnesota's bear range: CNF (Chippewa National Forest, central bear range; 1981–2012); VNP (Voyageurs National Park, northern fringe of range; 1997–2012); Camp Ripley Military Reserve (near southern edge of range; 1991–2012); NW (northwestern fringe of range; 2007–2012).

Figure 2. All radio-collared bears were marked with large ear tags so they were easily visible to hunters, who were asked to avoid shooting them. Both panels show GPS-collared bears (left panel GPS-Iridium collar, right panel GPS pod mounted on a VHF collar). The bear in the right panel was photographed by a trail camera of a local resident in the NW study site who has assisted in this project. It documented that this bear, which we could no longer locate by telemetry, was still alive, but the radiocollar had failed. (Photo credit right panel: Brent Hemly).

Figure 3. Bear #56, a 38-year-old female that has been monitored since the inception of this study in 1981, emerged from her den (hole at the base of stump) in the CNF study site on March 19, 2012. This early emergence was due to unusually warm weather (73°F at time of emergence).

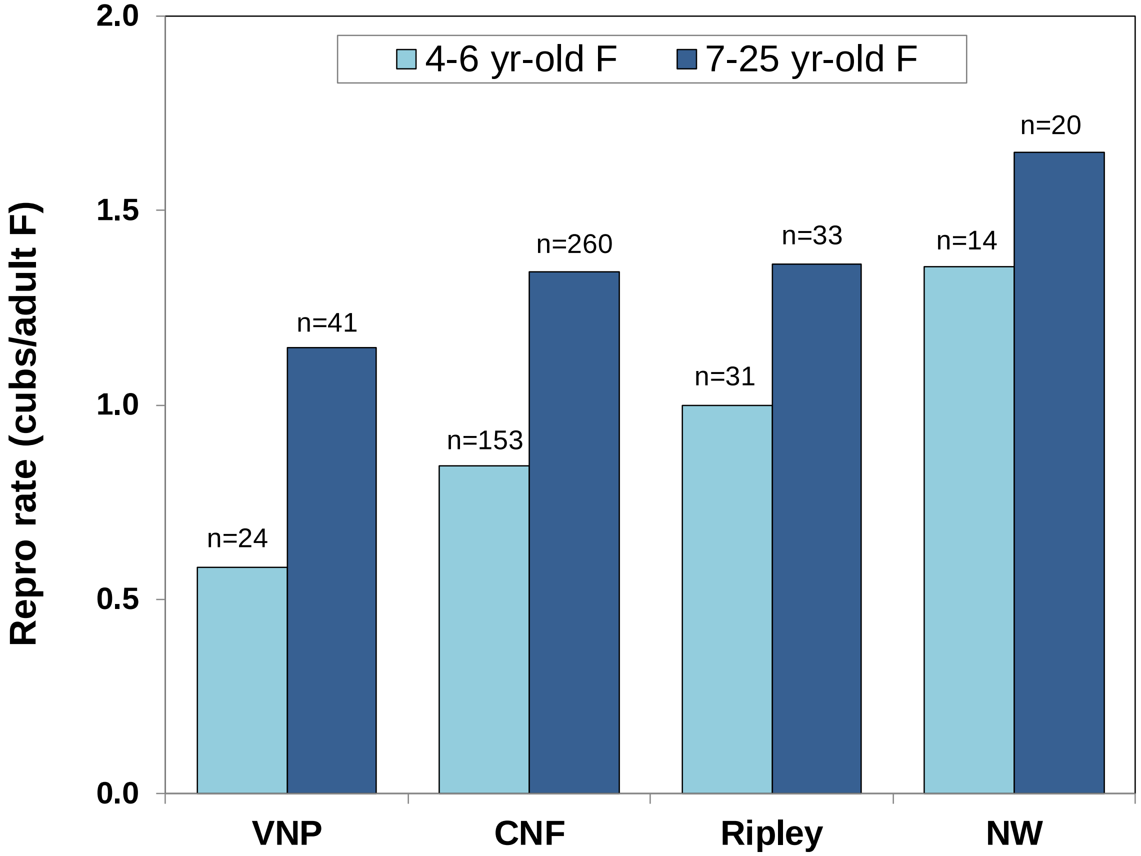

Figure 4. Reproductive rates of radiocollared bears within 4 study sites (see Figure 1) through March 2012. Sample sizes refer to the number of female bear-years of monitoring in each area for each age group. Data include only litters that survived 1 year (even if some cubs in the litter died).

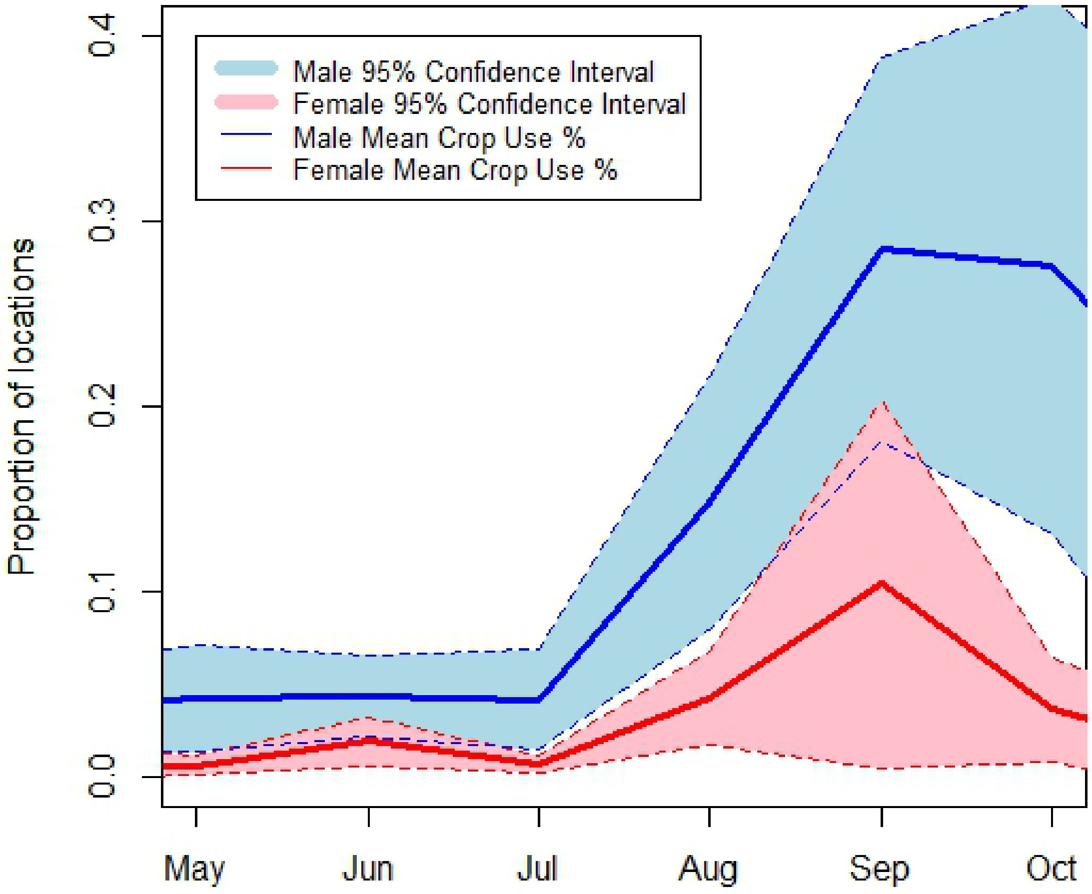

Figure 5. Proportion of GPS locations of radiocollared male and female black bears in NW Minnesota that were in crop fields each month, 2007–2011.

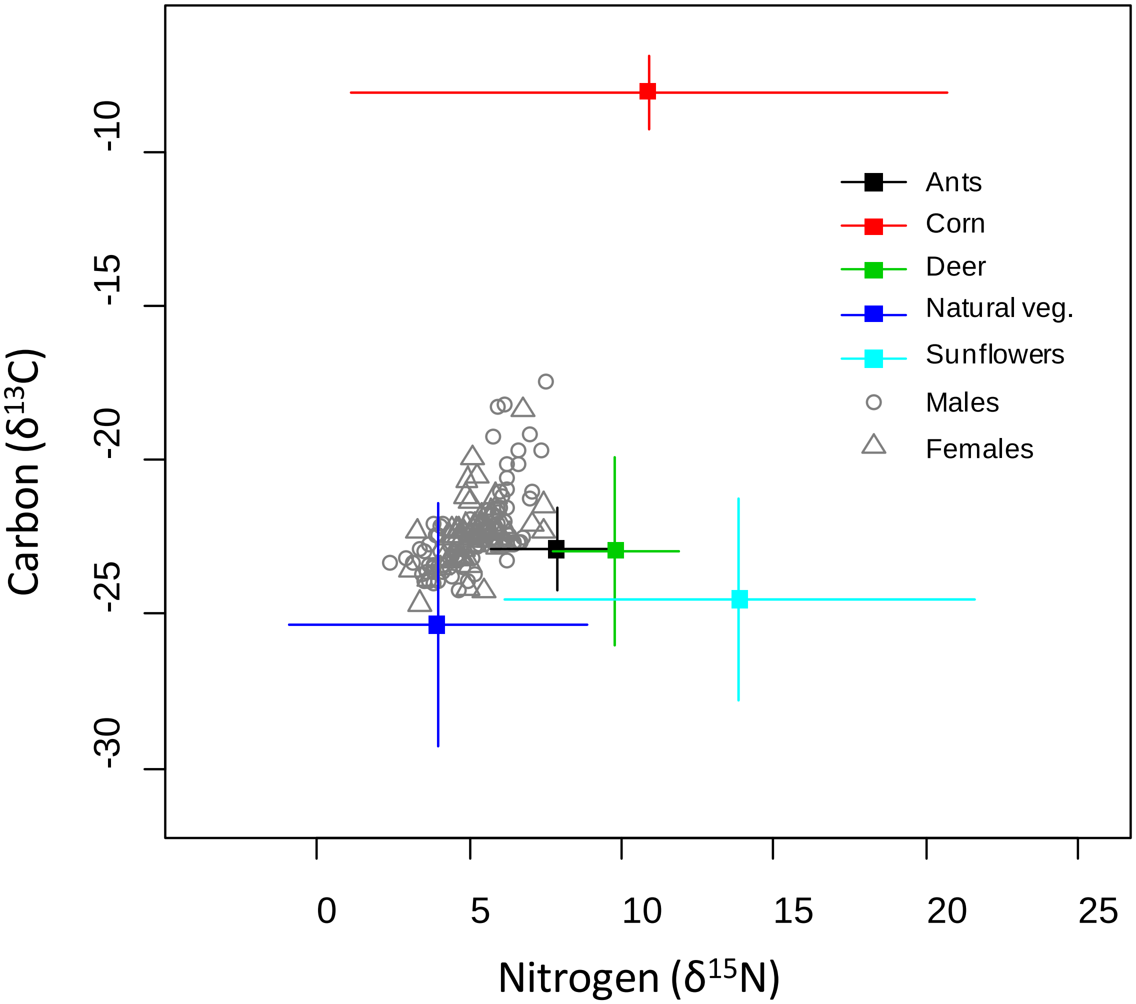

Figure 6. Stable isotope signatures obtained from hair samples of collared and hunter-killed black bears in NW Minnesota, 2007–2011 (*n* = 141) compared to mean isotope signatures (and 95%CI) of 5 types of bear foods that separated out using stable isotopes of C and N. Full hair samples of bears were used so data represent the year-round diet.

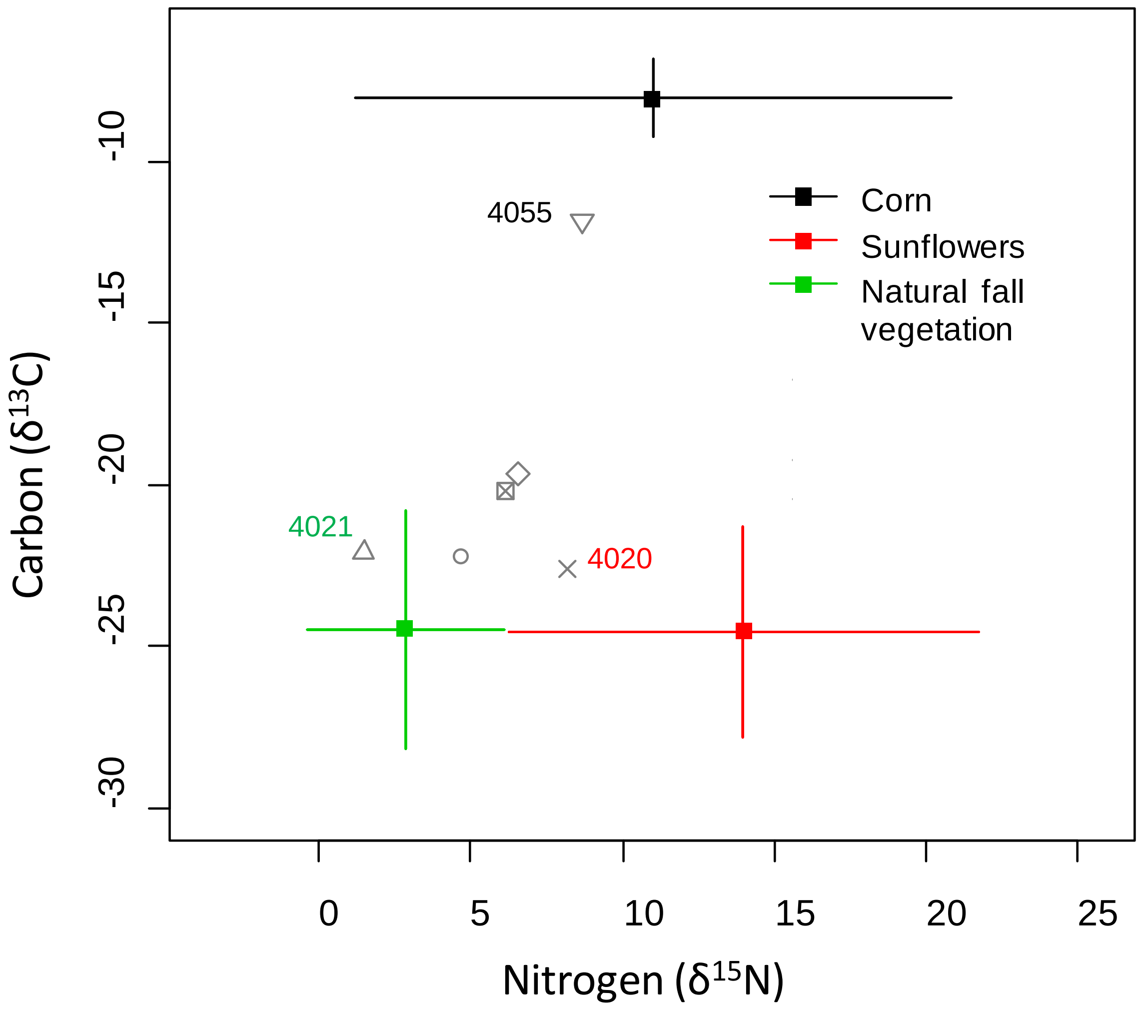

Figure 7. Stable isotope signatures of 6 GPS-collared bears in NW Minnesota during fall (based on only the proximal third of hair samples plucked in winter) matched against mean isotope signatures (and 95%CI) of 3 types of fall foods. Identification numbers of 3 individuals are highlighted who were located most or all of the time in either natural vegetation (4021), sunflowers (4020) or corn (4055) during the fall.

Figure 8. Adult male bear 4055, the largest bear handled during this 31-year study, fed in a cornfield all fall, 2011 (see Figure 7), and denned in this small woodlot adjacent to the field. The building in the photo was not inhabited or used.

