

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 2010–March 2011, we monitored 33 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. This marks the 30th year of our study. Hunting has been the primary source (~80%) of mortality in all areas, even though, for the last 10 years, hunters were asked not to shoot radiocollared bears and bears cannot be legally hunted in 2 of the areas (but can be hunted when they wander outside). Reproduction was highest at the fringes of the bear range (NW and Camp Ripley), due largely to an abundance of oaks and hazelnuts in these areas. Data from Global Positioning System (GPS)-radiocollars indicated that males in the NW made significant use of croplands (corn and sunflowers) from August–October (25% of fall locations). Females in this area rarely used crops, but instead spent much of their time in aspen woodlands and shrublands. Continuation of this work will aim to explain this sex-related disparity in habitat use and predict whether further expansion of the bear range is possible.

INTRODUCTION

Thirty years ago, the Minnesota Department of Natural Resources (MNDNR) initiated research on black bears, spurred by concerns of low population size, and commensurate with management programs to restrict the harvest and enable the population to grow. For the first 10 years, the study was limited to the Chippewa National Forest (CNF), near the center of the Minnesota bear range (Figure 1). 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).

These study sites differ enormously. 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 un hunted, 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 un hunted, 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 key respects: (1) it is largely agricultural (including croplands, like corn and sunflowers, that bears consume), (2) most of the land, including various small woodlots, is privately-owned, with some larger blocks of forest contained within MNDNR 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).

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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. Determine habitat use (including crop use) for bears in an agricultural fringe area; and
5. Predict range expansion of bears in northwestern Minnesota (not covered in this report).

METHODS

We attached radiocollars with breakaway and/or expandable devices to bears either when they were captured during the summer or when they were handled as yearlings in the den with their radiocollared mother. We trapped bears this year only in the NW study site, using barrel traps baited with raw bacon, and anesthetized them with ketamine-xylazine. In this area, we used principally GPS-collars, programmed to collect locations at designated intervals (varying from 6 hr to 20 min, depending on time of year). Most GPS collars used this year were “pods” (Telemetry Solutions, Concord, California) that were bolted onto normal VHF collars.

During December–March, we visited all radio-instrumented bears once or twice at their den-site. We immobilized bears in dens with an intramuscular injection of Telazol, administered with a jab stick or Dan-Inject dart gun. Bears were then removed from the den for processing, which included changing or refitting the collar, attaching a first collar on yearlings, measuring, weighing, and obtaining blood and hair samples. All GPS data were downloaded from collars of denned bears (GPS pods were either removed from the bear or downloaded at the densite). We also measured bioelectrical impedance (to calculate percent body fat) and vital rates of immobilized bears. Additionally, collaborators from the University of Minnesota (Dr. Paul Iaizzo) and Medtronic (Dr. Tim Laske) measured heart condition with a 12-lead EKG and ultrasound on a select sample of bears, and implanted (subcutaneously) a miniature heart monitoring device (developed for humans) that will record heart rate, body temperature, and activity throughout the year. Bears were returned to their dens after processing.

We assessed reproduction by observing cubs in dens of radiocollared mothers. We sexed and weighed cubs without drugging them. We evaluated cub mortality by examining dens of radiocollared mothers the following year: cubs that were not present as yearlings with their mother were presumed to have died.

During the non-denning period we monitored survival of radio-instrumented bears periodically through the summer. We listened to their radio signals, and if a pulse rate was in mortality mode (no movement of the collar in >4 hours), we tracked the collar on the ground to locate the dead animal or the shed radiocollar. During the hunting season (1 September–mid-October), hunters typically (but not always) reported collared bears that they killed.

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

We quantified food production and other site characteristics in representative habitats that bears used in the NW study site. We did this in 2 ways: (1) sampling GPS locations of bears from previous years (sampling them at the same time of year as the bear’s location), and (2) sampling random plots. Within each sampling plot, we separately estimated the percent areal coverage and productivity of all principal fruiting species that bears consume. We visually rated fruit production on a 0–4 scale (0 = no fruit, 2 = average fruiting, 4 = bumper crop). We also collected and counted fruits from a sample of bushes to enable conversion of our subjective ratings to estimates of biomass. For GPS bear plots we also quantified herbaceous, shrub, and canopy layers, soil moisture, and horizontal visibility.

RESULTS AND DISCUSSION

Radiocollaring and Monitoring

Since 1981 we have handled >800 individual bears and radiocollared >500. As of April 2010, the start of the current year's work, we monitored 33 collared bears: 5 in the CNF, 8 at Camp Ripley, 3 in VNP, and 17 in the NW. We captured 7 more bears in the NW study site during June (4 males, 3 females), and collared them, 6 with GPS-collars. However, 2 GPS collars put on bears in dens failed, 2 GPS collars put on in summer failed, 5 collars were dropped, and 1 dispersing yearling bear apparently traveled beyond the search area and could not be located. We collared 3 yearling bears during March, 2011.

Mortality

Legal hunting has been the dominant cause of mortality among radiocollared bears from all study sites; over the 30 years of our study, 77% of mortalities that we observed were due, or likely due to hunting. For the 2 sites where hunting was legal (CNF, NW), 81% of bears died from this cause (Table 1). In earlier years of this study, hunters were encouraged to treat collared bears as they would any other bear so that the mortality rate of collared bears would be representative of the population at large. With fewer collared bears left in the study, and the focus shifted to reproduction and habitat use rather than mortality, we sought to protect the remaining sample of bears. We asked hunters not to shoot radiocollared bears, and we fitted these bears with bright orange collars and colorful eartags so hunters could more easily see them. However, the mortality rate for collared bears has remained high even though some hunters reported avoiding them, and most of those who shot them said they saw the tags and were aware of our request to not shoot them. Ironically, on the CNF prior to asking hunters not to shoot collared bears (1981–2000), 81% died due to hunting, whereas since the request not to shoot collared bears (2001–2010), 92% died due to hunting.

This year hunters legally killed 3 collared NW bears, and we surmised that 1 other was killed by a hunter who cut off the collar and left it in the woods. Two of 5 collared bears were shot by hunters in the CNF. A number of other collars were found in the woods, but with no indication that the bears had been shot.

Although nuisance kills have been the second-most common cause of bear mortality overall, across all study areas and years (Table 1), few collared bears have been killed as nuisances in recent years (most of the nuisance-related mortalities among collared bears occurred in the 1980s). This year, 1 NW bear was killed as a nuisance; however this was an unusual case. This bear did not come from this area: it was orphaned as a cub, raised for a few months in captivity (at a rehab facility), and released in late fall with 3 other orphaned cubs in the Thief Lake WMA. In the spring, it traveled 26 km west and entered a small town, where it could not be scared away, so was shot. Another of this group of released orphans was a nuisance elsewhere, but not killed; it subsequently dropped its collar and was shot by a hunter. This was our first attempt to release orphaned, captive-raised cubs in the NW study site. We have had better success doing so within the CNF study site (1 was released in the CNF in late fall, 2010).

One other mortality of a collared bear occurred this year: a bear from Camp Ripley denned in a cornfield outside the Camp and was run over by the farmer's combine; its yearling offspring also died.

Reproduction

We visited 8 dens of females with cubs during March, 2011 (including 1 outside our 4 study sites). On the 4 study sites, since 1982, we have checked 251 litters with 644 cubs (2.6 cubs/litter), of which 52% were male (Tables 2–5). Overall, first-year mortality averaged 21%, and mortality of male cubs (26%) exceeded that of females (15%). However, there appears to have been a change in these rates through time: during the most recent 5 years (2007–2011), litters have been slightly larger (2.7 cubs/litter) with increased numbers of females (50.9% male), but also increased female cub mortality (26%, versus only 19% for males). The timing and causes of cub mortality are unknown.

Reproductive rates were highest in the 2 study areas at the periphery of the bear range (NW, Camp Ripley) and lowest in VNP (Figure 2). 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.

Habitat Use of 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 (Figure 3); they spent, on average 33% to nearly 50% of their time there, whereas this forest type comprised only 13% of the area of the region, in a patchy distribution (e.g., small WMAs, Agassiz NWR, and private lands). Lowland shrub was the second-most used habitat (~25% 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) only 8% of the time. Also, females rarely used agricultural crops (Figures 3–4). Their high reproductive rate (Figure 2), which should be reflective of high food availability, especially in fall, is thus somewhat of an enigma; however, there appeared to be an abundance of hazelnuts (*Corylus americana*, *C. cornuta*) and dogwood berries (mainly *Cornus racemosa* and *C. sericea*) in these habitats (quantification of the food abundance data by habitat is ongoing). Males, in contrast, were frequently found in croplands during fall, on average spending about 25% of their time there. Although about half the landscape was comprised of agricultural crops, the crops that bears consumed (mainly corn and sunflowers) represented a small areal coverage, equivalent to oak forests (2%); male bears spent more time feeding on crops than on acorns, even in years when acorns were plentiful. Both sexes were also periodically found in soybean and wheat fields (Figure 4); we are uncertain whether they fed on wheat, but have observed them occasionally feeding on soybeans.