A LONG-TERM ASSESSMENT OF THE VARIABILITY IN WINTER USE OF DENSE CONIFER COVER BY FEMALE WHITE-TAILED DEER

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SUMMARY OF FINDINGS

Our long-term study in north-central Minnesota was prompted by a management concern for increasing white-tailed deer (*Odocoileus virginianus*) densities relative to available dense conifer cover, and its goal was to improve our understanding of the biological value of dense conifer cover to deer on winter range. Winter severity varied widely, as did its effect on autumn migration of our radiocollared female deer to winter range. We monitored deer use of dense conifer cover by a mixture of very high frequency (VHF) telemetry and Global Positioning System (GPS) collar technology, each with advantages and disadvantages. The VHF and GPS data sets of our study cohort complement each other quite well and probably provide more of an enhanced understanding of winter use of habitat by deer than either data set would individually. The less sophisticated and less expensive VHF collars allowed us to collar and monitor the winter daytime locations of many deer long-term, facilitating a more in-depth examination of population-level habitat use patterns as a function of winter weather conditions. Conversely, the fine scale temporal locations of the GPS collars permitted us to more continuously (daytime and nighttime) monitor the winter locations (use of habitat) of a subsample of the study cohort during a briefer part of the study period and to explore more in-depth the variability of within and among individual response patterns. This summary presents the results and discussion of a significantly updated data analysis focused on the objectives described below.

INTRODUCTION

For at least 60 years, wildlife researchers and managers have been describing at varying levels of detail the prevalence of dense conifer stands and their use by northern white-tailed deer on winter ranges (Hamerstrom and Blake 1939, Verme 1965, Morrison et al. 2003, Hurst and Porter 2008) and documenting the negative impacts that winter weather conditions (e.g., ambient temperature, snow accumulation) have on deer survival and reproduction (Severinghaus 1947, Mech et al. 1971, Nelson and Mech 1986*a*, DelGiudice et al. 2002, Carstensen et al. 2009). These impacts have been related to nutritional restriction and poor condition, predation, or a combination (Severinghaus 1981, Nelson and Mech 1986*b*, DelGiudice 1998, Ballard et al. 1999, DelGiuidice et al. 2002).

Given the potential effects of winter severity on population performance of northern deer species, numerous studies have focused on the weather-moderating attributes of dense conifer stands, specifically assessing their potential value as thermal cover and snow shelter. Ozoga (1968) reported that within dense, even-aged conifer stands thermal ranges of ambient temperature were narrowest, average temperatures warmest, wind flow lowest, and relative humidity highest and most stable compared to other cover types, all suggestive of potential energetic benefits to deer during the coldest weather. However, there is little evidence from these studies or others conducted under controlled conditions that the potential energetic benefits of thermal cover actually translate to improved winter condition, reproduction, or survival of deer or other cervids (Freddy 1986, Cook et al. 1998, Beyer et al. 2010).

The potential value of dense conifer stands as snow shelter for deer in the northern Great Lakes region becomes particularly evident when snow cover accumulates to depths that physically impede their mobility, markedly increase energetic costs of movement, and decrease browse availability (Wetzel et al. 1975, Moen 1976, Morrison et al. 2003). Snow depths of ≥25-40 cm seriously restrict movements of white-tailed deer (Kelsall and Prescott 1971, Moen 1976, Tierson et al. 1985), but depths within conifer stands may be reduced by 25 to 36 percent due to interception of snowfall by canopies ≥70 percent (U. S. Army 1956, Ozoga 1968). Snow depth has been directly related to wolf predation (Nelson and Mech 1986*b*, DelGiudice 1998) and reduced overall winter survival, whereas no such relationships between ambient temperatures and survival were detected (Nelson and Mech 1986*a*, DelGiudice et al. 2002, 2006).

It is unclear whether minimum ambient temperature (i.e., air chill) or deepening snow cover has the most pronounced effect on deer use of dense conifer cover. A number of studies have indicated that low temperatures and cold winds (or air chill) may have the greatest impact on prompting deer to seek yarding areas with shelter, whereas movements within those areas and use of dense cover specifically may be most strongly influenced by increasing snow depths (Ozoga 1968, Morrison et al. 2003). Others have questioned the "need" for thermal cover when available nutrition is adequate to fulfill energetic requirements (Moen 1976, Cook et al. 1998), but even when it is not, the work of Cook et al. (1998) suggests that the weather-moderating influences of conifer cover may be too small, infrequent, and variable to convey biologically significant benefits. Finally, the thermal benefits afforded to free-ranging cervids from increased exposure to solar radiation in open areas are likely of greater relative value to their energetic balance and fitness than the potential thermal benefits associated with dense cover, particularly when ambient temperatures are coldest (Verme 1965, Moen 1973, Cook et al. 1998).

Given the wide variation of periodicity, intensity, and duration of climatic factors, such as ambient temperature and snowfall, winter severity and its effect on deer behavior can be highly variable from year to year (Verme and Ozoga 1971). Long-term studies provide the opportunity to capture a wide breath of environmental variability and a broader context within which to examine and maximize our understanding of relationships to specific aspects of wildlife behavior (DelGiudice and Riggs 1996).

Our long-term study in north-central Minnesota was prompted by a management concern for increasing deer densities relative to available dense conifer cover, and its goal was to improve our understanding of the biological value of dense conifer cover to deer on winter range. During a 16-year period winter severity varied widely, as did its effect on autumn migration of radiocollared female deer to winter range (DelGiudice et al. 2005, Fieberg et al. 2008). We monitored deer use of dense conifer cover by a mixture of very high frequency (VHF) telemetry and Global Positioning System (GPS) collar technology, each with advantages and disadvantages. Use of VHF telemetry from fixed-wing aircraft involved more individual deer, covered more years and more variable winter weather conditions, but "good" weather conditions (for flying) were required to obtain locations, greater spatial error was associated with them, and they were collected less frequently than GPS locations (1/hr or 1/4 hr).

OBJECTIVES

1. Examine and compare the observed variation of deer use of dense conifer cover, as monitored by VHF telemetry and GPS collar technology;
2. Assess the apparent influence of winter severity (ambient temperature, snow depth) and cover availability on deer use of dense cover.
3. Discuss the implications of the 2 data collection methods relative to our ability to learn about how environmental variability impacts habitat use, other behavioral responses, and ultimately, fitness.

METHODS

Study Area

Our study included 4 winter range sites located along the southeastern boundary of the Chippewa National Forest in north-central Minnesota, USA ($46^{o}52'$-$47^{o}15'$N and $93^{o}45'$-$94^{o}07'$W). The Willow (Wil), Inguadona (Ing), Shingle Mill (Shi), and Dirty Nose (Dir) sites were 20, 24, 23, and 13 km^2, respectively. The uplands were dominated by deciduous and mixed deciduous-conifer stands, whereas northern white cedar (*Thuja occidentalis*), black spruce (*Picea mariana*), balsam fir, and tamarack (*Larix laricina*) were most apparent on the lowlands (Doenier et al. 1997). The winter diet of deer on the 4 sites was highly diverse (about 36 browse species), but beaked hazel (*Corylus cornuta*), mountain maple (*Acer spicatum*), and red-osier dogwood (*Cornus sericea*) accounted for 82 and 89% of species browsed during mild and severe winters, respectively (DelGiudice et al. 2010).

We calculated a Minnesota Department of Natural Resources (MNDNR) winter severity index (WSI) by accumulating 1 point for each day with a snow depth ≥38 cm and 1 point for each day with an ambient temperature ≤ -17.7^{o}C during November-May. During winters 1990-1991 to 2004-2005, maximum WSIs ranged from 45 to 205, and snow depths in the open ranged from 0 to 98 cm. Monthly mean daily minimum and maximum temperatures ranged from -28^{o} to 13^{o}C and -15^{o} to 30^{o}C, respectively (National Oceanic and Atmospheric Administration 1990-2005).

Sixty-eight percent of 335 radiocollared female deer were classified as seasonal migrators, inhabiting spatially non-overlapping winter and spring-summer-fall home ranges (Fieberg et al. 2008). Annual mean migration distances ranged from 9.4 to 14.7 km (range for individuals = 1.5-34.8 km).

Wolf (*Canis lupus*) predation is the primary source of natural mortality of adult deer in north-central and northeastern Minnesota (Nelson and Mech 1986*a*,*b*; DelGiudice et al. 2002, 2006). Wolf numbers in northern Minnesota have been stable since the mid- to late 1990s (Erb 2008); the most recent (2008) estimate was 2,921 wolves. The most recent point estimate of the bear population was 17,500 (Garshelis and Noyce 2011).

Deer Capture, Handling, and Monitoring

We captured deer primarily by Clover traps (95%) during January-March 1991-2005. We reported complete details of handling elsewhere (DelGiudice et al. 2001, 2005), but relative to the objectives of this summary, each deer was fitted with either a VHF (Telonics, Inc., Mesa, Arizona) radiocollar or a GPS (G-2000, Advanced Telemetry Systems, Inc., Isanti, Minnesosta) collar during handling. Animal capture and handling protocols were approved by the University of Minnesota's Institutional Animal Care and Use Committee and met the guidelines recommended by the American Society Mammologists (Animal Care and Use Committee 1998).

During winter (1 November-14 May), we attempted to locate VHF-collared deer from fixed-wing aircraft (all during daylight hours) as many times as possible each week, given the inherent constraints (e.g., weather conditions) associated with this technique (Fuller et al. 2005, Kochanny 2009, Kochanny et al. 2009). We followed a total of 267 deer using VHF technology during winters 1993-1994 to 2004-2005. Most deer were followed for 1-2 years (mean = 1.8, interquartile range = 1-2). The number of locations per deer was highly variable (mean = 20.2, interquartile range = 3-25, min = 1, max = 144).

We deployed GPS collars on a total of 24 deer (1 during 2 winters) at least 1.5 years old. We pre-programmed collars to attempt a location either every hour or every 4 hours, depending on the date, life history events (e.g., fawning), and battery-life considerations. We followed 2 individuals in winter 2000-2001, 4 in 2001-2002, 10 in 2003-2004, and 9 in 2004-2005.

Habitat Composition of Sites

We used mirror stereoscopes (Leitz, Forestry Suppliers, Inc., Jackson, Missouri) and 9" x 9" and 4" x 6" *leaf-off*, color infrared air photos (1:15,840") to delineate and map forest stands according to a classification system used to assign dominant tree species, classes of height (<20', 20' ≤ *x* < 35', and ≥35') and conifer canopy closure (A, < 40%; B, 40% ≤ *x* < 70%; and C, ≥70%). For analyses in this report we grouped all classes of forest stands into 1 of 4 categories—conifers with canopy closure < 40%, 40-70%, or ≥70%, and "other." We collected 4-8 ground control points (GCP) for each photo using a Trimble Geo-Explorer GPS (Trimble Navigation Limited, Sunnyvale, California). We collected GCPs by averaging 300 points recorded at each location. These GCPs were then post-processed to improve accuracy by Trimble's Pathfinder software using a base station file from a base station located in Duluth. We digitized our original vector line coverage in EPPL7, a GIS developed by the Land Management Information Center, Department of Administration, State of Minnesota.

We performed all digitizing using the Universal Transverse Mercator (UTM) Zone 15 North Coordinate System. Habitat polygons were delineated down to a patch size of 0.5 ha. Once the digital line file was created, it was uploaded into ArcInfo 6.0 (Environmental Systems Research Institute [ESRI], Redlands, California), and the polygons were built and cleaned. Once the polygons were created, the attribute table was populated with the forest habitat data from the air photointerpretation.

Air photo flights were flown annually to capture any timber harvests that occurred on each study site. Once a cut had occurred cut alterations were digitized for each site, and the coverage was updated (Figure 1). This resulted in a new coverage for each study site for most years of the study. Beginning in 1999, all cuts were digitized using air photos scanned in high definition and “heads up digitizing” in Arcview 3.3 (ESRI). These photos were rectified on the screen using rectification points plucked from the 1991 U. S. Geological Survey (USGS), National Aerial Photography Program (NAPP) photos. In 2005, we updated all coverage’s attribute tables to account for forest habitat classification changes that occurred due to forest succession. We used ArcGIS (ArcMap Version 9.3.1) to depict and measure areas of the 4 study sites (km^2) and forest cover types (ha) within each site, and to overlay winter VHF- and GPS-derived locations of radiocollared deer.

Data Analyses

Similar to Kneib et al. (2011), we fit multinomial response models to VHF data with the following structure:

P(observation $\notin$ (conifer Class B or C) | $S_{i,j}$, $T_{i,j}$) = $A_{O,}$ / D

P(observation $\in$conifer Class B | $S_{i,j}$, $T_{i,j}$) = $A_B \exp(\beta_{B,0} + \beta_{B,S}S_{i,j} + \beta_{B,T}T_{i,j}) / D$

P(observation $\in$conifer Class C | $S_{i,j}$, $T_{i,j}$) = $A_C \exp(\beta_{C,0} + \beta_{C,S}S_{i,j} + \beta_{C,T}T_{i,j}) / D,$

with $S_{i,j}$ and $T_{i,j}$ giving the snow depth (cm) and minimum daily temperature (^oC) measured on day i of year j, respectively, and $D = A_O + A_B \exp(\beta_{B,0}+\beta_{B,S}S_{i,j}+\beta_{B,T}T_{i,j}) + A_C$ exp $\exp(\beta_{C,0}+\beta_{C,S}S_{i,j}+\beta_{C,T}T_{i,j})$, a normalizing constant that ensures the probabilities sum to 1. The availabilities, A_O (includes “other” and conifer class A), A_B, and A_C, were adjusted yearly to account for timber harvest and succession (Fig. 1). The β’s quantify the increase in use of conifer cover classes B and C (relative to an “other” category) as a function of snow depth and minimum daily temperature. If all β’s are 0, then we recover a null model that assumes use of each habitat type is proportional to its availability.

Rather than use random effects to account for repeated measures and within-animal correlation (as in Kneib et al. 2011), we used a generalized estimating equation approach to inference (Zeger et al. 1988). Specifically, we estimated regression parameters using a working independence assumption with custom-written code and built-in optimizers (“optim”) within program R (R Core Development Team 2009). We accounted for the repeated measures design by using a non-parametric bootstrap, re-sampling individuals with replacement. Thus, we treated the observations as though they arose from a 2-stage cluster design, with the first stage representing individual animals on the study site (sampled independently) and the second stage representing locations of these animals (Clark and Strevens 2008, Fieberg et al. 2010). This approach has the advantage of simplicity, but more importantly, the regression parameters reflect population-level response patterns that are of primary interest to managers (Fieberg et al. 2009).

We constructed date-time plots of GPS data to explore diurnal and seasonal patterns of habitat use, as well as among-individual variability in these patterns. Specifically, for each deer, we constructed a level or image plot with the *x*-axis depicting Julian date (23 January-14 April) and *y*-axis depicting hour of day (0-23), with color used to indicate the cover type associated with each observed location. In addition, we overlaid time series of estimated snow depths to explore habitat use patterns relative to changes in snow depth.

RESULTS

VHF Data

At all 4 sites deer made greater use of more open habitat types (i.e., "other") compared to moderately dense (Class B) and dense (Class C) conifer cover, particularly when snow depths were shallow to moderate (< 40 cm, Figure 2). However, population-level responses to increases in snow and ambient temperature were most pronounced at Wil and Shi, the 2 sites with the largest amount of Class C (Fig. 1). Estimates of $\beta_{C,S}$ were positive for the Wil, Shi, Dir, and Ing sites and significantly different from 0 ($\alpha = 0.05$) at all but Dir (Table 1), suggesting deer increased their use of conifer Class C as snow depth increased (Figure 2A-D). When there was no snow cover, the probability of use of Class C was ≤ 0.22 for all 4 sites (Figure 2A-D); however, at maximum snow depths (90-100 cm) the probability of use of this type was 3X and at least 2X greater at Wil and Shi, respectively, than at Dir and Ing. The estimate of $\beta_{B,S}$ was also positive and significantly different from 0 for Shi (Table 1), suggesting increased use of Class B at this site as snow depth increased. Simultaneously, deer use of "other" decreased dramatically with increasing snow depths at Wil and Shi.

Estimates of $\beta_{C,T}$ also were positive for all 4 study sites and significantly different from 0 for Wil and Shi (Table 1), suggesting increasing use of conifer Class C as daily minimum temperatures increased (Figure 2E-H). The estimate of $\beta_{B,T}$ was also positive and significantly different from 0 for Shi (Table 1, Figure 2G).