We highlight an interesting example of a male bear that fed in a sunflower field because it exemplifies several notable issues with management implications. This adult male, collared since 2008, has lived principally in Agassiz NWR. In 2010 he rarely moved outside the refuge throughout the year. In fall he found a sunflower field just outside the western edge of the refuge (Figure 5). He first entered this field on August 1, but did not use it on a daily basis until August 13, at which time he used it every day through September 24, almost entirely at night (Figure 6). He then left, but returned again on September 30 and used the field daily until October 6, at which point he traveled 10 km to immediately den (8 October) in a wilderness area in central Agassiz (he has denned in this vicinity each year). Clearly this bear was entirely reliant on anthropogenic food in the fall, feeding in the cropfield daily for 7 weeks; notably, there are very few oaks in Agassiz NWR, so this sunflower field was probably the closest concentrated food source available. We visited his den, an excavation into a peat “island”, on March 10, at which time he weighed 185 kg (406 lbs).

FUTURE DIRECTIONS

We plan to continue monitoring bears on these 4 study sites, although sample sizes have been greatly diminished by the exceedingly high harvest of collared bears in the past few years. We will continue to collect GPS-collar data in the NW study site. In addition to gaining information from radiocollars, we have been and will continue to interview farmers to collect additional data on bear use of crops. This will yield a historical perspective on crop use, and provide insights into specific varieties of corn and sunflowers used by bears. Moreover, we have obtained a collection of hair samples from hunter-killed bears in the NW for stable isotope analysis to ascertain the importance of corn in the diet, relative to gender and location. Ultimately we aim to create a habitat suitability map and thereby predict how far the bear population is likely to expand in this part of the state.

ACKNOWLEDGMENTS

We thank the collaborators in this study: B. Dirks at Camp Ripley, Dr. P. Iaizzo at the University of Minnesota, and Dr. T. Laske at Medtronic, Inc. A. Tri and M. Lyons assisted with fieldwork in the NW.

Table 1. Causes of mortality of radiocollared black bears ≥ 1 year old from the Chippewa National Forest (CNF), Camp Ripley, Voyageurs National Park (VNP), and northwestern (NW) Minnesota, 1981–2010. 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	10	259
Likely shot by hunter ^a	8	1	0	3	12
Shot as nuisance	22	2	1	1	26
Vehicle collision	12	8	1	1	22
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	18	352

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

Table 2. Black bear cubs examined in dens of radiocollared mothers in or near the Chippewa National Forest during March, 1982–2011. High hunting mortality of radiocollared bears has 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%	—
Overall	176	463	2.6	52%	19%

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

Table 3. Black bear cubs examined in dens in or near Camp Ripley Military Reserve during March, 1992–2011.

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%	—
Overall	33	78	2.4	51%	22%

^a Cubs that were absent from their mother's den as yearlings were considered dead. Blanks indicate no cubs were born to collared females or collared mothers with cubs died before the subsequent den visit to assess cub survival.

Table 4. Black bear cubs examined in dens in Voyageurs National Park during March, 1999–2011. 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%	—
Overall	26	58	2.2	50%	27%

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

Table 5. Black bear cubs examined in dens in northwestern Minnesota during March, 2007–2011.

Year	Litters checked	No. of cubs	Mean cubs/litter	% Male cubs	Mortality after 1 yr ^a
2007	2	6	3.0	33%	100% ^b
2008	5	15	3.0	67%	22%
2009	1	3	3.0	33%	33%
2010	6	17	2.8	41%	13%
2011	2	4	2.0	75%	—
Overall	16	45	2.8	50%	20% ^c

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

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

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

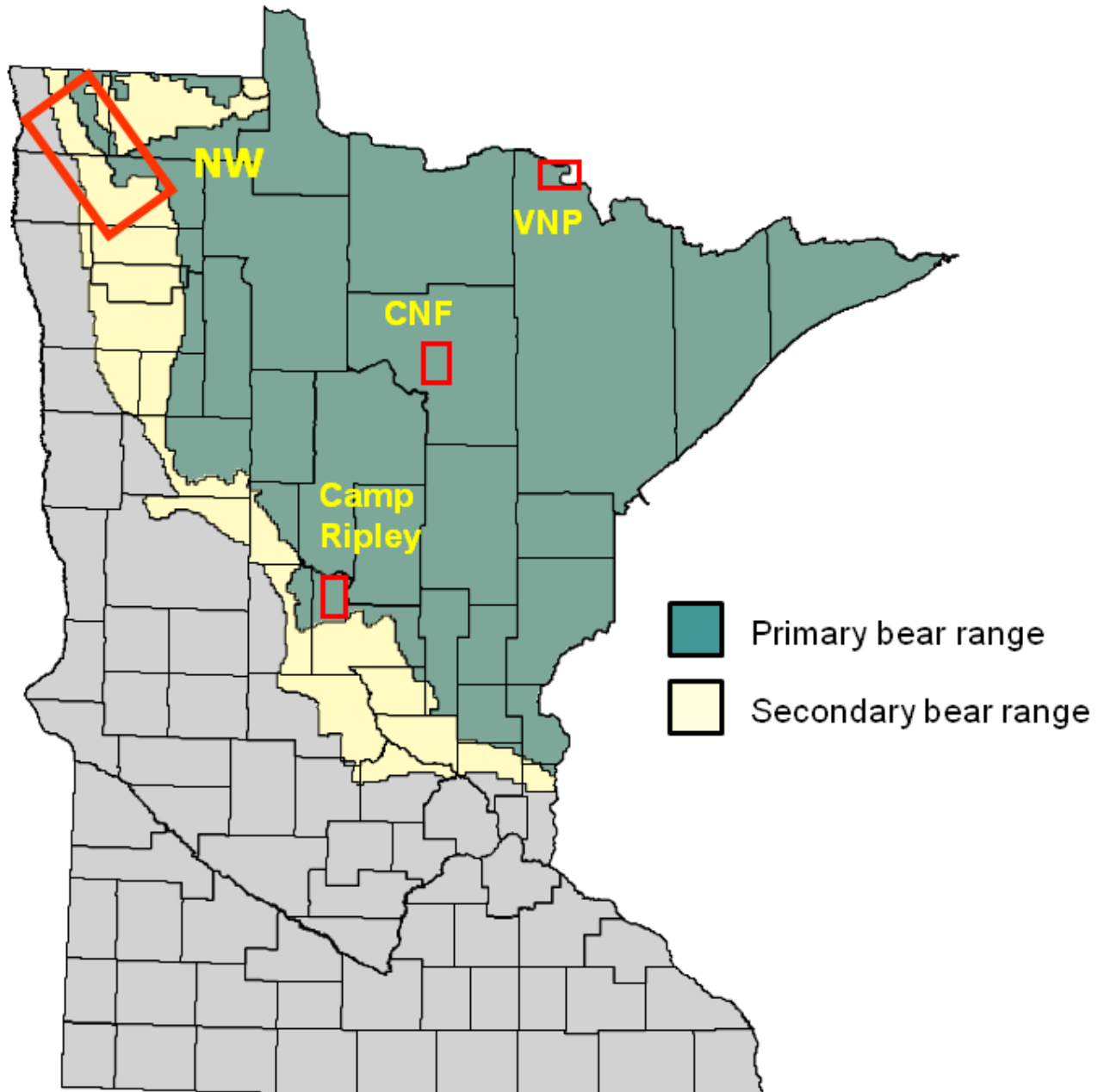


Figure 1. Location of 4 study sites within Minnesota's black bear range: CNF (Chippewa National Forest, central bear range; 1981–2011); VNP (Voyageurs National Park, northern fringe of range; 1997–2011); Camp Ripley Military Reserve (near southern edge of range; 1991–2011); and NW (northwestern fringe of range; 2007–2011).

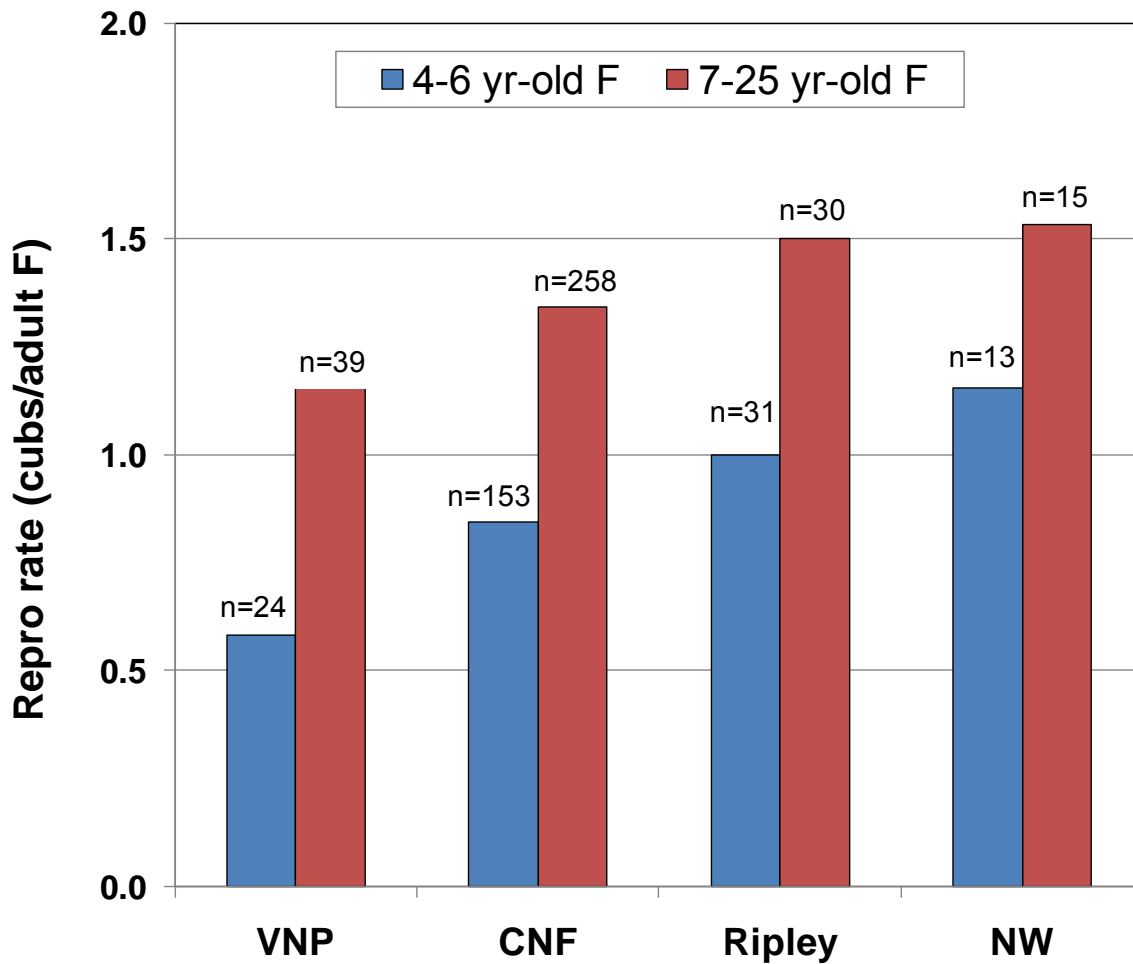


Figure 2. Reproductive rates of radiocollared black bears within 4 study sites (see Figure 1). Sample sizes refer to the number of female bear-years of monitoring in each area for each age group. Data include only litters that survived 1 year (even if some cubs in the litter died).

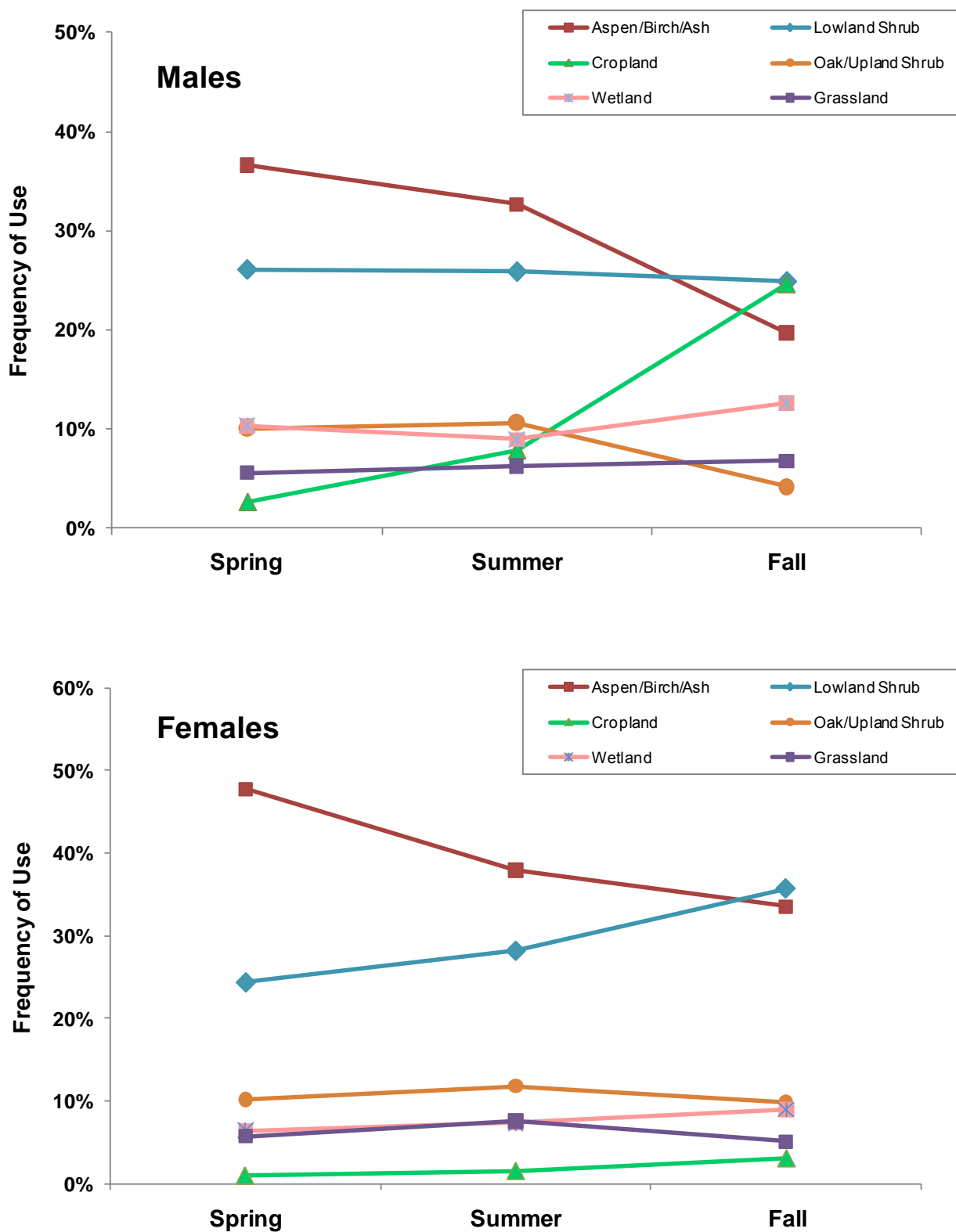


Figure 3. Seasonal habitat use of GPS-collared black bears in northwestern Minnesota, 2007–2010. Values for each season do not sum to 100%, because bears also infrequently used habitat types other than those graphed. Seasons were defined as follows: spring (April [den exit]–mid-June); summer (mid-June–August); fall (September–denning).