GPS Data

Maximum snow depths were moderate during 3 of the 4 winters when GPS collars were deployed on deer; however, during winter 2000-2001, it peaked at 80 cm. There was significant among-animal variability in their propensity to use conifer classes A, B, C, or "other" (Figure 3). Some individuals were almost always located in a single habitat type. For example, in 2002, Deer 709, 773, and 592 were most typically in open habitats (i.e., "other"). Similarly, Deer 513 (in 2005) was almost always in Class B, and Deer 541 in both 2004 and 2005, was almost always in conifer Class C, despite the very different snow depths in these winters. During moderately severe winter 2000-2001, the 2 GPS-collared deer at Wil both made intense use of Class C for 2-4 weeks. Some animals used a variety of habitat types, but exhibited significant inertia relative to specific types, such that individuals tended to use the same cover type for long periods of time (e.g., see Deer 551 and 464 in 2001). Any diurnal pattern was relatively weak; animals largely seemed to make similar use of habitat types during the day and night (Figure 3).

DISCUSSION

The VHF and GPS data sets of our study cohort complement each other quite well and probably provide more of an enhanced understanding of winter use of habitat by deer than either data set would individually. The less sophisticated and less expensive technology of the VHF collars allowed us to collar and monitor the winter daytime locations of many deer long-term (12 years), facilitating a more in-depth examination of population-level habitat use patterns as a function of environmental conditions (i.e., winter weather). Conversely, the more recently developed and expensive GPS collars permitted us to more continuously (daytime and nighttime) monitor the winter locations (use of habitat) of a subsample of the study cohort during a briefer part of the study period. Consequently, it was difficult to assess temporally changing use patterns in response to weather conditions, but the finer scale temporal locations allows for a more in-depth exploration of the variability of within- and among-individual response patterns. Specifically, it became clear from GPS data that individuals respond differently during the same type of winter conditions, whether it be relative to use of specific habitat types (e.g., dense conifer cover) as noted here or relative to seasonal migration or winter food habits (Fieberg et al. 2008, also see our current research summary on food habits).

As indicated by our data at the population level (by VHF-collared deer) and the individual level (GPS-collared deer), habitat availability must be considered when interpreting patterns of habitat use. Overall, deer made greater use of dense conifer cover and increased their use of this habitat type more abruptly relative to increasing snow depths on the Wil and Shi sites where it was most available. Studies have shown that northern deer concentrate their winter movements on landscapes that possess at least 50% suitable (i.e., dense conifer) cover (Weber et al. 1984, Doepker and Ozoga 1991, Potvin and Boots 2004). The influence of availability on deer use of other resources, such as food, has been discussed (Mautz 1978, DelGiudice et al. 1989), and this interacts with deer use of winter complexes (Ozoga 1968, Morrison et al. 2003, Felix et al. 2007, Hurst and Porter 2008).

Whereas at the individual level our GPS data demonstrate there is substantial variability among deer in their use of dense conifer cover relative to snow depth, at the population level, the similarity in the deer's increased use of this type relative to increasing snow depths and increasing daytime ambient temperatures was striking, again, on the sites where dense conifer was most available (Wil and Shi). The most reasonable interpretation of this pattern may have more to do with the deer's decreased use of dense cover as daytime ambient temperatures dropped below freezing to benefit from increased exposure to solar radiation (Verme 1965, Moen 1973, Cook et al. 1998).

There are several important implications of our findings relative to our ability to learn about the specific value and importance of conifer cover to deer relative to winter severity. Studies must be long enough to observe deer behavioral responses to winter weather conditions ranging from mild to severe. Given the pronounced among-animal variability we observed in habitat use by our GPS-collared deer, the study cohort must be large enough to confidently assess a population-level response. And because our data suggest that deer use of conifer cover may depend heavily on its availability and arrangement with other habitat types, to adequately assess its use and the value of this habitat type will require large study sites and the ability to assess the health and fitness of these deer over time.

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Table 1. Regression parameter estimates (95% percentile-based bootstrap confidence intervals[1]) from habitat use models fit to very high frequency (VHF) location data collected from 267 adult (≥1.5 years old), female white-tailed deer, north-central Minnesota, 1 November-14 May 1993-1994 to 2004-2005.

	Study sites			
Parameter[2]	Willow	Dirty Nose	Shingle Mill	Inguadona
$\beta_{B,0}$	1.289 (0.618, 1.828)	-0.2 (-0.876, 0.401)	-0.376 (-0.985, 0.236)	1.009 (0.508, 1.309)
$\beta_{C,0}$	0.52 (0.123, 0.902)	0.997 (0.212, 1.683)	-1.17 (-1.849, -0.518)	0.365 (-0.205, 0.821)
$\beta_{B,S}$ (snow)	0.002 (-0.006, 0.013)	0.011 (-0.001, 0.02)	**0.024 (0.013, 0.033)**	-0.004 (-0.014, 0.008)
$\beta_{B,T}$ (temp)	0.007 (-0.012, 0.028)	0.015 (-0.011, 0.052)	**0.032 (0.002, 0.068)**	-0.011 (-0.03, 0.012)
$\beta_{C,S}$ (snow)	**0.018 (0.011, 0.025)**	0.002 (-0.01, 0.018)	**0.04 (0.026, 0.054)**	**0.009 (0.001, 0.017)**
$\beta_{C,T}$ (temp)	**0.018 (0.005, 0.032)**	0.013 (-0.011, 0.048)	**0.042 (0.023, 0.063)**	0.002 (-0.015, 0.015)

[1] Confidence intervals that do not include 0 are in bold, indicating a statistically significant result (at $\alpha = 0.05$).

[2] Canopy closure classes ≥70% and 40%≤ x< 70% are denoted by C and B, respectively. Snow depth (cm) and minimum daily temperature (^{o}C) are denoted by S and T, respectively.

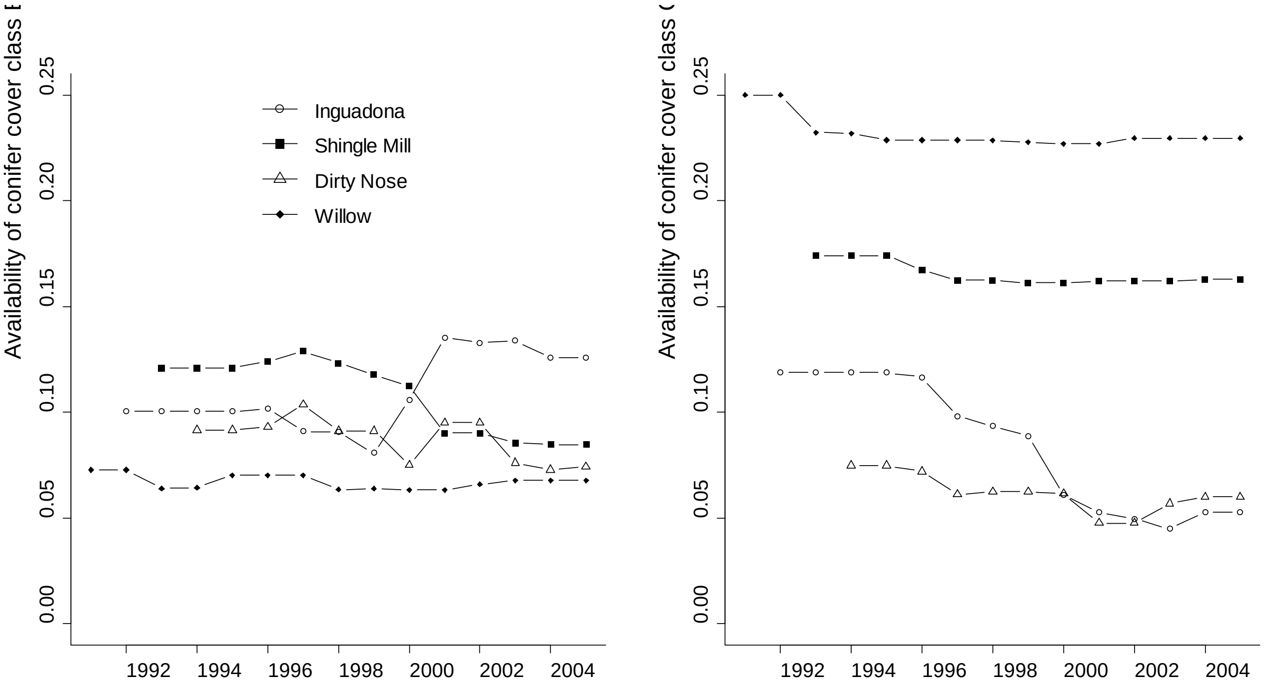

Figure 1. Proportional availability (versus time) of conifer cover classes B (40% ≤ *x* < 70% canopy closure, left panel) and C (≥70% canopy closure, right panel) for each of 4 study sites, north-central Minnesota, 1991-2005. First-year baseline was dependent on the year the site was incorporated into the study and its habitat quantified.

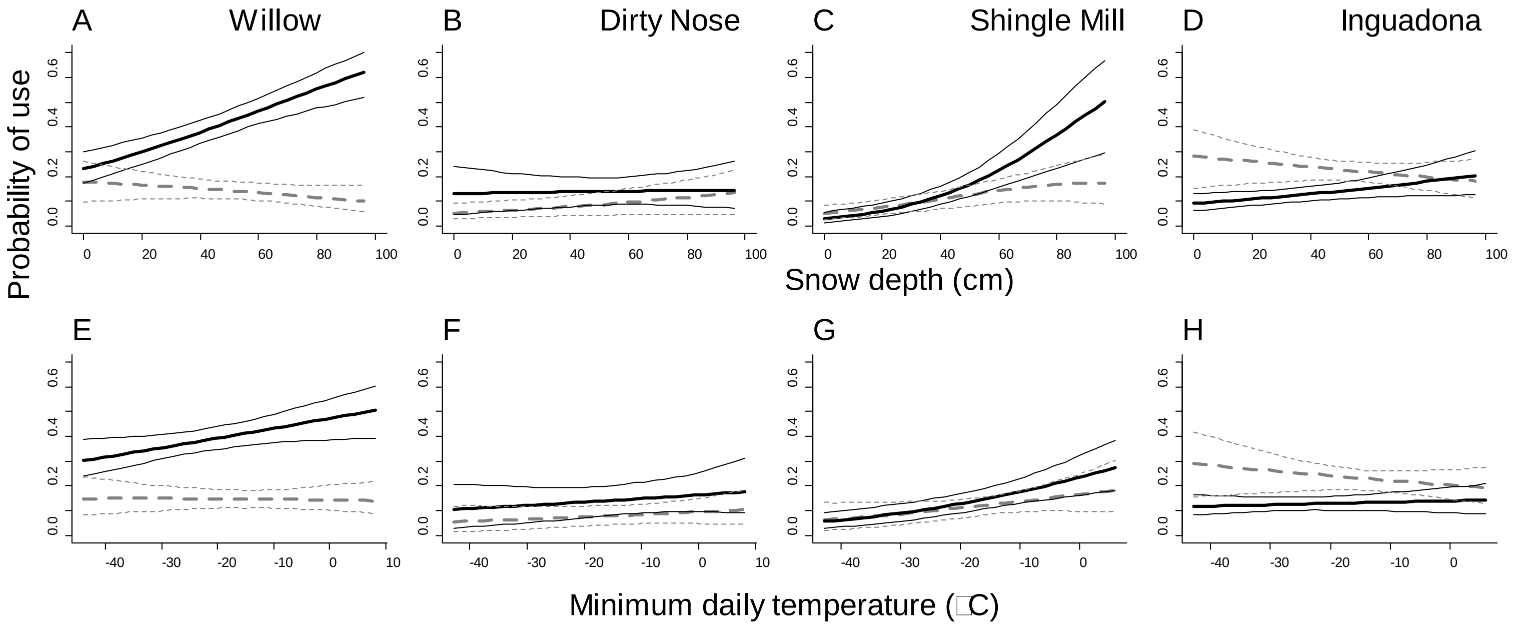

Figure 2. Model-based predicted probabilities of adult (≥1.5 years old), female white-tailed deer using conifer class C (≥70% canopy closure, dotted gray line) and conifer class B (40% ≤ *x* < 70% canopy closure, solid black line) during daytime hours (i.e., 0730-1700 hr) as a function of snow depth (panels A-D) and minimum daily temperature (panels E-H), for each of 4 study sites, north-central Minnesota, 1 November-14 May 1993-1994 to 2004-2005. Lighter lines give point-wise 95% bootstrap confidence intervals.

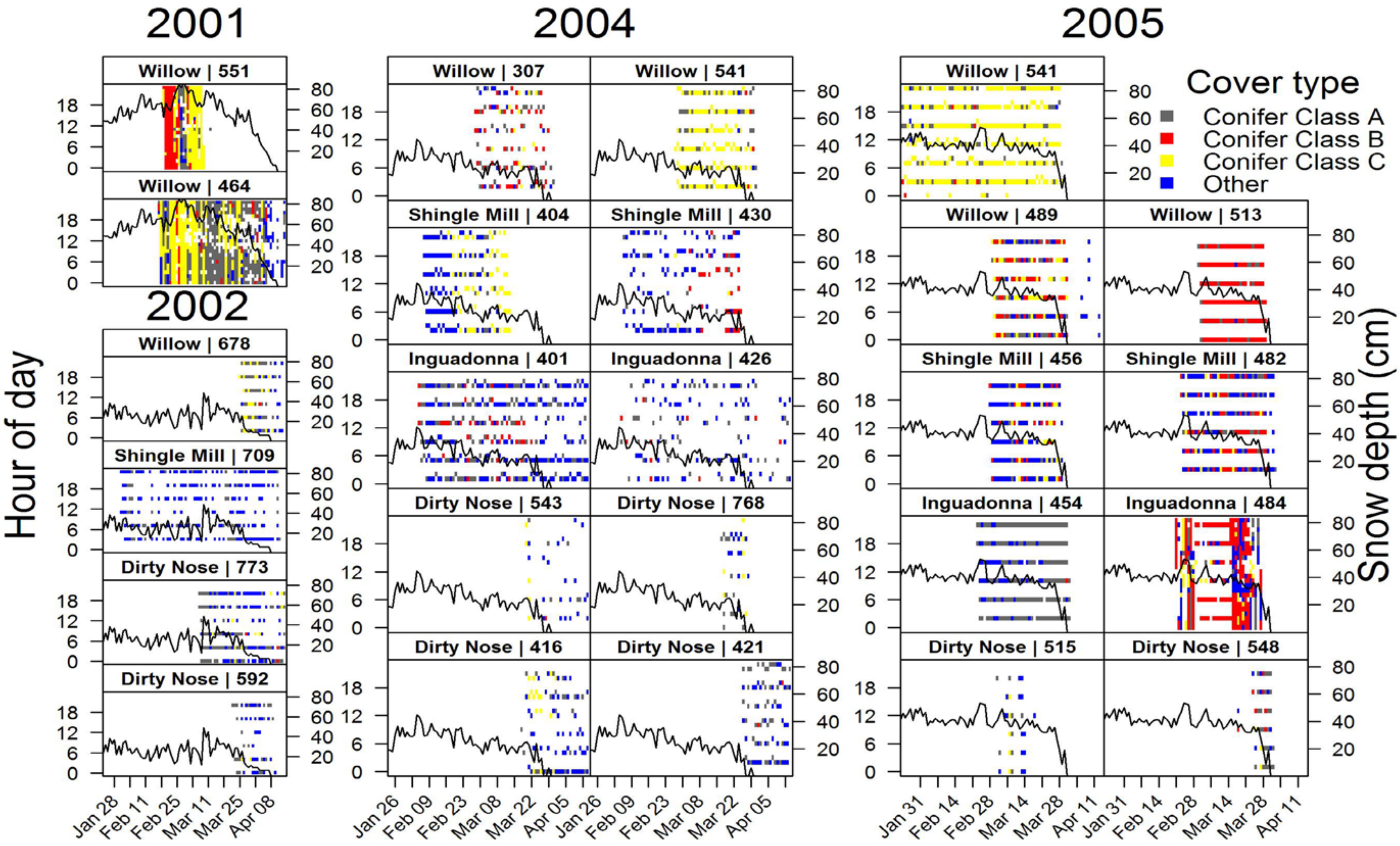

Figure 3. Date-time plots illustrating use of conifer classes A (< 40% canopy closure), B (40% ≤ *x* < 70% canopy closure), C (≥70% canopy closure) and “other” (open and hardwood types) by adult (≥1.5 years old), female white-tailed deer monitored using GPS collars collecting locations hourly or every 4 hours on 4 study sites, north-central Minnesota, 23 January-14 April 2001, 2002, 2004, and 2005.