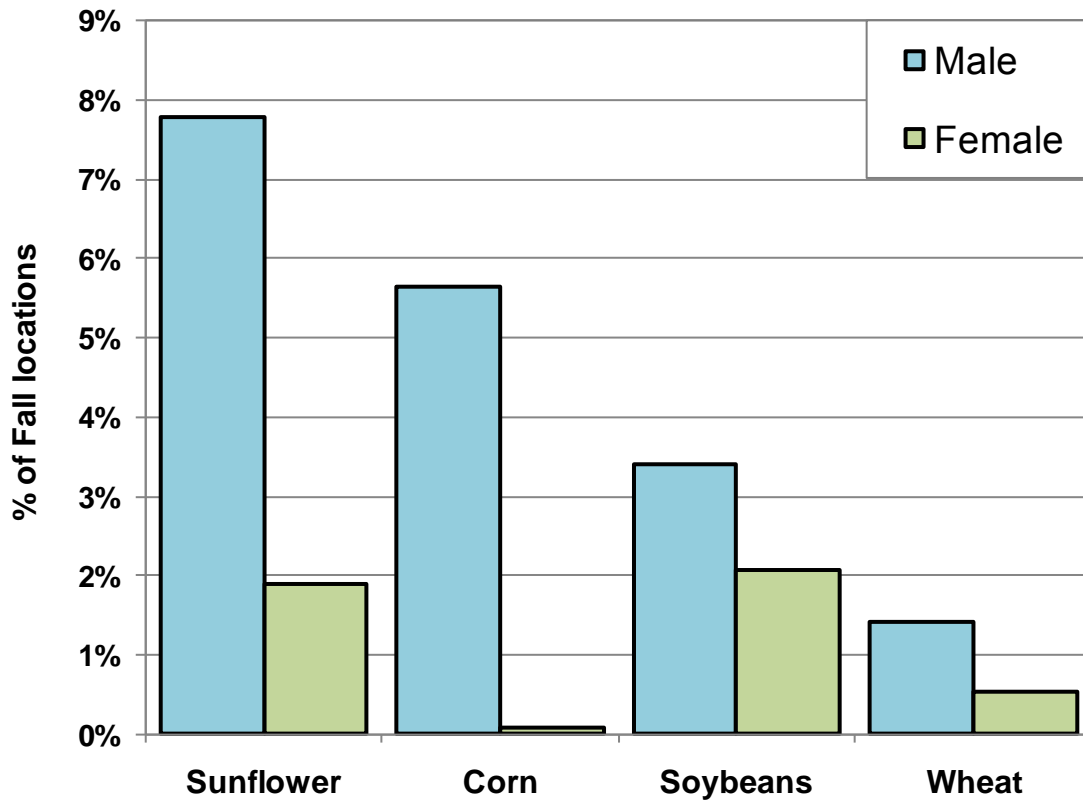


Figure 4. Use of crops by GPS-collared black bears in northwestern Minnesota during fall 2010. Values for each sex do not sum to the total cropland use shown in Figure 3, because the data graphed here are only for 1 year.

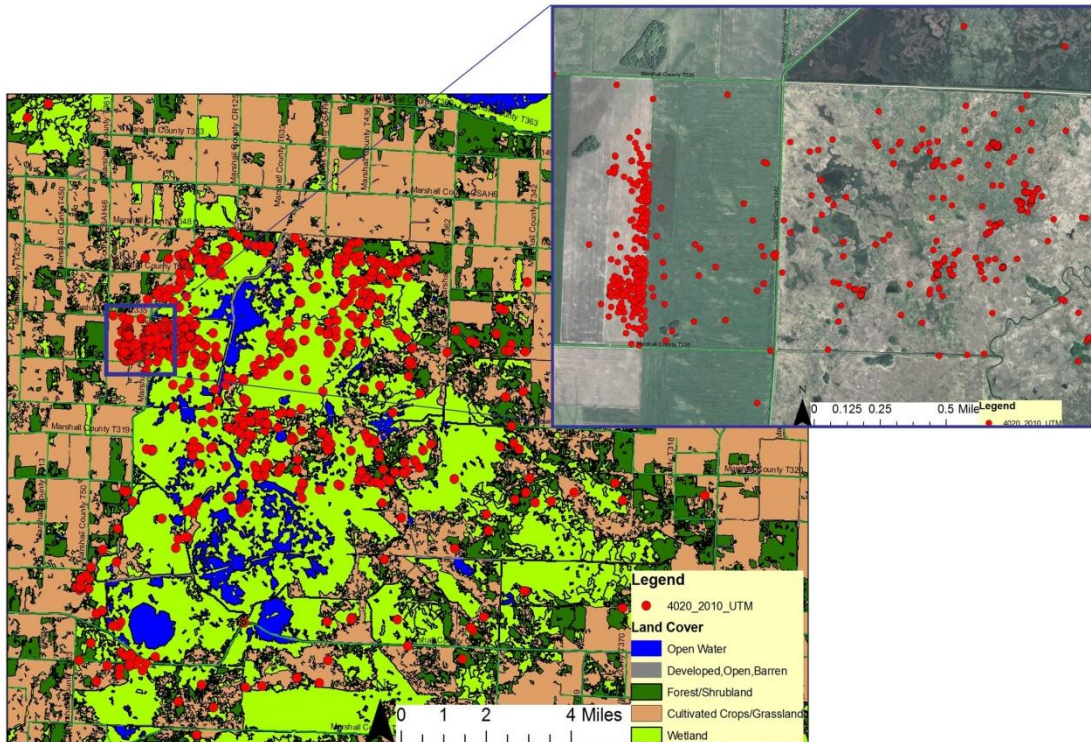


Figure 5. Locations of a GPS-collared male black bear in and near Agassiz NWR (most of the light-green colored area is inside the refuge) during 2010. The enlarged block (satellite photo) shows an area of concentrated use in a sunflower field during August–September, before he denned in central Agassiz in early October.

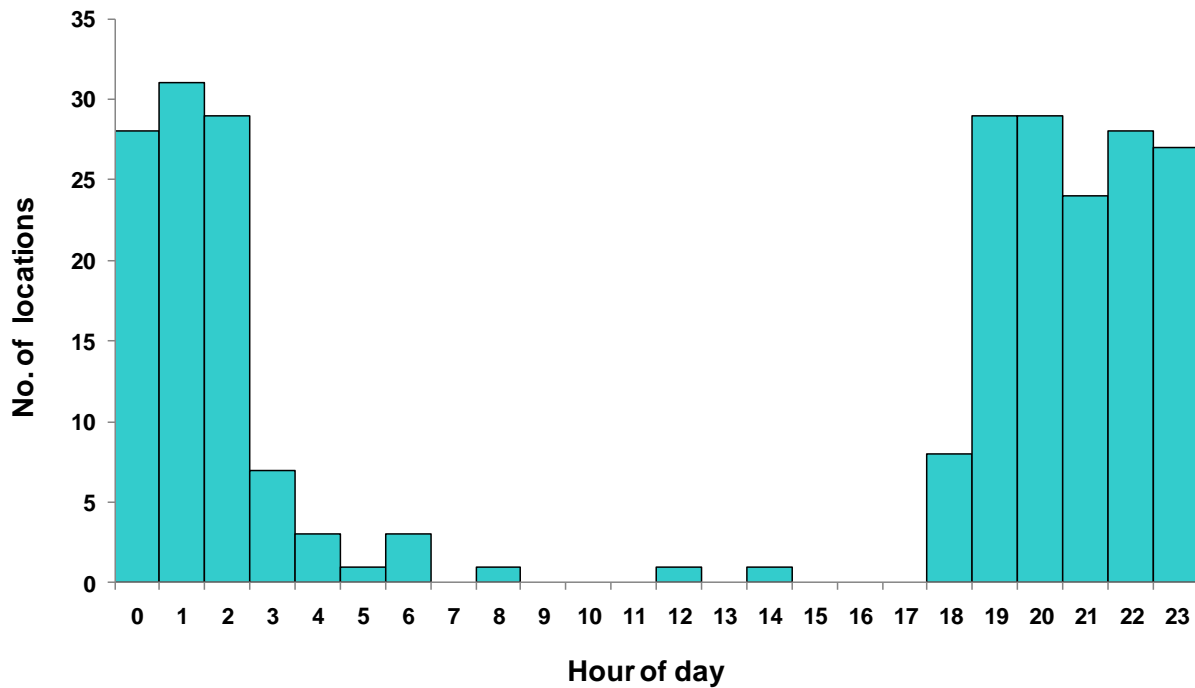


Figure 6. Hourly use of sunflower field by the male black bear shown in Figure 5.

MOOSE POPULATION DYNAMICS IN NORTHEASTERN MINNESOTA

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SUMMARY OF FINDINGS

We captured and radiocollared a total of 150 adult moose (55 adult males and 95 adult females) between 2002 and 2008. As of 1 April 2011, 114 collared moose (50 adult males and 64 adult females) have died. Annual mortality rates varied among years, and generally were higher than found elsewhere in North America. Estimates of fertility for this population were also low compared with other North American moose populations. Data analyses from this research are progressing and 3 manuscripts are published, 1 manuscript is in press, and 2 other manuscripts have been submitted to peer-reviewed journals.

INTRODUCTION

Moose (*Alces alces*) formerly occurred throughout much of the forested zone of northern Minnesota. Today they are restricted to the northeastern-most counties, including all of Lake and Cook counties, and most of northern St. Louis County. We initiated a research project in 2002 to better understand the dynamics of this population. Fieldwork, including aerial mortality checks and necropsy of dead animals, continued through 1 April 2011. We are in the process of analyzing data and preparing manuscripts. The following report will discuss preliminary findings.

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

METHODS

We captured a total of 150 moose in southern Lake County and southwestern Cook County between 2002 and 2008, attached radiocollars, and collected blood, hair, fecal and tooth samples. See Lenarz et al. (2009) for greater detail on the study area and research methods. We monitored a sample of up to 78 radiocollared moose weekly to determine when mortality occurred. We calculated annual non-hunting mortality rates ($1 - \text{survival}$) using the Kaplan-Meier procedure (Kaplan and Meier 1958) modified for staggered-entry (Pollock et al. 1989) and censored all moose killed by hunters, that died from capture mortality, that had emigrated from the study area, or experienced apparent transmitter failure. We used a Cox Proportional Hazard (CPH) model (Cox 1972, SAS PROC PHREG, SAS Institute 2008) to test for a difference in annual survival between sexes. Beginning in 2004, we used helicopter surveys in late May–early June (MJ) to estimate fertility of radiocollared females and a survey the following year in late April–early May (AM) to estimate survival of calves born during the previous spring.

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RESULTS AND DISCUSSION

As of 1 April 2011, 114 collared moose (50 adult males and 64 adult females) have died. In addition, 1 moose slipped its collar, 1 moose moved out of the study area, and we lost contact (apparent transmitter failure) with 2 moose (1 of the moose with a dead collar was recaptured in January 2011 as part of a new research project). Moose that died within 2 weeks of capture (6) were designated as capture mortality. Hunters killed 17 moose, 2 were poached, and 12 were killed in collisions with vehicles (cars, trucks, or trains). The remaining mortality (77) was considered to be non-anthropogenic, and causes included wolf predation (10), bacterial meningitis (1), and unknown (66).

The unknown mortality appeared to be largely non-traumatic. In 51% of the cases, the intact carcass was found with only minor scavenging by small mammals or birds. Wolves (*Canis lupus*) and black bears (*Ursus americanus*) were the primary scavengers in 34% of the cases. We were unwilling to attribute predation as the cause of death in these cases, because there was little evidence that a struggle had preceded death. In 15% of the cases, we were unable to examine the carcasses or only found a collar with tooth-marks.

Annual non-hunting mortality rates (1 June–31 May) for adult moose averaged 20% for males (SE = 5, 0–40%, $n = 8$) and 21% for females (SE = 3, 5–30%, $n = 8$; Table 1). Sex did not contribute to the prediction of survival ($\chi^2 = 0.001$, $P = 0.98$), which implies that there was no difference in survival rates (non-hunting) between adult male and female moose. Non-hunting mortality was substantially higher than documented for populations outside of Minnesota (generally 8 to 12%; Peterson 1977, Mytton and Keith 1981, Bangs 1989, Larsen et al. 1989, Ballard 1991, Kufeld and Bowden 1996, Bertram and Vivion 2002,) and similar to that observed for adult moose in northwestern Minnesota (21%; Murray et al. 2006).

Serum samples from 91 radiocollared adult female moose were collected between 2002 and 2008 and analyzed by radioimmunoassay for concentrations of progesterone. Using a pregnancy threshold of 2.0 ng/ml progesterone, annual pregnancy rate varied from 55 to 100% ($\bar{x} = 80\%$, SE = 8, $n = 5$). Boer (1992), in his review of moose reproduction in North America found that adult pregnancy rates across North America averaged 84%. Although the pregnancy rates of yearlings tend to be lower than for adult moose (Schwartz 1997), our sample included only 1 yearling. Our estimates may be biased low, because 4 cows that tested negative in 2003 (55% pregnancy rate) were subsequently observed with a calf.

Between 2004 and 2010, 222 radiocollared adult females gave birth to a minimum of 196 calves (115 singles, 39 twins, and 1 set of triplets; M. W. Schrage, Fond du Lac Resources Management Division, unpublished data). The annual ratio of calves:radiocollared females ranged from 0.53 to 1.13 ($\bar{x} = 0.87$, SE = 0.07, $n = 7$). These estimates were biased low, because in 6 of 7 years, radiocollared females not observed with calves during the late MJ survey were subsequently observed to be accompanied by a single calf ($\bar{x} = 3$, SE = 0.6, $n = 7$). It is also possible that post-natal mortality occurred prior to the MJ survey. Nonetheless, these estimates are low compared with other locations in North America. Boer (1992), for example, reported estimates ranging from 0.88 to 1.24 calves/adult female in moose populations above and below K-carrying capacity, respectively.

During the past 3 years, 6 manuscripts discussing the results of this research have been prepared for publication. Most recently, a paper entitled “Spending degrees of freedom in a poor economy: a case study of building a sightability model for moose in northeastern Minnesota” was accepted for publication in the *Journal of Wildlife Management*. Two additional manuscripts evaluating habitat utilization by northeastern moose have been submitted to the journal *Alces*. Finally, data from our research served as the basis for a chapter in a Master of Science thesis on characteristics of post-partum areas for moose in northeastern Minnesota.

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Table 1. Annual adult mortality of moose in northeastern Minnesota, USA, 2002–2009. Estimates censored for hunting, capture mortality, and apparent transmitter failure.

Year ¹	Male	Female	Combined
2002	7% (25) ²	30% (29)	23% (54)
2003	25% (21)	20% (34)	21% (55)
2004	8% (32)	5% (42)	6% (74)
2005	24% (21)	29% (30)	26% (51)
2006	40% (10)	27% (22)	31% (32)
2007	20% (8)	19% (49)	18% (57)
2008	0% (7)	21% (38)	16% (45)
2009	33% (4)	13% (33)	16% (37)
Mean	18%	21%	20%

¹ Period: 1 June–31 May.

² Sample size as of 31 May.

SURVIVAL AND CAUSES OF MORTALITY FOR FISHER AND MARTEN IN MINNESOTA

John Erb, Barry Sampson, and Pam Coy

SUMMARY OF FINDINGS

As part of a larger project on *Martes* ecology in Minnesota, we began monitoring survival of radiocollared fishers (*Martes pennanti*) and martens (*Martes americana*) during winter 2007–2008. Including the pilot year of the study, a total of 128 martens (58 females, 70 males) and 65 fishers (36 females, 29 males) have been radiocollared. An additional 6 animals (3 martens, 3 fishers) were ear-tagged only. Of the 128 martens radiocollared, 51 are still actively monitored (18 females, 33 males), radio-contact has been lost on 23 (8 slipped collars, 15 missing), and 54 deaths have occurred. Of the 54 known marten deaths (26 females, 28 males), most have been from regulated fur trapping ($n = 16$; 13 males, 3 females) and predation ($n = 29$; 19 females, 10 males). Of the 29 predation events, 21 marten were killed by mammalian predators, 7 by raptors, and 1 by an unknown predator. While predation mortality of marten has occurred in most seasons, the majority has occurred during late winter and spring. While total marten mortality has not been noticeably sex-biased, predation mortality has been very female-biased (~2 females:1 males), while harvest mortality of marten is significantly male-biased (~4 males:1 females). The combination of male-biased harvest mortality and female-biased non-harvest mortality may produce offsetting effects on the population sex ratio. Of the 65 fishers radiocollared, 24 are still being monitored (14 females, 10 males), radio-contact was lost on 17 (12 belting hardware failures, 4 missing, 1 collar removed), and 24 deaths (12 females, 12 males) have occurred (12 [8 females, 4 males] were killed by other predators [scavenging by an eagle can't be ruled out in 1 case], 4 [1 female, 3 males] died from unknown but apparently natural causes, 4 [1 female, 3 males] were legally trapped, 2 [1 male, 1 female] were struck by vehicles (both while apparently dispersing in the fall), 1 male was accidentally trapped out of season, and 1 female was illegally trapped). Although sample size is small, 10 of the 12 predation deaths of fishers took place from late winter through spring. Seven of the 8 female fisher predation mortalities were attributed to other mammalian carnivores, while 3 of the 4 male fisher predation mortalities were attributed to raptors (all bald eagles [*Haliaeetus leucocephalus*]). Of greatest significance, 7 of the 8 female fishers killed by predators were adults, and 5 of the 7 were killed while they still had dependent young in natal dens, indirectly resulting in the death of their 14 kits. The deaths of these 5 kit-rearing females represent 36% 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 dominant source of mortality.