A LONG-TERM ASSESSMENT OF THE EFFECT OF WINTER SEVERITY ON THE FOOD HABITS OF WHITE-TAILED DEER

Glenn D. DelGiudice, Barry A. Sampson, and J. Giudice

SUMMARY OF FINDINGS

Nutrition and food source are key considerations of management strategies for winter habitat enhancement for white-tailed deer (*Odocoileus virginianus*). Yet little has been reported on the effects of variation in winter severity, snow depths particularly, on browse availability and diet composition of free-ranging deer. Based on a 13-winter data set, we examine whether browse availability and diet composition (browse) change as a function of winter severity (mild to historically severe) and spatial location (4 sites). The winter browse diet of deer was highly diverse during mild and historically severe winters; diversity and number of browse species available along feeding trails and used by deer did not change with increases in daily snow cover to 98 cm. Overall, their winter diet included 35 species of browse. However, 90% of browse species had mean relative availability values of < 8% (of total available), and mean relative use of most (75%) was < 5.8% (of total browsed). On average, beaked hazel and mountain maple were the 2 most commonly used species. Browsing intensity (proportion of total available stems used) was the strongest signal as snow depths exceeded 40 cm, which in terms of energy costs for mobility is a critical threshold for deer. This work is the end-result of an ongoing data analysis effort to implement statistical methods which address specific predictions associated with our study objectives.

INTRODUCTION

Seasonality of the northern white-tailed deer's diet is directly influenced by seasonal variation in abundance, availability, phenology, and nutritional quality of the plants on which they rely. Deer have adapted to the nutritional restrictions of winter, in part, by transitioning to a diverse browse-dominated diet early in the season (Aldous and Smith 1938, Wetzel et al. 1975, DelGiudice et al. 1989). Generally, the nutrient quality and digestibility of browse are relatively low (Verme and Ullrey 1972), but as snow depths increase, browse serves as a more abundant, accessible, and energetically affordable food source compared to ground forage, acorns, and leaf litter (Rogers et al. 1981, Ditchkoff and Servello 1998, Tremblay et al. 2005). Still, during winters of uncommonly, deep or prolonged, restrictive snow conditions, progressive undernutrition either directly or as a contributing factor leads to increased mortality and to subsequent reproductive consequences (Mech and Karns 1977, Verme and Ullrey 1984, DelGiudice 1998).

Diet diversity is integral to the deer's ability to limit the rate of condition deterioration during winter (Verme and Ullrey 1972). Winter diets of northern deer typically include at least 24 species of browse; however, as few as 6 species may account for up to 75 percent of their diet, the remainder being consumed in relatively modest or even trace amounts (Aldous and Smith 1938, Rogers et al. 1981, DelGiudice et al. 1989). The apparent importance of individual species tends to vary regionally, but the value of the full diversity of their seasonal diets may be to dilute ingestion of secondary compounds, which at high concentrations affect the palatability of plants and inhibit rumen microbial function and digestion.

Annually, seasonal migration of deer to winter range, use of dense conifer cover, and their mortality rates may be highly variable in the Great Lakes states, but they are directly influenced by winter severity (Nelson and Mech 1986; DelGiudice et al. 2002, 2006; Beyer et al. 2010; Fieberg et al. 2008). It has been postulated that fall migration is an anti-predatory strategy for deer and a survival adaptation allowing deer greater access to more available food sources on their winter ranges (Severinghaus and Cheatum 1956, Nelson 1998). Still, long-term studies in northern Minnesota have revealed that during moderately severe to severe winters, most deer migrate to winter range and winter mortality, particularly from wolf predation can be unusually high (DelGiudice et al. 2002, 2006; Fieberg et al. 2008). During the historically severe winter of 1995-1996, surplus-killing of deer was apparent in northern Minnesota, but much of this predation was compensatory, as many of the dead dear were severely undernourished and moribund (DelGiudice 1998).

Nutrition and food source are key considerations of management strategies for winter habitat enhancement for deer (Minnesota Department of Natural Resources [MNDNR] 1985). Yet little has been reported on the effects of variation in winter severity, snow depths particularly, on the diet composition of free-ranging deer. From our long-term study (13 of 15 years), we qualitatively describe winter browse use by deer, including variability among sites and years.

OBJECTIVES

1. We examine whether diet composition (browse) changes as a function of winter severity and spatial location (sites) by testing the following 2 predictions:
   a. Mean diversity and number of browse species (i.e., richness) used will increase with snow depth beyond 18 and 30 cm.
   b. Mean proportion of available stems (all species) browsed will be positively correlated with snow depth or cumulative days of deep snow ("cumdeep").
2. We also examine whether browse availability (along random feeding trails) changes as a function of winter severity and among sites by testing the following predictions:
   a. Mean number and diversity of browse species along feeding trails will decrease as a function of increasing snow depth or cumulative days of deep snow.
   b. Variation in the number and diversity of available browse species among feeding trails will decrease as a function of increasing snow depth or cumulative days of deep snow.
   c. Mean abundance of available browse (stems) along feeding trails will decrease as a function of increasing snow depth or cumulative days of deep snow.

STUDY AREA

Our study included 4 winter range sites located along the southeastern boundary of the Chippewa National Forest in north-central Minnesota, USA (46°52'-47°15'N and 93°45'-94°07'W). The Willow (Wil), Inguadona (Ing), Shingle Mill (Shi), and Dirty Nose (Dir) sites were 20, 24, 23, and 13 km^2, respectively. Deciduous and mixed deciduous-conifer stands, including trembling aspen (*Populus tremuloides*), balsam poplar (*Populus balsamifera*), paper birch (*Betula papyrifera*), black ash (*Fraxinus nigra*), balsam fir (*Abies balsamea*), red pine (*Pinus resinosa*), and jack pine (*Pinus banksiana*) were predominant on uplands (Doenier et al. 1997). Northern white cedar (*Thuja occidentalis*), black spruce (*Picea mariana*), balsam fir, and tamarack (*Larix laricina*) were most prevalent on the lowlands. Common woody browse species included beaked hazel (*Corylus cornuta*), mountain maple (*Acer spicatum*), sugar maple (*Acer saccharum*), trembling aspen, and ironwood (*Ostrya virginiana*), among other species.

We calculated a MNDNR winter severity index (WSI) by accumulating 1 point for each day with a snow depth ≥38 cm and 1 point for each day with an ambient temperature ≤ -17.7°C during November-May. During winters 1990-1991 to 2004-2005, maximum WSIs ranged from 42 to 195, and maximum mean weekly snow depth ranged from 0 to 88 cm. Monthly mean daily minimum and maximum temperatures ranged from -28° to 1°C and -15° to 11°C, respectively (National Oceanic and Atmospheric Administration 1990-2005). During 1971-2000, the mean annual snowfall was 134 cm, and the mean temperature for January (coldest month) was -13.5°C (National Oceanic and Atmospheric Administration 2006).

METHODS

We compared available browse and winter food habits of white-tailed deer over a 13-year study period among 4 study sites by way of a technique employed earlier in northeastern Minnesota (Wetzel et al. 1975, DelGiudice et al. 1989). We collected browse availability and use data within 2-3 days of a fresh snowfall; just enough snow to determine fresh tracks along a feeding trail was necessary. Each of the 4 sites was divided into 10 grid cells so numbered. During 2-week intervals between mid-January and late-March, 6 cell numbers were drawn randomly without replacement for each of the 4 sites directing personnel where to go on each site to collect data on browse availability and use by deer along fresh feeding trails. In the field, the team would go to one of the randomly selected cells of a site, locate a primary, regularly used deer trail, and from that trail find a fresh feeding trail. We numbered feeding trails consecutively from 1 to 72 across study sites and 2-week intervals. Even-numbered trails were followed to the left, whereas odd-numbered trails were followed to the right. We used the SAS-generated random number between 1 and 20 (SAS Institute, Inc. 1988) associated with each numbered trail (1-72) to dictate the number of meters (between 1 and 20) we would travel before collecting data. These measures were taken to ensure that observers could not avoid dense browse patches or likewise be attracted to sparse patches. We recorded Universal Transverse Mercator coordinates for the starting point of each trail. We followed each trail for 200 m, counting current year's growth (CYG) of all twigs not browsed and all freshly browsed CYG (recorded separately) by species, in most cases, within 1 m of each side of the trail. During mid-January to late-March of 12 (Dir and Shi) and 13 (Wil and Ing) years, 1,036 feeding trails (with ≥400 CYG available, ≥30 CYG stems browsed), typically 200 m each, were followed to record browse availability and use.

Data Analyses

For the purposes of our data analyses, it was important to decide what constituted a reasonable feeding trail (i.e., minimum number of browse stems [CYG] available and browsed). Given our primary questions are about change or differences in browse availability and use over space and time, we based our analyses on records (i.e., feeding trails) where deer had a minimum level of stems to choose from and where they browsed on a minimum number of stems. Based on the density histogram of total browse available (Figure 1), we considered ≥400 available stems (~10th percentile) a reasonable cut-point. For total stems browsed, ≥30 stems (~4th percentile) was a reasonable cut-point (Figure 1). Using these criteria, we excluded 149 (13%) records, which left a reduced data set that was still reasonably large (1,036 records).

We used box-plot statistics to examine and compare descriptive statistics (median, mean, and variation) of (1) browse species used and available along white-tailed deer feeding trails, (2) niche breadth (Smith's index of heterogeneity) of their browse diet, (3) proportion of available browse species used (i.e., browsing intensity), and (4) abundance of browse stems available relative to shallow (< 19 cm), moderate (19-30 cm), and deep (>30 cm) snow cover on the 4 study sites and overall. We also used regression models to estimate the relationship between the proportion of browsed stems used and a continuous covariate for snow depth on the 4 sites and overall. A non-parametric bootstrap with B = 200 replicates was used to compute approximate 95% confidence intervals for the mean function.

RESULTS AND DISCUSSION

Sample sizes (i.e., numbers of feeding trails) were reasonably well-distributed among years and the 4 study sites, except during winters 1991-1992 (Shi and Dir sites were not yet part of the study) and 1997-1998 (first of 3 consecutive mild winters) (Figure 2). Sampling also was evenly distributed among the 4 study sites during winters of shallow (< 19 cm), moderate (19-30 cm), and deep (31-98 cm) snow cover, which limits concern of unintentional bias relative to snow conditions.

A total of 38 species of browse were available to deer along feeding trails during the 13-winter study period in north-central Minnesota, and 35 species were browsed (Figure 3). However, on average, most of these species contributed a small percentage to the available and used CYG stems. Specifically, 90% of browse species had mean relative availability values of < 8% (of total available), and mean relative use of most (75%) was < 5.8% (of total browsed). On average, beaked hazel and mountain maple were the 2 most commonly used species (Figure 3). Highly diverse winter diets, with a dominance by just a few species, such as beaked hazel and mountain maple, have been reported for deer in northeastern Minnesota as well (Wambaugh 1973, Wetzel et al. 1975, DelGiudice et al. 1989); however, these were short-term studies capturing limited variability of environmental conditions (e.g., snow cover, ambient temperature).

We observed no apparent change in mean or median number of browse species available and used along feeding trails as snow depth increased to 100 cm (Figure 4). Consequently, our findings do not support predictions 1a and 2a of an increase in diet richness and diversity (i.e., mean number and diversity of browse species) or of a decrease in mean number and diversity of browse species available along feeding trails with increasing depth of snow cover, respectively (Figure 4). Increasing snow depths and the associated increasing energetic costs of mobility for deer have been associated with reduced movements and home ranges, and increased use of dense conifer cover (Wetzel et al. 1975; DelGiudice et al. 2012, see our research summary use of cover), which presumably could cause or prompt deer to feed where the diversity of available and most choice browse species is less (Aldous and Smith 1938). Hobbs (1989:24) estimated that most (75%) of the negative impact of winter conditions on mule deer (*O. hemionus*) results from diminished energy intake caused by decreased forage availability. In northeastern Minnesota, availability and use of specific species of browse were reported to have changed from early to later winter with snow conditions, but overall diversity of availability and diet were not assessed (Wetzel et al. 1975, DelGiudice et al. 1989). An increase in diet diversity and number of species used as snow depths increased would indicate deer were increasingly relying on less preferred species. While overall we found no such change in diet diversity, there may be species-specific cases of this, which we will be investigating. Additionally, Figure 4 shows pronounced variation among feeding trails, but no indication that trails became more similar with respect to species richness of browse available or used as snow depths increased.

For assessments of niche breadth of the deer's winter browse diet, findings using Smith's index of heterogeneity (accounts for species richness and eveness) and Simpson's index of diversity and eveness of browse use were somewhat contradictory. Smith's index indicates that median niche breadth increased slightly when snow cover exceeded 31 cm (Figure 5); but most deer (i.e., feeding trails) exhibited relatively high resource/diet heterogeneity even when snow depths were shallow (< 19 cm). The variation among feeding trails was greater than that among study sites. In this specific application of Smith's index, support of predictions 1a and 2a (as above) is relatively weak and should be interpreted with caution. Contrary to Smith's index, Simpson's indices of diversity and eveness for browse use indicate that median diet diversity decreased slightly with increasing snow depth (>19 cm). These differences may reflect sampling variability; consequently, our computational approach may require more careful thought.

Similar among our 4 study sites, there was no evidence of a change in median or mean abundance of available browse stems when snow cover was shallow (< 19 cm), moderate (19-30 cm), or deep (31-98 cm) (Figure 6). Accordingly, the data did not support a prediction (2c) of decreased abundance of available browse along feeding trails as snow depth increased.

Our examination of changes in browsing intensity (i.e., proportion of available stems browsed) appeared to offer a clearer signal and interpretation than diversity indices at this point in our analyses of food habits relative to depth of snow cover. Median and mean browsing intensity were greatest when snow depth exceeded 30 cm (Prediction 1b), but variation among feeding trails also increased, and the patterns were similar among study sites (Figure 7). Wetzel et al. (1975) reported that deer browsing intensity (measured increasing diameter at point of browsing) increased as winter progressed for certain species (e.g., mountain maple, red maple [*Acer rubrum*], beaked hazel, and choke cherry [*Prunus virginianus]*), particularly in high use areas in northeastern Minnesota. We also used regression models to examine proportion of stems browsed versus snow depth on a continuous scale for all 4 sites (Figure 8). All regression coefficients were significant (at alpha = 0.05), but snow depth and site explained only 16% of the variation in proportion of total stems browsed. However, most interesting, the shape of the mean function matched Prediction 1b reasonably well, and more specifically, the model suggested that the mean proportion of stems browsed increased when snow depth exceeded 40 cm on all 4 sites (Figure 8). This is a critical depth for deer where energetic costs of movement become critical and costly bounding becomes common (Moen 1976). (We may attempt this analytical approach with the other endpoints [e.g., diversity indices], but because the signal-to-noise ratio is likely smaller, interpretations may be more difficult and less straightforward.)

This large winter food habits data set is unique in that it was accumulated by assessing browse (*by species*) available to and used by white-tailed deer over 13 winters, which allowed us to capture a wide breadth of winter conditions to assess impacts on overall diversity and on specific species. Further, our examination of the deer's food habits was part of a comprehensive, long-term study of many other aspects of their ecology, including nutritional condition, survival, cause-specific mortality, reproduction, and habitat use in a region of highly variable winter severities. Our study's deer survival and reproduction findings have shown that this is a region where deer thrive, despite the primary source of natural mortality of adults being predation by wolves (*Canis lupus*), black bears (*Ursus americanus*), and bobcat (*Felis rufus*) and relatively easy hunter access. With nutrition being central to all other aspects of an animal's ecology, these findings strongly suggest that this is a region of quality habitat and a reasonably good winter food supply. Increased understanding of the value of overall diet diversity to the winter nutritional condition and survival of deer, as well as of the potential value of key browse species has strong management implications with respect to their habitat in northern Minnesota. We will conclude our analyses of food habits with a focus on some of these key browse species.