INTRODUCTION

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

transition zone, including southeast Minnesota (Swanson et al. 1945, Balser and Longley 1966). The impacts of unregulated harvest and habitat alteration were equally as detrimental to fisher, with populations substantially reduced by the 1930s.

Legally, fisher and marten were unprotected in Minnesota prior to 1917, after which harvest season length restrictions were implemented. These protections were removed in the mid-1920s, and remained so until all harvest was prohibited in 1929. Seasons remained closed until 1977 for fisher and 1985 for marten, when limited harvests were reinstated. 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 the past 4 years were reduced from 16 days to 9 days, and in 2010, the fisher harvest limit was reduced from 5 to 2. During this 4-year period, harvests have averaged ~2,000 and 1,400 for marten and fisher, respectively.

While both species appear to have naturally re-colonized a significant portion of their historic range, Minnesota-specific information on survival and causes of mortality is limited. Except for harvest data, we are aware of only 1 published field study in Minnesota. Specifically, Mech and Rogers (1977) opportunistically 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 (including 4% poaching mortality on males) for adult male and female fisher, respectively, in Massachusetts. Excluding the first 4-5 months of life, juvenile non-harvest mortality rates have been estimated to be 28% in Maine (Krohn et al. 1994), and 0% (females) and 23% (males) in Massachusetts (York 1996). While mortality may be higher in the first months of life than the rest of the year, if we assume a similar non-harvest mortality rate during the first 4-5 months of life, we calculate that annual non-harvest mortality for juveniles would be ~56% in Maine. Combining minimum summer survival estimates for kits with telemetry estimates of survival the rest of the year, York (1996) estimated ~22% (females) and 67% (males) annual non-harvest mortality for juveniles in Massachusetts. Kelly (1977, in Paragi et al. 1994) reportedly estimated 18% annual mortality of juveniles and 44% annual mortality for adult fisher in New Hampshire. More recently, Koen et al. (2007) estimated annual mortality rate (including harvest mortality) of fishers in Ontario to be 55–67% for males, and 29–37% for females. While non-harvest mortality of adult fishers is often presumed to be 'low,' it has not always proven to be the case. Furthermore, there is limited data on which to assess the amount of geographic or temporal variation in non-harvest mortality of fisher.

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 1997, 2009), Quebec (Potvin and Breton 1997), and Newfoundland (Fredrickson 1990). While we do not summarize details of these studies here, a couple of conclusions are worthwhile. First, when comparing across studies, annual adult non-harvest mortality rates varied from ~0.07 to 0.48. Juvenile data were rarely separated, but a few studies pooled ages, and mortality rates also fell within the above interval. While this variability may be attributable to both sampling and biological variability, the wide range suggests that it is risky to assume results from any area are applicable elsewhere. Secondly, at least 1 study (Maine; Hodgman et al. 1997) has documented significantly higher natural mortality for females compared to males, and others researchers have postulated this to

be common given the typical male-biased harvest, 50:50 sex ratio at birth, and often balanced adult sex ratio (Strickland et al. 1982, Strickland and Douglas 1987). Due to male-biased harvest and our *assumed* sex-related equality in non-harvest mortality, our marten population model 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 and statistically-oriented analysis until a later time.

STUDY AREA

Marten research is focused on 1 study area located in northeastern Minnesota (Figure 1; Area 1), though an occasional marten is captured and radiocollared in Area 2. Area 1 (~700 km²) includes 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 (1,075 km²), our primary fisher study area, includes 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. 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.

We physically immobilized captured animals using metal 'combs' to restrict them to a small portion of the trap, or we restrained the animal against the side of the trap by pulling its tail through the cage mesh. We injected animals 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, we reversed the xylazine 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, Kentucky) 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, Kentucky) 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, tail, and hind foot lengths, and chest, neck, and head circumferences. We removed guard hair samples for possible genotyping, and for evaluating the use of stable isotope analysis for deciphering food habits (Ben-David et al. 1997). To determine which females were pregnant in mid-winter, and eventually the percent of those that actually produce a litter in spring, we attempted to draw blood samples to measure serum progesterone concentrations (Frost et al. 1997). Antibiotics were administered subcutaneously to all animals prior to release.

During the pilot year, we deployed several radiocollar designs on fisher, including an ATS M1585 zip-tie collar (~43 g), an ATS M1930 collar (~38 g), and a Lotec SMRC-3 collar (~61 g; deployed on adult males only). Since the pilot year, we have primarily deployed ATS M1940 (~43 g) or Sirtrack TVC-162 collars (~45 g) on fisher. The majority of martens 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 128 martens (58 females, 70 males) and 63 fishers (36 females, 27 males) have been radiocollared. An additional 6 animals (3 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.

Of the 128 martens radiocollared, 51 are actively being monitored (18 females, 33 males), radio-contact has been lost on 23 (8 slipped collars, 15 missing), and 54 deaths have occurred. Of the 54 known marten deaths (26 females, 28 males), most have been from regulated fur-trapping ($n = 16$; 13 males, 3 females) and predation ($n = 29$; 19 females, 10 males). Four animals 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 1 unknown natural cause. Four martens also died from capture/collar related complications. The status of one additional animal is unknown at this time, pending retrieval of the collar/animal from an underground location.

Of the 29 predation events, 21 marten were killed by mammalian predators, 7 by raptors, and 1 by an unknown predator. After censoring 5 individuals whose deaths occurred with 14 days post-capture, 75% of the predation mortalities occurred in late-winter through spring (i.e., February–May; Figure 2). Forensic (DNA) analysis of samples collected from predated marten (mammalian predation only) is incomplete. To date, DNA analysis has confirmed bobcat (*Lynx rufus*) predation in all 4 cases for which analysis is complete. Felids (bobcat or lynx [*Lynx canadensis*]) are the likely predator in 2 additional cases for which partial information (inconclusive DNA or obvious field sign) is available. Remaining forensic analysis is pending, and field evidence suggests fox (*Vulpes vulpes*) or fisher may be responsible for at least a couple predation deaths on marten.

While total marten mortality has *not* been noticeably sex-biased (52% male), and is similar to the sex ratio of the radiocollared sample (55% males), predation mortality has been female-biased (~2:1), with males comprising only 30% of the predation deaths. Conversely, harvest mortality of martens has been significantly male-biased (~4:1), comprising 81% of the harvest mortalities. Considering the timing of mortality in relation to the apparent start of the biological year for marten (~3rd week of April), current data suggests that harvest mortality of males may be *comparatively* additive (i.e., little natural mortality occurs on males post-harvest), whereas harvest mortality on females may be *comparatively* compensatory (i.e., a large percentage of natural mortality occurs on females post-harvest). If the overall population sex ratio for martens is reasonably balanced (which our initial data suggest), the subsequent sex-biases we have observed in number of predated (female-biased) and harvested (male-biased) marten suggest differential vulnerability, not differential abundance. Regardless, the combination of male-biased harvest mortality and female-biased non-harvest mortality may produce offsetting effects on the population sex ratio.

Of the 65 fishers radiocollared (36 females, 29 males), 24 are still being monitored (14 females, 10 males), radio-contact has been lost with 17 animals (12 shed their collars due to belting design failures, 4 are missing, and 1 collar was removed due to neck abrasion), and 24 deaths have been confirmed (12 females, 12 males). In addition, 3 juvenile males were ear-tagged only. Of the 24 known deaths (12 females, 12 males), 12 (8 females, 4 males) were killed by other predators (scavenging by an eagle cannot be ruled out in 1 case), 4 (1 females, 3 males) died from unknown but apparently natural causes, 4 were legally trapped (1 females, 3 males), 2 (1 male, 1 female) were struck by vehicles (both while apparently dispersing in the fall), 1 male was accidentally trapped out of season, and 1 female was illegally trapped.

Although sample size is small, 83% (10 of 12) of the predation deaths of fishers took place in late winter and spring (i.e., February–May; Figure 3), similar to the pattern observed with marten. Three of the four male fisher predation deaths were attributed to bald eagles, although we cannot rule out scavenging in 1 case (only the radiocollar was retrieved directly underneath an active eagle nest). The fourth male fisher predated appears to have been killed by a mammalian predator, but full necropsy has not yet occurred. Conversely, only 1 of the 8 female predation deaths was attributed to a raptor (great-horned owl [*Bubo virginianus*]) 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 (fisher was cached) strongly indicates bobcat or lynx in another.