ACKNOWLEDGMENTS

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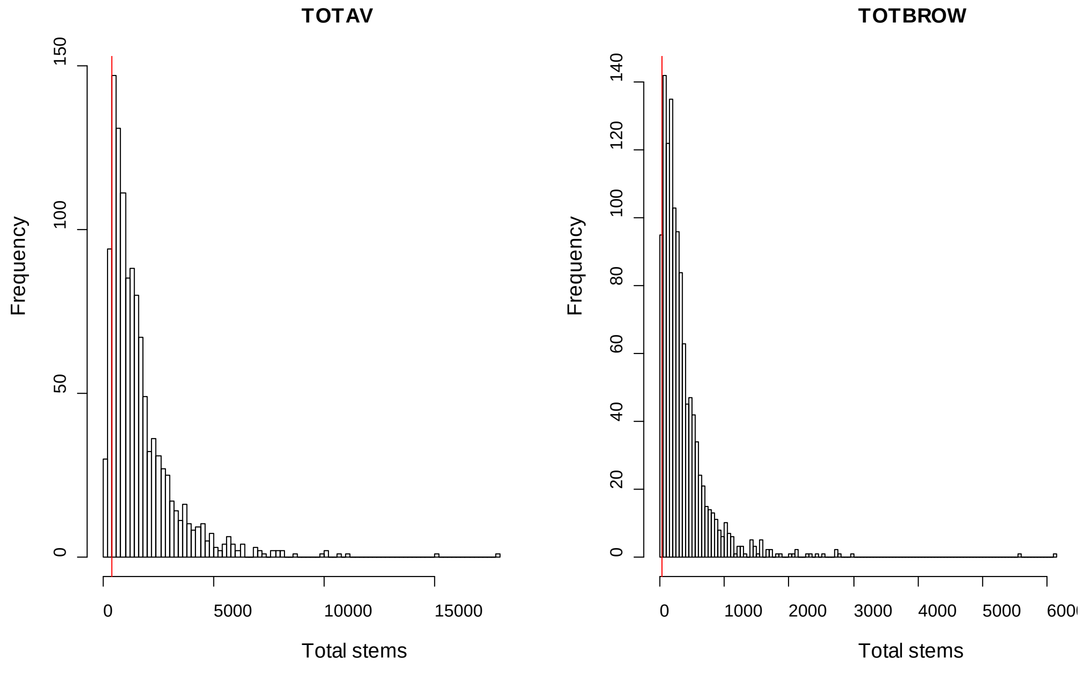

Figure 1. Density histograms of total browse stems available (left, TOTAV) and used (right, TOTBROW) by white-tailed deer on 4 study sites, north-central Minnesota, mid-January to March 1992-2005. Browse availability and use were not monitored during winter 1998-1999.

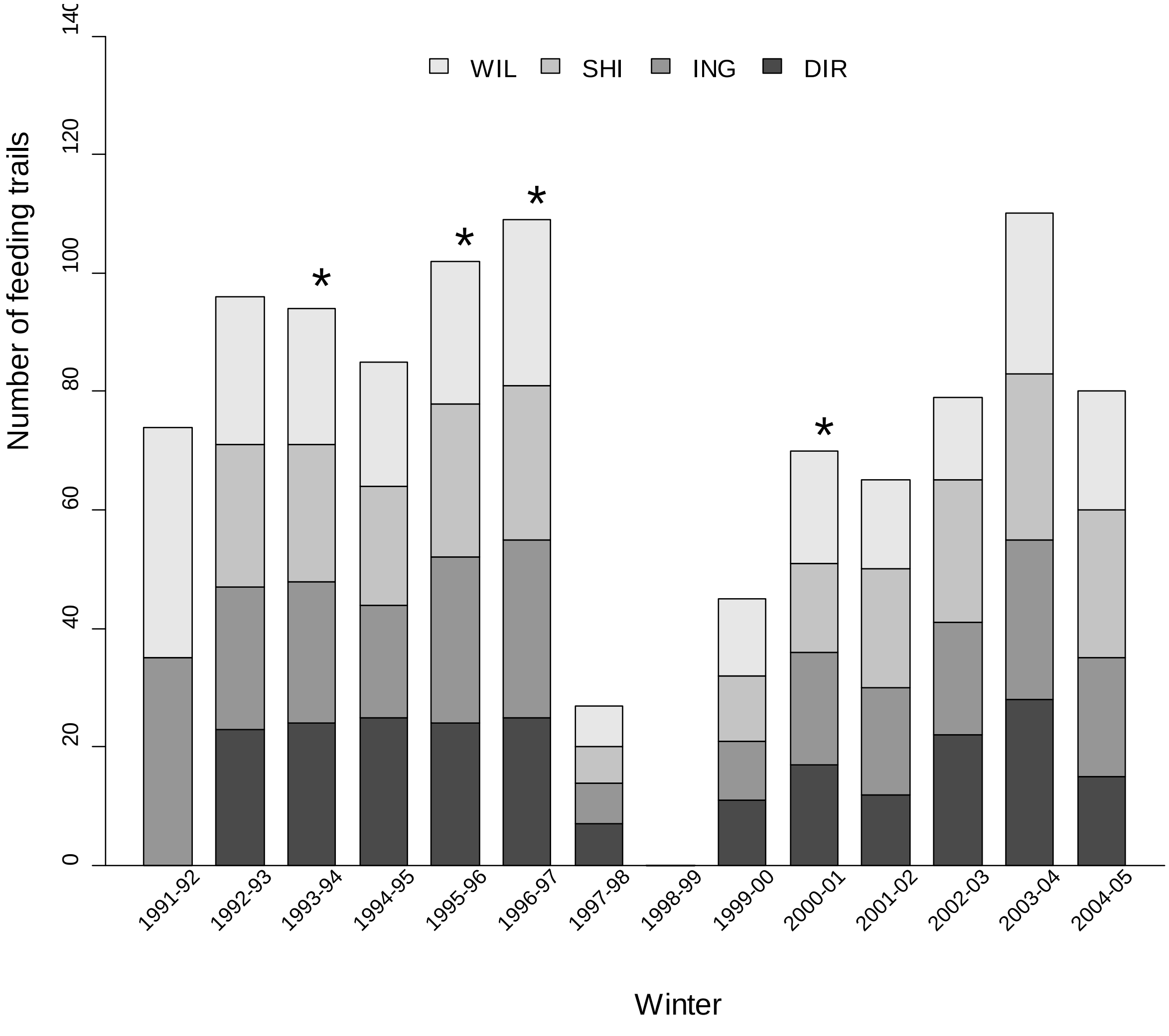

Figure 2. Distribution of total sample sizes (feeding trails with ≥400 browse stems available and ≥30 stems browsed) among years and 4 sites (WIL, SHI, ING, DIR) in a study of winter food habits of white-tailed deer, north-central Minnesota, winters 1991-1992 to 2004-2005. Browse availability and use were not monitored during winter 1998-1999. Asterisks denote the 4 most severe winters based on winter severity index and snow depth.

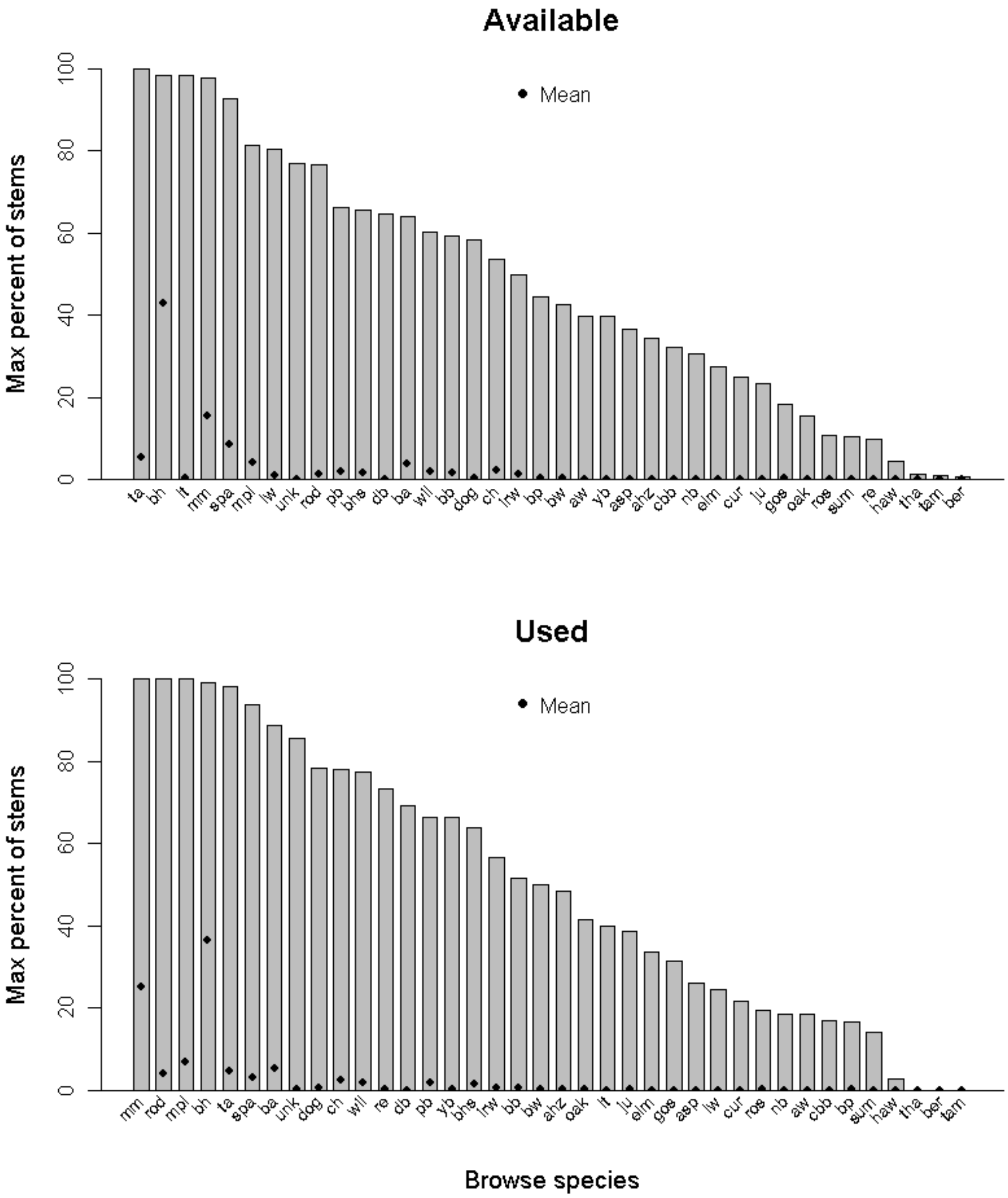

Figure 3. Mean and maximum percent of browse stems (by species) available and used by white-tailed deer on 4 sites (WIL, SHI, ING, DIR), north-central Minnesota, winters 1991-1992 to 2004-2005. Browse availability and use were not monitored during winter 1998-1999. (ta = trembling aspen, bh = beaked hazel, lt = Labrador tea, mm = mountain maple, spa = speckled alder, mpl = maple species [spp.], lw = leatherwood, unk = unknown spp., rod = red osier dogwood, pb = paper birch, bhs = bush honeysuckle, db = dwarf birch, ba = black ash, wil = willow spp., bb = blueberry, dog = dogwood spp., ch = cherry spp., irw = ironwood, bp = balsam poplar, bw = basswood, aw = arrowwood, yb = yellow birch, asp = aspen spp., ahz = American hazel, cbb = cranberry bush, nb = nannyberry, elm = elm spp., cur = currant spp., ju = juneberry, gos = gooseberry, oak = oak spp., ros = soe spp., sum = sumac, re = red elder, haw = hawthorn, tha = thornapple, tam = tamarack, ber = berry spp.)

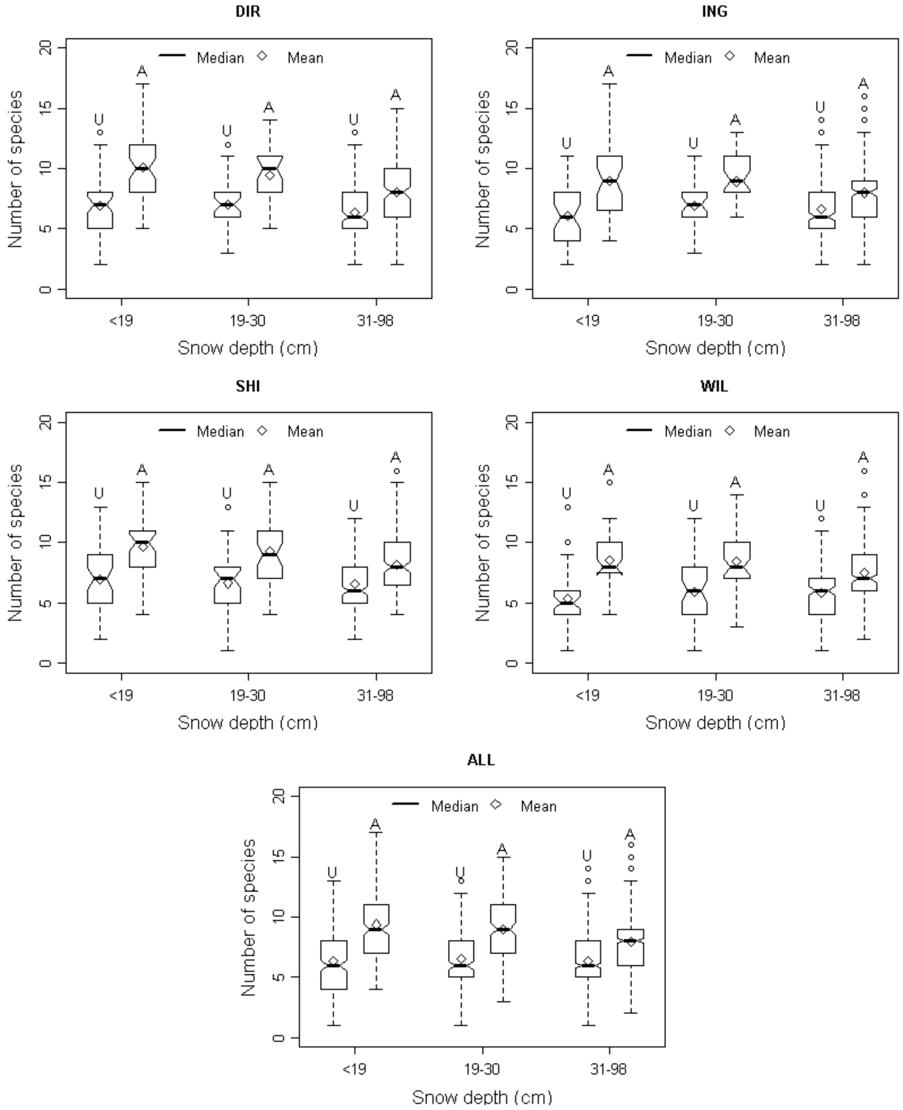

Figure 4. Mean and median number of browse species used (U) and available (A) along feeding trails of white-tailed deer when snow cover was shallow (< 19 cm), moderate (19-30 cm), and deep (31-98 cm) on 4 sites (WIL, SHI, ING, DIR), north-central Minnesota, winters 1991-1992 to 2004-2005. Browse availability and use were not monitored during winter 1998-1999. Box “hinges” are approximate 1st and 3rd quartiles (interquartile range contains ~75% of data); “whiskers” provide about a 95% confidence interval when based on asymptotic normality of the median.

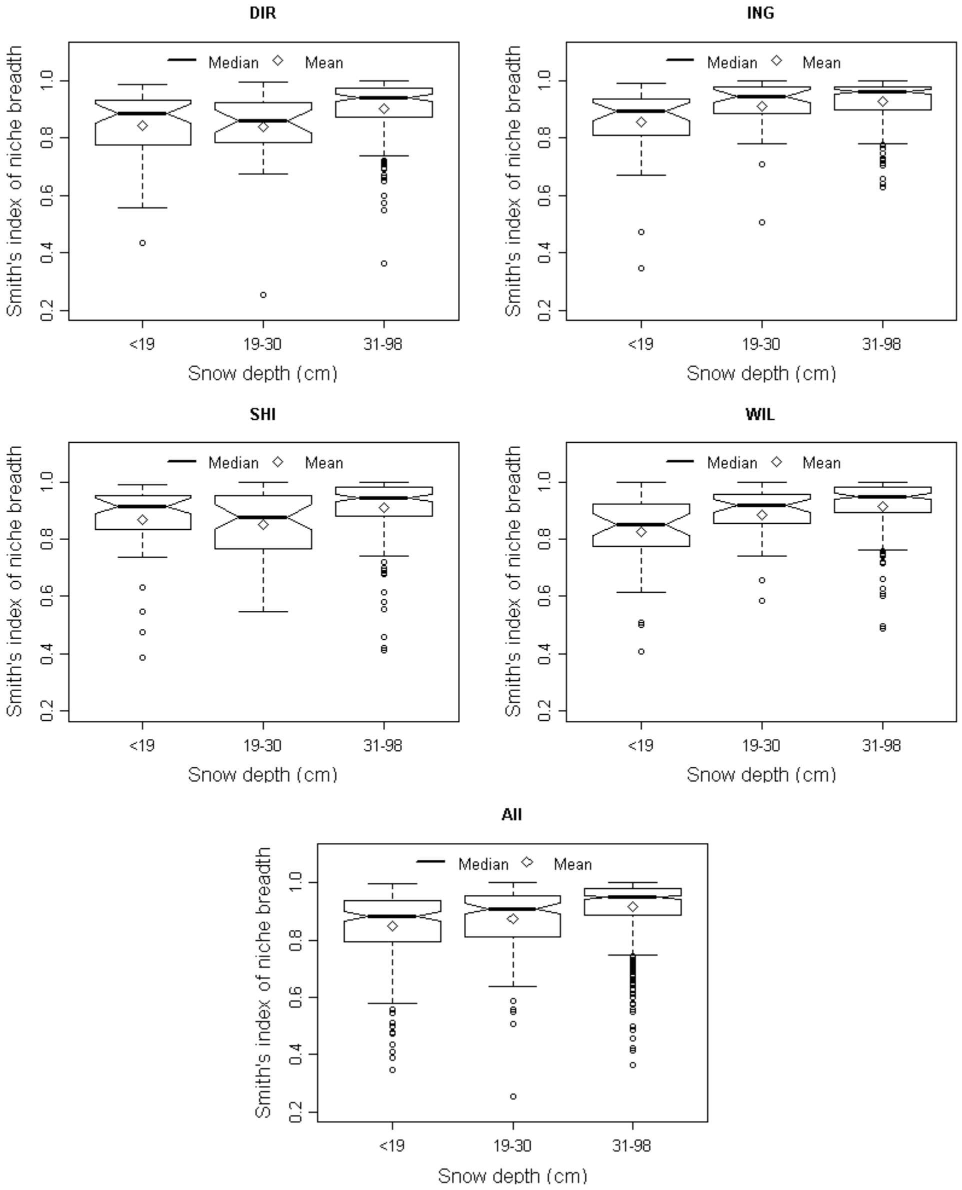

Figure 5. Median and mean niche breadth (Smith's index of heterogeneity, 1 = max diversity) of winter browse diet of white-tailed deer when snow cover was shallow (< 19 cm), moderate (19-30 cm), and deep (31-98 cm) on 4 sites (WIL, SHI, ING, DIR), north-central Minnesota, winters 1991-1992 to 2004-2005. Browse availability and use were not monitored during winter 1998-1999.

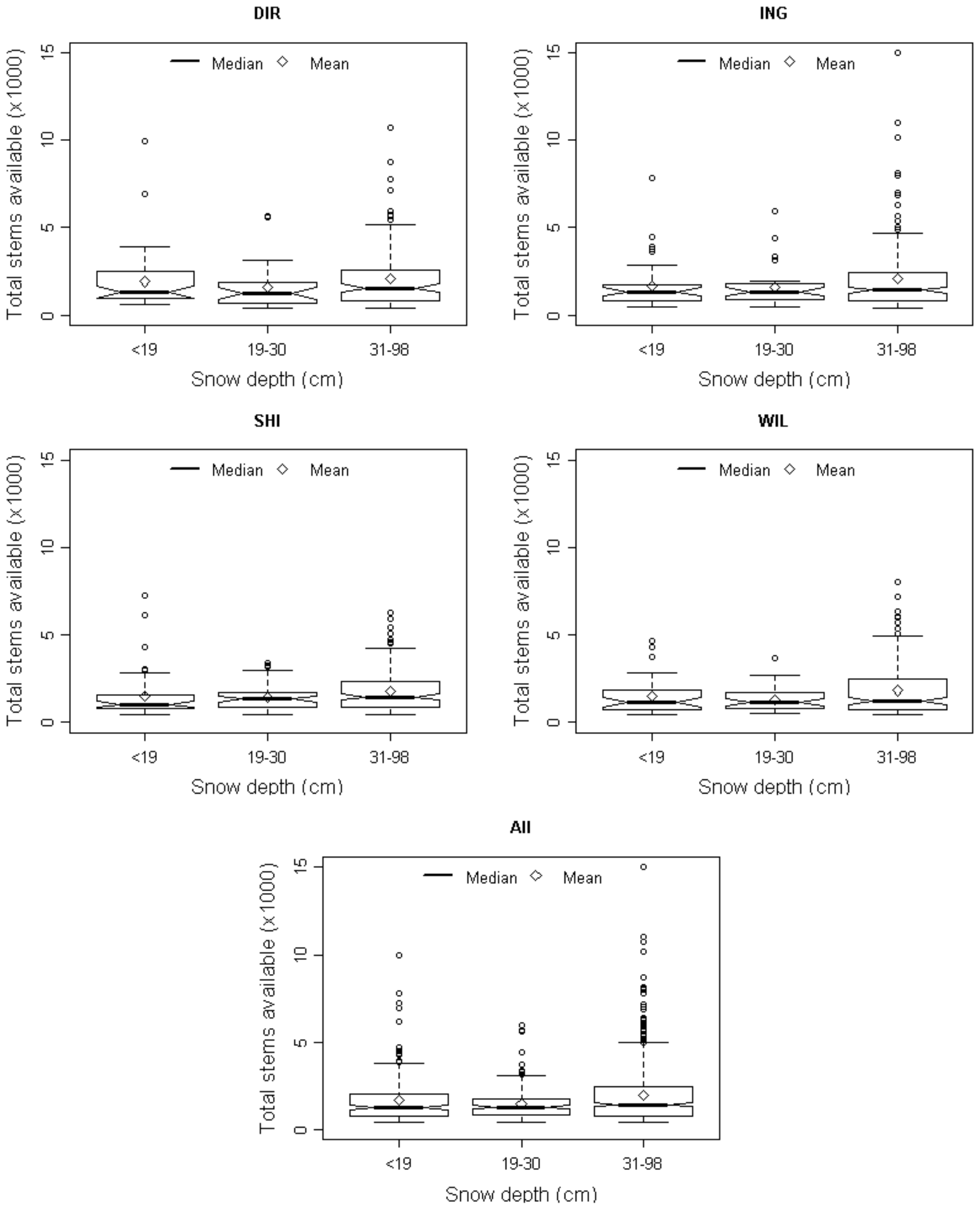

Figure 6. Median and mean abundance of browse stems available along feeding trails of white-tailed deer when snow cover was shallow (< 19 cm), moderate (19-30 cm), and deep (31-98 cm) on 4 sites (WIL, SHI, ING, DIR), north-central Minnesota, winters 1991-1992 to 2004-2005. Browse availability and use were not monitored during winter 1998-1999.

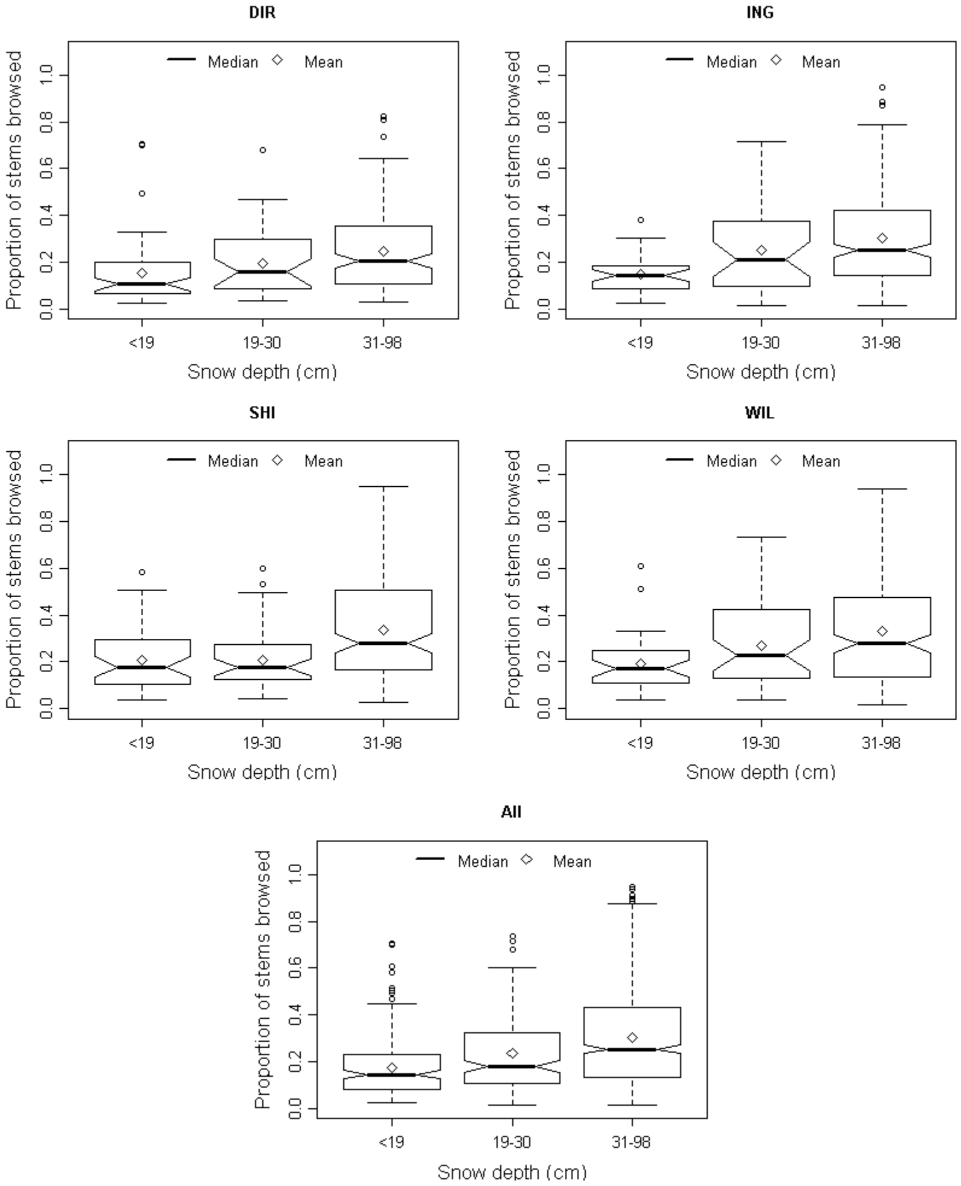

Figure 7. Median and mean proportion of available browse stems used by white-tailed deer when snow cover was shallow (< 19 cm), moderate (19-30 cm), and deep (31-98 cm) on 4 sites (WIL, SHI, ING, DIR), north-central Minnesota, winters 1991-1992 to 2004-2005. Browse availability and use were not monitored during winter 1998-1999.

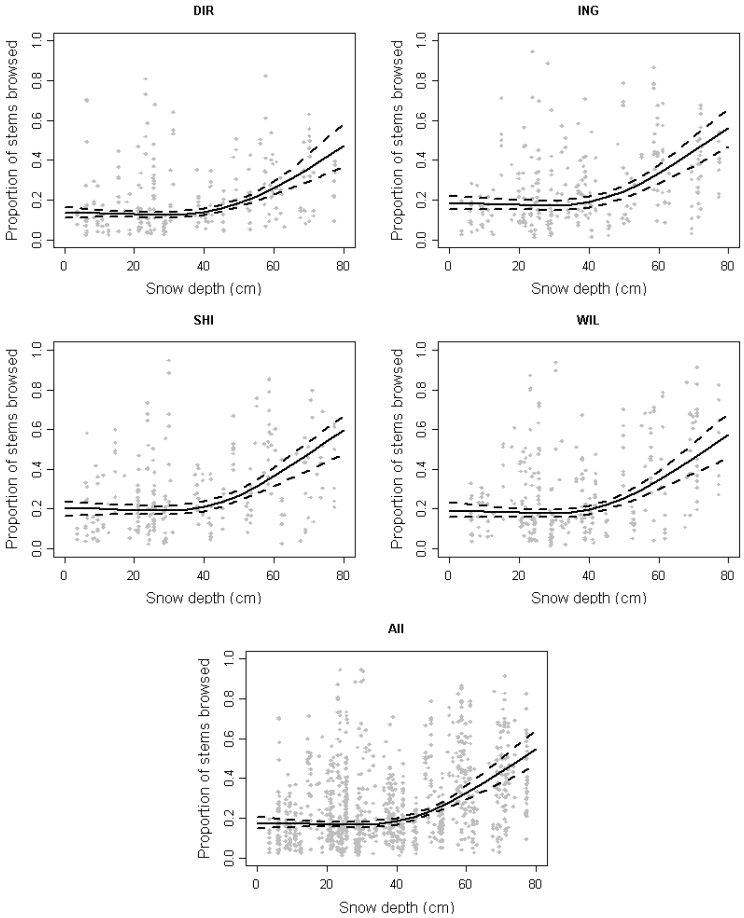

Figure 8. Regression relationship of the proportion of total browse stems used by white-tailed deer versus snow depth on 4 sites (WIL, SHI, ING, DIR), north-central Minnesota, winters 1991-1992 to 2004-2005. Browse availability and use were not monitored during winter 1998-1999. All regression coefficients were significant (at alpha = 0.05). Dashed lines represent 95% confidence intervals.

REPRODUCTIVE ECOLOGY OF FISHER AND MARTEN IN MINNESOTA

John Erb, Pam Coy, and Barry Sampson

SUMMARY OF FINDINGS

As part of a larger project on *Martes* ecology in Minnesota, we began monitoring reproductive success of radio-collared fishers (*Martes pennanti*) and martens (*Martes americana*) during spring 2009. Including the pilot year of the study, a total of 169 martens (80F, 89M) and 80 fishers (44F, 36M) have been radio-collared. To date, age and reproductive status have been confirmed on 18 adult (≥ 2 years old) female martens. Pooling years, pregnancy rate has been ~43% for 2-year-old martens, and ~90% for martens 3 years or older. Average size of 10 marten litters confirmed to date is 2.9. Based on initial data, it appears marten kits are typically born in mid- to late-April. A total of 18 marten natal or maternal dens have been located, of which 56% have been in tree (primarily cedar) cavities and 44% in underground burrows. We have also confirmed both age and reproductive status for 36 female fishers. Average size of fisher litters is 2.6 (range = 1–4). Initial data suggests that litter size and pregnancy rate for 2 year old fishers is lower than for older females (litter size: 2.1 versus 2.9; parturition rate: 53% versus 94%). 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. A total of 43 fisher natal or maternal dens have been confirmed, all but 2 being in elevated tree cavities. Cavities have been located in both live trees (73%) and snags (27%) with an overall average dbh of 20.6 in. Fisher dens have been located primarily in aspen (73%; average dbh ~21 in.) and oak (18%; average dbh ~19 in.) trees and most female fishers appear to move kits from their natal den to at least 2 different maternal dens prior to June 1.