Of greatest significance, 7 of the 8 female fishers killed by other predators were adults, and 5 of those 7 were killed while they still had dependent kits in natal or maternal dens, indirectly resulting in the death of 14 kits. The deaths of these 5 kit-rearing females represent 36% 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. In addition, the need to locate suitable (and multiple) natal/maternal dens likely increases movement. Preliminary data from temperature data-loggers attached to radiocollars suggest 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 concealment cover is diminished (i.e., before ‘green-up’) and 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 comparatively short- (e.g., periodic fluctuations in prey) or long-term (e.g., suspected decline in natal and maternal den availability) changes affecting fisher energetics/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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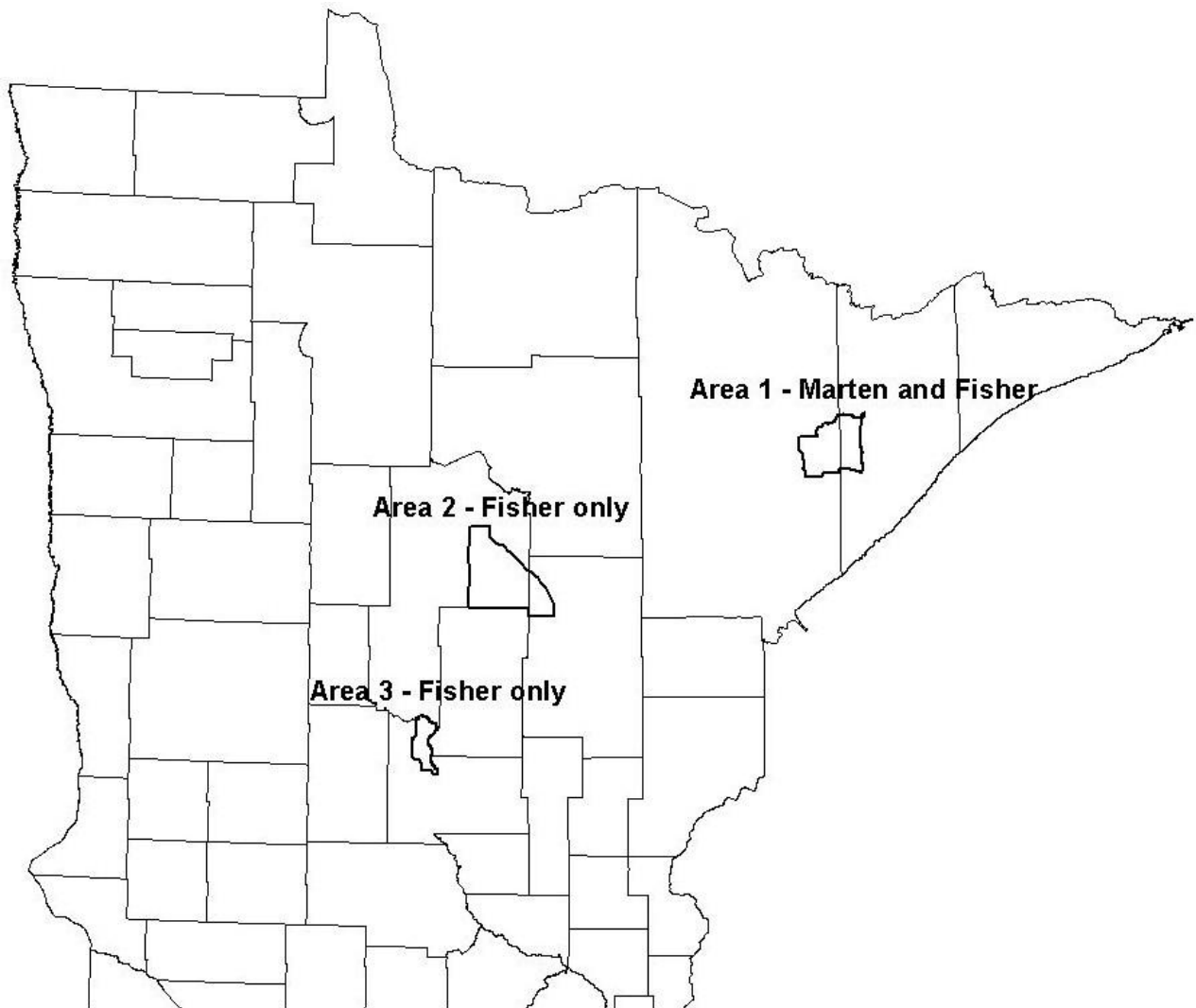


Figure 1. Fisher and marten study areas in Minnesota 2007–2010.

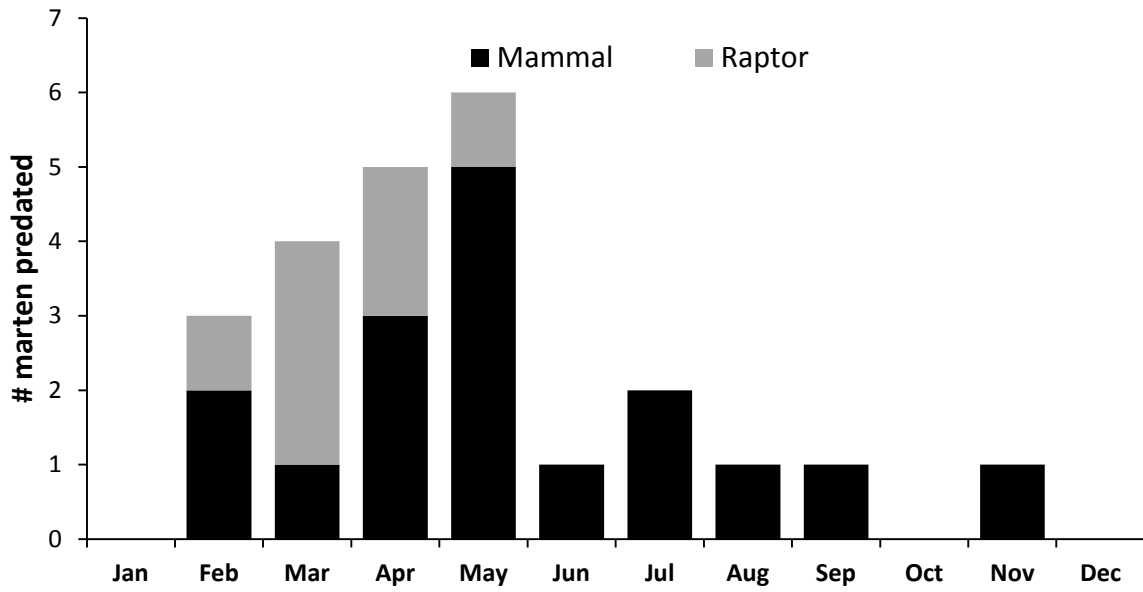


Figure 2. Seasonal timing of marten deaths attributable to predation in northeastern Minnesota, 2007–2010.

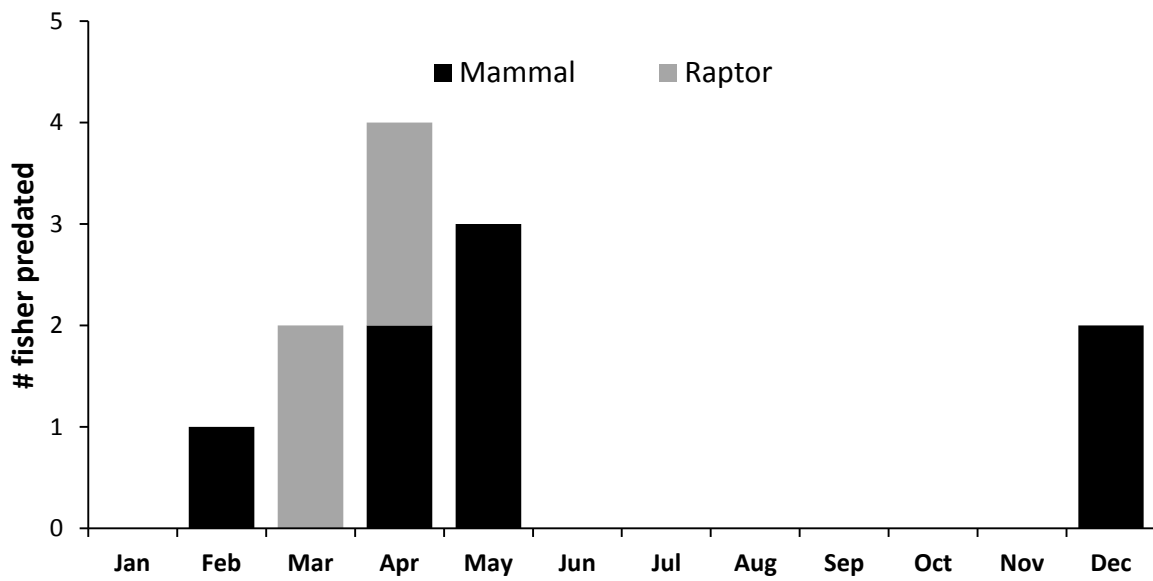


Figure 3. Seasonal timing of fisher deaths attributable to predation in north-central Minnesota, 2007–2010.