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. 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 spatially variable, though generally 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 from 2007 - 2011 were reduced from 16 days to 9 days, and starting in 2010, the fisher harvest limit was reduced from 5 to 2. During this 5-year period, harvests have averaged ~2,100 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, Mead 1994). 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 in some species even if fetuses are resorbed (Conaway 1955), and may be affected by observer variability (Johnson et al. 1995). 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 (Mead 1994, Frost et al. 1999).

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 ≥ 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 are often highly variable. Conversely, in Minnesota, Kuehn (1989) did not detect changes in fisher pregnancy rate (from CL analysis) in spite of a 64% decline in a presumably important prey species (snowshoe hare).

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 many 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 1, 2 and 4. We defer a more complete evaluation of results until additional data are 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 radio-collared in Area 2 (Figure 1). Area 1 (~700 km^2) 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 km^2), 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). Fisher were either ear-tagged with a monel # 3 tag in one ear (National Band and Tag Co., Newport, KY) and a 2-piece plastic mini-tag (Dalton I.D. Systems, UK) in the other ear, or with a monel # 3 tag in both ears. Marten were ear-tagged with a monel #1 tag (National Band and Tag Co., Newport, KY) in each ear.

During processing, 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 as a precaution against infection (Kreeger et al. 2002) from minor wounds that may have occurred while in the trap, and because of certain invasive procedures utilized during handling (ear-tagging, removal of tooth).

During the pilot year, we deployed several radio-collar designs on fisher, including an ATS M1585 zip-tie collar (~43 g), an ATS M1930 collar (~38 g), and a Lotek 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 to provide ancillary information on winter activity and spring den attendance patterns, as well as to provide information on time of death for other study objectives.

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 den 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 'transferrable' 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 ft. 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 ft. coarse woody debris sampling transects are established, originating from the subplot center, and oriented at 30º, 150º, and 270º. 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; 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 the transects, and repeated once within each 10-day interval from 1 December – 1 April, 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≈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 protocols 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 detected 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 169 martens (80F, 89M) and 80 fishers (44F, 36M) have been radio-collared. Herein we provide a basic summary of data collected to date on den use chronology, den structures, pregnancy status, and litter size. Because tooth aging has not yet been completed for all animals, some of which 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). As of this writing, spring 2012 reproductive status assessment is largely complete for radio-collared fishers, but still ongoing for many martens.

Treating females that were alive during multiple parturition periods (years) as independent units, and excluding females known to be 1 year of age, a total of 31 female martens have been available for monitoring during the kit-rearing season. However, to date we have only confirmed age and reproductive status for 18 female martens, 10 of which we have also confirmed litter size (Table 1). Pooling years, pregnancy rate has been ~43% for 2-year-olds, and ~90% for martens 3 years or older. Because it has been comparatively 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 10 litters confirmed to date is 2.9 (Table 1). 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-December through early February), 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 18 marten natal or maternal dens have been located to date (Table 2). Based on 14 dens confirmed prior to June 1 of each year, 64% have been in tree cavities, whereas 36% have been in underground burrows (Table 2). We have confirmed only 5 maternal dens used after June 1, and 4 were in underground burrows situated in rock-laden soils with the remaining one in above- and below-ground structure created from an exposed root mass (Table 2). Most female marten appear to move their kits from their natal den to 1 or more maternal dens in the first 6 weeks following birth.

Similar to marten, we treat female fishers that were alive during multiple parturition periods (years) as independent units. Excluding individuals known to be 1 year of age during the parturition period, we have confirmed both age and reproductive status for 36 female fishers (Table 3). Pooling all female fishers that produced a litter that we were able to count (n=27), average litter size is 2.6 (range = 1–4). Age-specific sample sizes are small (Table 3), but there is some indication that average litter size for 2-year-olds is lower than for older females (~2.1 versus 2.9). There is also some indication that birth rates are lower for 2-year-olds compared to older females. Parturition rate for known 2-year-olds (n=17) is 53% whereas parturition rate for female fishers known to be ≥3 years of age (n=17) is 94% (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 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 seasonal hormone profiles developed in Maine (Frost et al. 1999). Furthermore, although blood has not been collected on all females, all those confirmed pregnant in mid-winter (progesterone) produced litters in spring, suggesting that overwinter disruption of pregnancy is not common.

We have confirmed a total of 43 fisher natal or maternal dens (Table 4). All but 1 natal/maternal den located prior to June 1 of each year has been in an elevated tree cavity. One additional maternal den located after June 1 was in a hollow log on the ground (15.7 in. diameter sugar maple). Of the natal/maternal dens located prior to June 1, detailed measurements have only been completed on 38. Cavities have been located in both live trees (73%) and snags (27%) with an overall average dbh of 20.6 in. Dens have been located in aspen (73%; average dbh ~ 21 in.), oak (18%; average dbh ~ 19 in.), white pine (5%; average dbh ~23 in.), and 1 each in a red maple, sugar maple, white cedar, and an elevated hollow aspen log (Table 4). Although monitoring has not been standardized across animals, most female fishers appear to move kits from their natal den to at least 2 different maternal dens prior to June 1.

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Table 1. Parturition status and litter size for radio-collared female marten[1] in Minnesota.

ID	Year	Age	Litter	Litter size
M10-616	2011	2	N	
M10-7311	2011	2	N	
M11-621	2011	2	N	
M11-726	2012	2	N	
M09-254	2010	2	Y	>=3
M09-280	2010	2	Y	>=3
M12-016	2012	2	Y	
M09-264	2009	2 or 3	N	
M09-262	2009	2 or 3	Y	
M09-243	2009	3	N	
M09-262	2010	3 or 4	Y	4
M09-247	2009	5	Y	4
M11-780	2011	5	Y	1
M08-140	2008	9	Y	
M09-286	2009	9	Y	>=3
M08-140	2009	10	Y	>=2
M09-286	2010	10	Y	>=4
M09-237	2010	11	Y	4
M12-014	2012		Y	
M12-024	2012		Y	1

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

Table 2. Natal and maternal den structures used by radio-collared 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
M12-016	2012	5/1	natal	tree cavity	live cedar
M12-014	2012	5/7	natal	tree cavity	live cedar
M09-286	2010	5/7	natal	tree cavity	21.5" dbh live cedar
M12-024	2012	5/9	natal	tree cavity	14.6” dbh black ash
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	18.6” dbh live cedar
M09-286	2009	5/22	maternal	tree cavity	20.9" dbh live cedar
M09-254	2010	5/26	maternal	underground burrow	rock-laden soil
M09-286	2010	6/12	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
M09-254	2010	7/12	maternal	underground burrow	rock-laden soil

Table 3. Parturition status and litter size for radio-collared female fishers[1] in Minnesota.

ID	Year	Age	Litter	Litter size
F09-362	2009	1 or 2	N	
F10-501	2010	1 or 2	N	
F09-364	2009	2	N	
F10-503	2010	2	N	
F11-009	2011	2	N	
F11-651	2011	2	N	
F11-008	2011	2	N	
F12-078	2012	2	N	
F12-074	2012	2	N	
F12-536	2012	2	N	
F08-375	2008	2	Y	>=2
F09-360	2009	2	Y	2
F08-304	2009	2	Y	2
F08-077	2009	2	Y	4
F10-328	2010	2	Y	2
F10-501	2011	2	Y	2
F11-316	2011	2	Y	1
F10-503	2011	2	Y	2
F12-054	2012	2	Y	2
F11-009	2012	3	N	
F09-394	2009	3	Y	3
F08-353	2009	3	Y	3
F08-375	2009	3	Y	3
F10-507	2010	3	Y	3
F11-008	2012	3	Y	2
F11-316	2012	3	Y	2
F09-380	2009	4	Y	3
F09-394	2010	4	Y	2
F08-353	2010	4	Y	3
F10-507	2011	4	Y	3
F08-353	2011	5	Y	3
F09-394	2011	5	Y	3
F10-507	2012	5	Y	4
F09-394	2012	6	Y	3
F08-353	2012	6	Y	3
F09-370	2009	11	Y	3
F09-461	2010		Y	3
F11-052	2011		Y	>=1
F12-073	2012		Y	2

[1] 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
F10-507	2011	3/9	natal	tree cavity	16.9” dbh aspen snag?
F09-394	2012	3/13	natal	tree cavity	24" dbh live red oak
F08-353	2011	3/16	natal	tree cavity	16.4" dbh live aspen
F10-503	2011	3/16	natal	tree cavity	26.1” dbh aspen snag
F08-353	2012	3/16	natal	tree cavity	live aspen
F09-394	2012	3/22	maternal	tree cavity	live aspen
F08-353	2010	3/24	natal	tree cavity	15.1" dbh live aspen
F09-394	2010	3/26	natal	tree cavity	24.9” dbh live aspen
F10-507	2010	3/26	natal	tree cavity	25.6” dbh live oak
F10-507	2012	3/26	natal	tree cavity	20.3" dbh live aspen
F12-073	2012	3/26	natal	tree cavity	22.2" dbh live aspen
F12-054	2012	3/26	natal	tree cavity	16.7" dbh live aspen
F11-316	2012	3/27	natal	tree cavity	26.2" dbh live aspen
F09-394	2011	3/28	natal	tree cavity	15.8” dbh live aspen
F11-008	2012	3/28	natal	tree cavity	live sugar maple
F10-501	2011	3/30	natal	tree cavity	20.3” dbh live cedar
F11-316	2011	3/31	natal	tree cavity	13.6” dbh aspen snag
F11-052	2011	4/1	natal	tree cavity	23.8" dbh live aspen
F08-375	2009	4/7	natal	tree cavity	21.9” dbh w. pine snag
F08-353	2009	4/8	natal	tree cavity	23.2" dbh live aspen
F09-360	2009	4/8	natal	tree cavity	15.3” dbh aspen snag
F10-503	2011	4/8	maternal	tree cavity	18” dbh live red maple
F09-394	2009	4/9	natal	tree cavity	13.8” dbh aspen snag
F09-394	2010	4/9	maternal	tree cavity	22.1” dbh live aspen
F09-394	2011	4/9	maternal	tree cavity	24” dbh live aspen
F09-461	2010	4/11	natal	tree cavity	18.3" dbh live oak
F08-353	2011	4/11	maternal	tree cavity	19.0" dbh live aspen
F10-507	2011	4/12	maternal	tree cavity	15.1” dbh live oak
F10-507	2012	4/12	maternal	tree cavity	aspen
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
F11-316	2011	4/14	maternal	tree cavity	16.6” dbh live oak
F09-370	2009	4/15	natal	tree cavity	23.5” dbh aspen snag
F09-394	2009	4/18	maternal	tree cavity	21.5” dbh live aspen
F12-073	2012	4/18	maternal	tree cavity	aspen snag
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
F10-503	2011	4/25	maternal	tree cavity	23.7” dbh live pine
F09-394	2011	5/4	maternal	tree cavity	19.8” dbh live aspen
F10-507	2011	5/9	maternal	Hollow log off ground	18.3” diam. aspen
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 on ground	15.7" diam. sugar maple

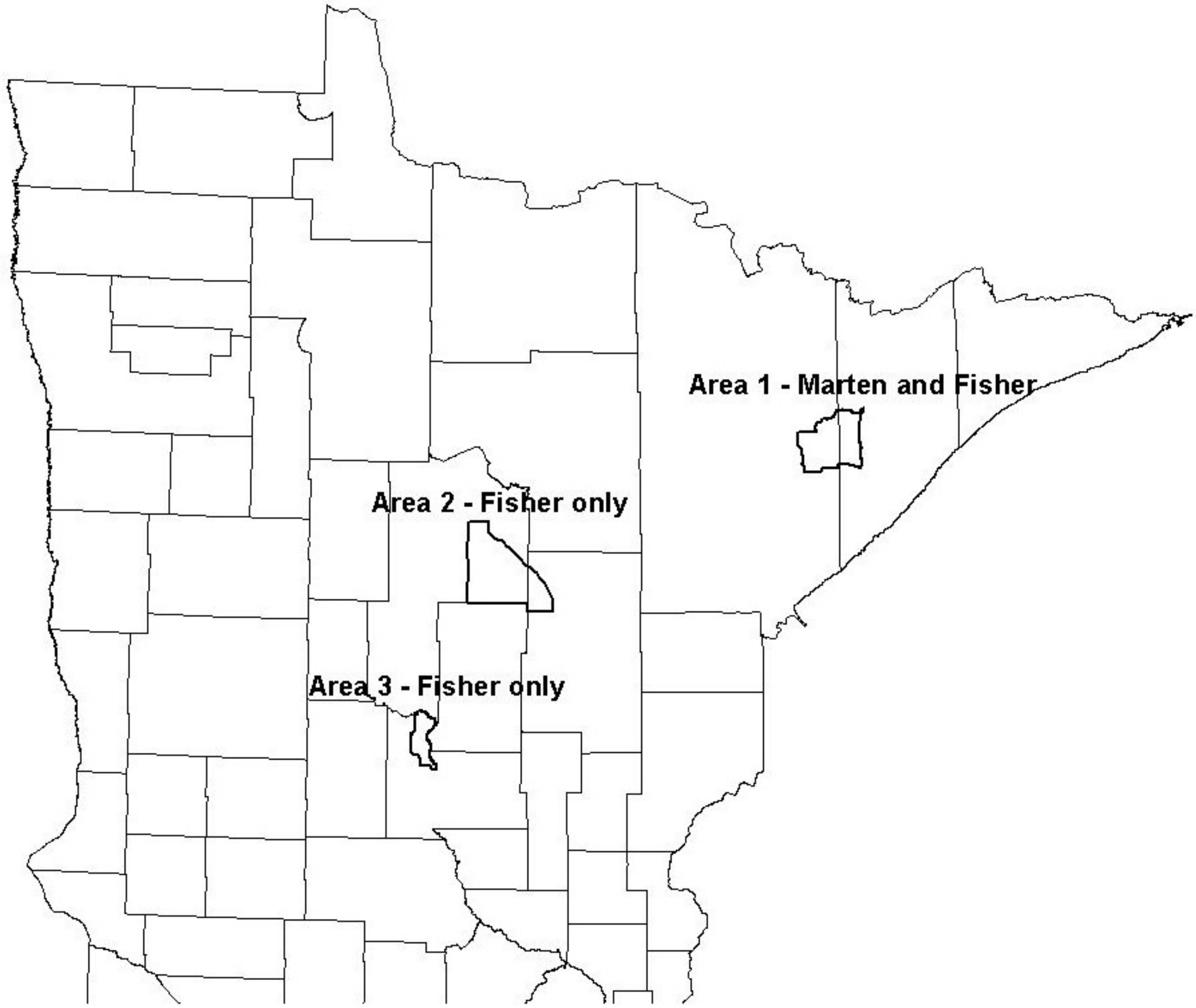

Figure 1. Fisher and marten study areas in Minnesota, 2008-2012.

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 radio-collared fishers (*Martes pennanti*) and martens (*Martes americana*) during winter 2007-08. Including the pilot year of the study, a total of 169 martens (80F, 89M) and 80 fishers (44F, 36M) have been radio-collared. An additional 7 animals (4 martens, 3 fishers) were ear-tagged only. Of the 169 martens radio-collared, 48 are still actively monitored (22F, 26M), radio-contact has been lost on 29, 5 whose collars are inaccessible have either slipped their collars or died, and 87 deaths have been confirmed (of which 10 were censored due to death within 2 weeks of capture). Of the 77 non-censored marten deaths (31F, 46M), most have been from regulated fur trapping (n=35; 26M, 9F) and predation (n=37; 22F, 15M). Approximately 81% of the 37 marten predation deaths have been attributed to other mammalian carnivores and 19% to raptors. Although predation mortality of martens has occurred in most seasons, the majority has occurred during late winter and spring. No significant sex bias has been observed in marten deaths attributed to natural causes (n=42; 48% males), while human-caused marten mortality has been significantly male-biased (n=35; 74% males).

Of the 80 fishers radio-collared, 23 are still being monitored (14F, 9M), radio contact was lost on 24, and 33 deaths (18F, 15M) have occurred. Of the 33 fisher deaths, 11 (33%) were attributed to humans [2 (1M, 1F) hit by cars and 9 (6M, 3F) trapped], whereas 22 (67%) were attributed to natural causes (17 (5M, 12F) predation deaths and 5 (3M, 2F) from unknown natural causes). Nearly all of the predated fishers were killed in spring. Eleven of the 12 female fisher predation mortalities were attributed to other mammalian carnivores, while 4 of the 5 male fisher predation mortalities were attributed to raptors (all bald eagles). Of greatest significance, 11 of the 12 female fishers killed by predators were adults, and 10 of the 11 adult females were killed while they still had dependent young in natal dens, indirectly resulting in the death of all their offspring. The deaths of these 10 nursing females represent ~28% of the adult female fishers monitored during the kit-rearing season since the study began. We hypothesize that the timing and magnitude of female mortality is a result of increased movement and increased vulnerability at this time of year. However, it remains unclear whether the 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 largest single 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. While marten harvest is legal in approximately the northern 50% of the state, most 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 from 2007-2011 were reduced from 16 days to 9 days, and starting in 2010, the fisher harvest limit was reduced from 5 to 2. During this 5-year period, harvests have averaged ~2,100 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 radio-collared 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 fisher 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 juvenile fishers 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 juvenile fishers 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.

Natural mortality, particularly via predation, appears more common with martens. Marten survival data is available from Wisconsin (McCann et al. 2010), 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). Although 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 were 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 previously projected a very female-biased population, contradicting our preliminary capture results and suggesting that our model inputs were overestimating female survival, underestimating male survival, or incorrectly assuming 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 statistical 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 radio-collared in Area 2 (Figure 1). Area 1 (~700 km^2) 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 km^2), 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). Fisher were either ear-tagged with a monel # 3 tag in one ear (National Band and Tag Co., Newport, KY) and a 2-piece plastic mini-tag (Dalton I.D. Systems, UK) in the other ear, or with a monel # 3 tag in both ears. Marten were ear-tagged with a monel #1 tag (National Band and Tag Co., Newport, KY) in each ear.

During processing, 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 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). Antibiotics were administered subcutaneously to all animals prior to release as a precaution against infection (Kreeger et al. 2002) from minor wounds that may have occurred while in the trap, and because of certain invasive procedures utilized during handling (ear-tagging, removal of tooth).

During the pilot year, we deployed several radio-collar 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, are 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 169 martens (80F, 89M) and 80 fishers (44F, 36M) have been radio-collared. An additional 7 animals (4 martens, 3 fishers) were ear-tagged only. 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.

Excluding 10 martens that were censored due to death within 14 days post-release, 48 (30%) of the 159 collared martens are actively being monitored, radio contact has been lost on 29 (18%), the status of 5 (3%) is uncertain, and 77 (49%) have died (Table 1). Of the 77 non-censored deaths, most have been from regulated fur trapping (n=35; 9F, 29M) and predation (n=37; 22F, 15M). Five animals, all males, died of other natural causes, including being crushed by a tree, perforation and blockage of the intestine from a piece of bone, starvation related to an intestinal polyp, and 2 from unknown natural causes.

Of the 37 non-censored predation deaths, evidence suggests 30 (81%) were killed by mammalian predators and 7 (19%) by raptors. Approximately two-thirds of the predation mortalities have occurred in late-winter through spring (i.e., Feb – May; Figure 2). Forensic (DNA) analysis of samples collected from predated marten (mammalian predation only) is incomplete. To date, DNA analysis has confirmed bobcat predation in all 5 cases for which analysis is complete. Felids (bobcat or lynx) are the likely predator in 5 additional cases for which partial information (inconclusive DNA or obvious field sign) is available. Remaining forensic analysis is pending, and field evidence suggests fox or fisher may be responsible for at least a couple predation deaths on marten.

Excluding censored animals, our sample of radio-collared marten has been comprised of 45% females, while females have accounted for 40% of the total marten deaths, 17% of the total deaths due to harvest, and 59% of the predation deaths. Combining predation with other sources of natural mortality, females represent 52% of natural deaths.

Of the 80 fishers radio-collared, 23 (29%) are still being monitored, radio contact has been lost with 20 (25%), the status of 4 (5%) is uncertain, and 33 (41%) have died (Table 1). Of the known deaths, 33% were attributed to humans [2 (1M, 1F) hit by cars and 9 (6M, 3F) trapped] while 67% were attributed to natural causes [17 (5M, 12F) predation deaths (scavenging by an eagle can't be ruled out in 1 case) and 5 (3M, 2F) from unknown natural causes].

All of the predation deaths of fishers took place in late winter and spring (i.e., Feb. – May; Figure 3), similar to the pattern observed with marten. Four of the 5 male fisher predation deaths were attributed to bald eagles, though we can't rule out scavenging in 1 case (only the radio-collar was retrieved directly underneath an active eagle nest). The fifth male fisher predated appears to have been killed by a wolf. Conversely, only 1 of the 12 female predation deaths was attributed to a raptor (Great-horned owl suspected). We are awaiting forensic DNA analysis on many fishers killed by mammalian predators. However, bobcat was confirmed (DNA and via trail camera) in one case, and field evidence strongly indicates bobcat in 1 additional case, and bobcat or lynx in another. Wolf predation is suspected in the death of 1 female fisher.

Of greatest significance, 11 of the 12 female fishers killed by other predators were adults, and 10 of those 11 were killed while they still had dependent kits in natal or maternal dens, indirectly resulting in the death of all their dependent offspring. The deaths of these 10 kit-rearing females represent ~28% of the adult females monitored during the kit-rearing season since the study began.

We hypothesize that 2 broad factors may explain the high mortality of kit-rearing females during late-winter and spring: increased movement, and increased vulnerability (independent of movement). First, female fishers likely have high energetic demands after 'emerging' from winter, compounded by the added energy demands of gestation and lactation. Furthermore, the need to locate suitable (and multiple) natal or maternal dens likely increases movement. Preliminary data from temperature data loggers attached to radio-collars suggests that fishers do spend increasing amounts of time (compared to winter) outside of den and rest sites during late-winter and spring. Regardless of the motivation for increasing activity, increased movement likely increases the risk of predation. Secondly, independent of their activity level, fishers may be more vulnerable in spring because: 1) concealment cover is diminished (i.e., before 'green-up'); 2) other predators may also increase activity in spring.

Regardless of the explanation, and acknowledging the limited sample size, it seems unlikely that the high level of predation we have observed to date on nursing female fishers is sustainable, which may partially explain the recent decline in fisher abundance. However, many of 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 ~ 7 years, suggesting that other factors may be altering the 'system'. While it seems unlikely that the fisher mortality pattern we have observed to date is consistent with past dynamics, it remains uncertain whether the changing dynamics are related to short-term (e.g., periodic fluctuations in prey) or long-term (e.g., suspected decline in natal and maternal den availability) changes affecting fisher energetics or activity, the result of relatively rapid changes in the predator community (i.e., the rapid increase in bobcats, which are known or suspected to be responsible for many of the female fisher predation deaths), or some other unknown factor.

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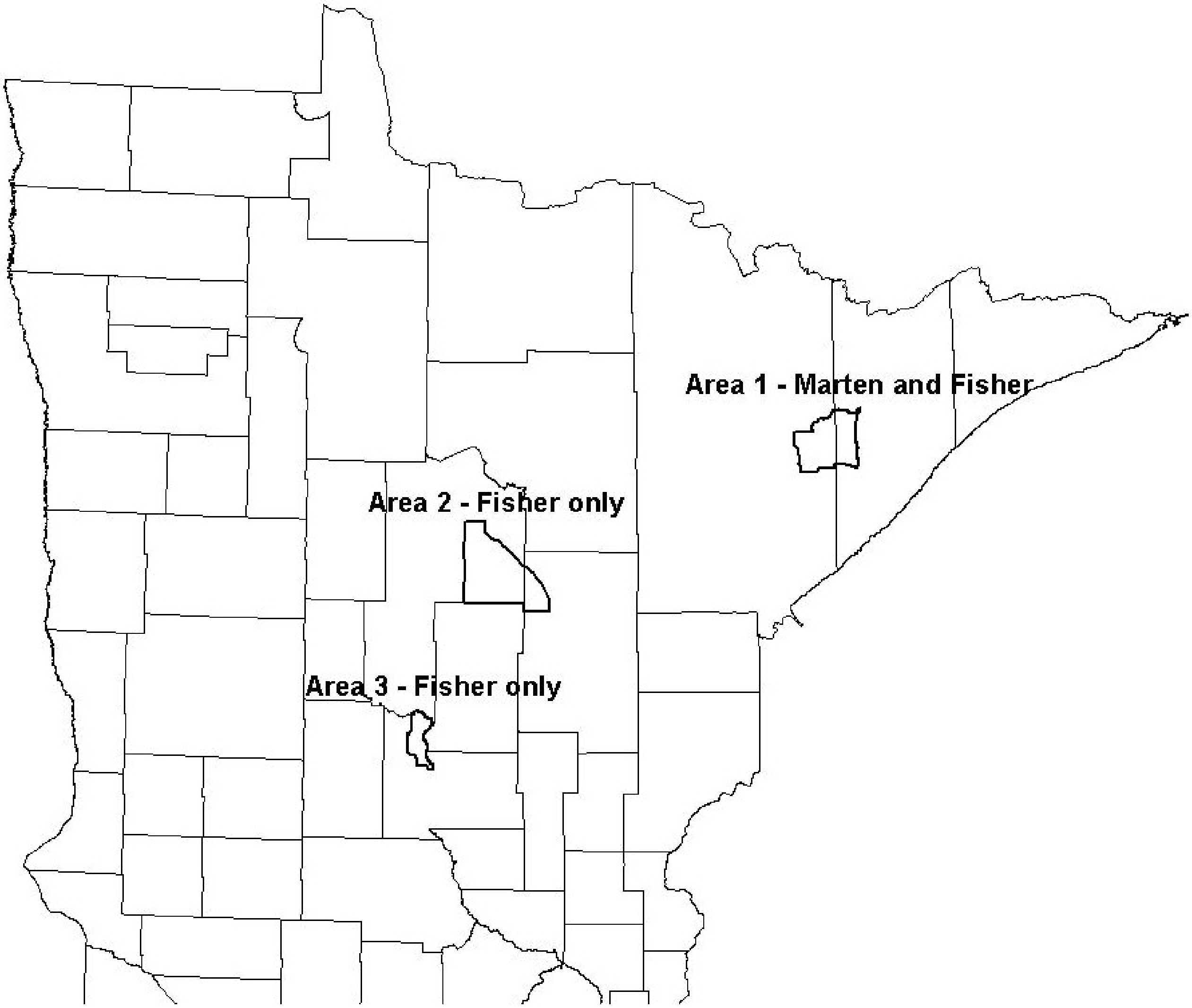

Figure 1. Fisher and marten study areas in Minnesota 2008-2012.

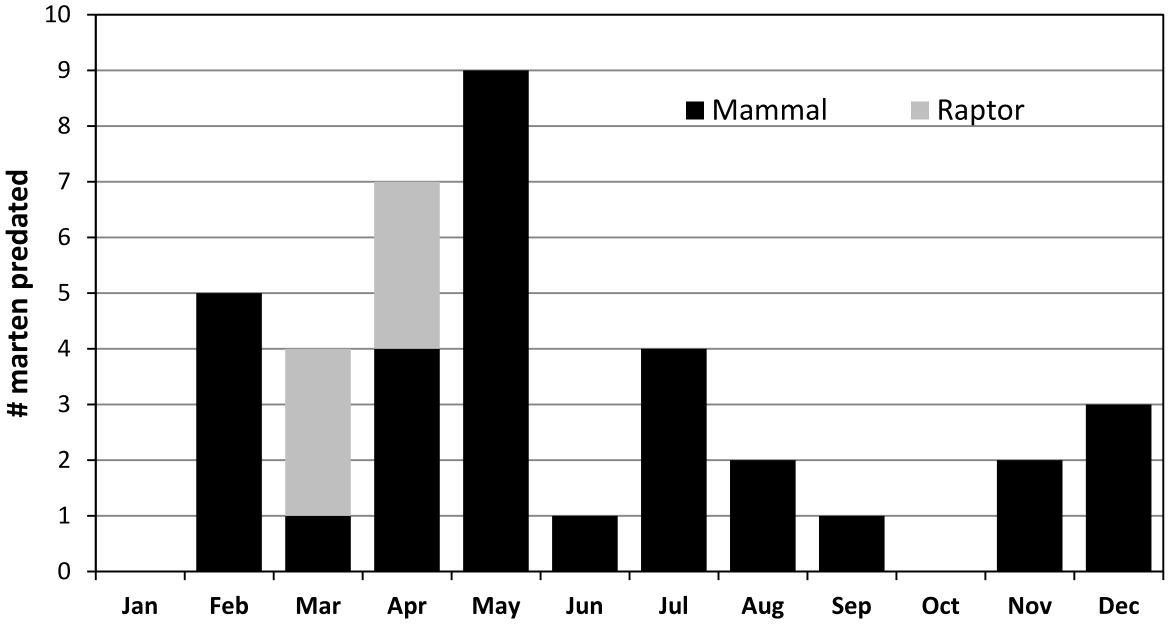

Figure 2. Seasonal timing of marten deaths attributable to predation in northeast Minnesota, 2007-2012.

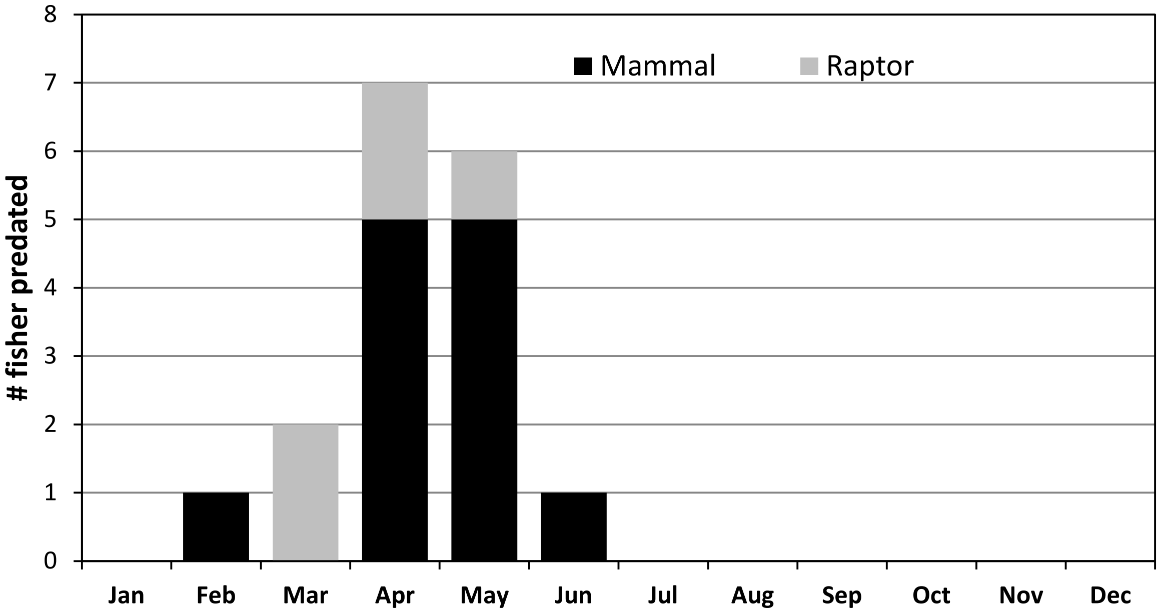

Figure 3. Seasonal timing of fisher deaths attributable to predation in north-central Minnesota, 2007-2012.

Table 1. Number[1] and status of fishers and martens radio-collared in Minnesota from winter 2007 through winter 2011.

Sex*Species	# Active	# Deaths	# Missing	# Slipped Collars	Unknown[2]	Total
Male Martens	26	46	10	3	1	86
Female Martens	22	31	12	4	4	73
Male Fishers	9	15	3	7	2	36
Female Fishers	14	18	3	7	2	44

[1] Excludes radio-collared animals that died within 2 weeks of capture and release

[2] Unknown represents collars not yet retrieved from tree cavities or underground locations (presumed dead or slipped collars), or retrieved but with uncertainty whether the animal slipped the collar or had died.