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, we have captured 128 martens (58 females, 70 males) and 65 fishers (36 females, 29 males). To date, age and reproductive status have been confirmed on 12 adult (≥ 2 years old) female martens, 83% of which produced litters. Of the 10 that produced litters, we have obtained litter counts for 8 (average minimum litter size = 3.4). Of the 15 natal or maternal dens identified, 53% have been in underground burrows, commonly in rock-laden soils, while 47% have been in elevated tree cavities (primarily cedar trees). We have also confirmed litters for 21 adult (≥ 2 years old) female fishers, all but 1 for which we have confirmed litter size (average litter = 2.7). Excluding 2 potential juveniles, 83% of adult (≥ 2 years old) female fishers produced litters, though initial data suggests that pregnancy rate and average litter size is smaller for 2 year old fishers compared to older adults. All of the fisher natal or maternal dens we have located prior to June 1 ($n = 23$) have been in elevated cavities of large diameter (average diameter at breast height [dbh] = 20.6) live trees or snags, predominantly in aspen (75%) and oak (17%). The only fisher maternal den located after June 1 was in a hollow log on the ground. Fisher kits appear to be born during the last 2 weeks of March, while marten parturition appears to be centered on the last 2 weeks of April. Both species appear to move their kits from the natal den to 1 or more different maternal dens in the first 6 weeks following birth.

INTRODUCTION

American marten and fisher are native to Minnesota, but reliable documentation of their historic distribution is limited. Undoubtedly, northeastern Minnesota was a stronghold for the marten population, though notable numbers likely occurred in the northern border areas as far west as Roseau County. Limited information suggests they occurred as far south as Crow Wing County and as far southwest as Polk County. As a result of unregulated harvest, martens were considered rare in Minnesota by 1900, and extensive logging and burning around the turn of the century further contributed to the near extirpation of martens from Minnesota by the 1930s (Swanson et al. 1945). Fishers in Minnesota appear to have historically occupied a larger geographic area than martens, extending further south and west into the hardwood dominated transition zone, including southeastern Minnesota (Swanson et al. 1945, Balser and Longley 1966). The impacts of unregulated harvest and habitat alteration were equally as detrimental to fishers, 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 half 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 half 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 the past 4 years were reduced from 16 days to 9 days, and in 2010, the fisher harvest

limit was reduced from 5 to 2. During this 4-year period, harvests have averaged ~2,000 and 1,400 for marten and fisher, respectively.

While both species appear to have naturally re-colonized a significant portion of their historic range, Minnesota-specific information on reproductive ecology is limited to carcass (corpora lutea, placental scar) data collected from harvested animals primarily from 1985-90 (Kuehn 1989; Minnesota Department of Natural Resources [MNDNR], unpublished data). Reproductive data are 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 are 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 (Payne 1982, Gilbert 1987, 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 1997, 2009; Aune and Schladweiler 1997; MNDNR, 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 (PS); review in Powell 1993).

Of greater concern is the possibility that ovary, and to lesser degree uterine, analyses might consistently overestimate parturition rate, thereby 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; Crowley et al. 1990; Paragi 1990; MNDNR, unpublished data), while more 'direct' estimates of average parturition rate from radio-marked animals have been lower (46–75%; Crowley et al. 1990, Paragi 1990, Arthur and Krohn 1991, Paragi et al. 1994, York 1996, Truex et al. 1998, Higley and Mathews 2009), and are often highly variable. Conversely, Kuehn (1989) did not detect changes in pregnancy rate (from CL analysis) during a 64% decline in snowshoe hare indices in Minnesota.

For marten, several largely ovarian-based estimates of annual pregnancy rate have often been in the range of 80–90% (Archibald and Jessup 1984; Strickland and Douglas 1987;

Aune and Schladweiler 1997; 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 (Strickland and Douglas 1987, Thompson and Colgan 1987, Aune and Schladweiler 1997, 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, 75, 81, and 92% of adult females were lactating for their 3 different study areas (Phillips 1994, Payer 1999), similar to much of the CL-based pregnancy studies.

Understanding reproductive ecology of these species also necessitates gathering information on natal and maternal den structures and selection of den-sites. Natal dens are the structures where kits are born, whereas maternal dens are sites used subsequently by the female with her dependent young. Although data are 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), 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 is collected or additional analysis is completed.

STUDY AREA

Marten research is focused on 1 study area located in northeastern Minnesota (Figure 1; Area 1), though an occasional marten is captured and radio-collared in Area 2 (Figure 1). Area 1 (~700 km²) is composed of approximately 69% mixed coniferous-deciduous forest, 15% lowland conifer or bog, 5% upland coniferous forest, 4% gravel pits and open mines, 3% regenerating forest (deciduous and coniferous), 2% shrubby grassland, 1% marsh and fen, 1% open water, and < 1% deciduous forest. Area 1 is 90% public ownership, including portions of the Superior National Forest and state and county lands. Fishers are also present in this area at low to moderate density.

Fisher research will take place in 3 areas (Figure 1; Areas 1, 2, and 3). The work in Area 3 is a collaborative effort between Camp Ripley Military Reservation, Central Lakes Community College, and the MNDNR. 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 (1,075 km²), our primary fisher study area, is composed of 74% deciduous forest, 11% open water, 5% lowland conifer or bog, 5% marsh and fen, 2% regenerating forest (deciduous and coniferous),

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

METHODS

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

To immobilize animals, we used metal 'combs' to restrict the animal to a small portion of the trap, or restrained the animal against the side of the trap by pulling its tail through the cage mesh. Animals were injected with a hand-syringe using a 10:1 mixture of ketamine and xylazine (fisher: 30 mg/kg ketamine and 3 mg/kg xylazine; marten: 20 mg/kg ketamine, 2 mg/kg xylazine) (Kreeger et al. 2002). After processing, the xylazine was reversed with yohimbine at a dosage of 0.1 mg/kg (marten) or 0.15 mg/kg (fisher). Fisher were either ear-tagged with a monel # 3 tag in one ear (National Band and Tag Co., Newport, Kentucky) and a 2-piece plastic mini-tag (Dalton I.D. Systems, United Kingdom) 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, Kentucky) 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, tail, and hind foot lengths, and chest, neck, and head circumferences. 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 concentrations (Frost et al. 1997). All blood samples were sent to the University of Minnesota Veterinary Diagnostics Lab for progesterone analysis. Antibiotics were administered subcutaneously to all animals prior to release.

During the pilot year, we deployed several radiocollar designs on fisher, including an ATS M1585 zip-tie collar (~43 g), an ATS M1930 collar (~38 g), and a Lotec SMRC-3 collar (~61 g; deployed on adult males only). Since the pilot year, we have primarily deployed ATS M1940 (~43 g) or Sirtrack TVC-162 collars (~45 g) on fisher. The majority of martens have been fitted with Holohil MI-2 collars (~31 g). We retrofitted each collar with a temperature data logger 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, Wisconsin) 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, although 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). 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 1 June) 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 30°, 150°, and 270°. Deviating from FIA protocol, we also establish 3 (not 1, as with FIA) 0.003-acre (6.8-ft radius) circular micro-plots for estimating sapling density, each micro-plot situated at the end of the 3 coarse woody debris sampling transects. Details of vegetation sampling methods within each subplot will be outlined in subsequent years as results become available. Herein, we simply note that we will collect quantitative data on: 1) mean DBH and basal area of live trees, overall and by species; 2) percent 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 a transect and repeated once within each 10-day interval (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 ($n \approx 125$ in each study area) consist of 10 sampling locations (2 parallel lines of 5 stations) spaced 20 m 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 30 m off the trail. In spring, we count snowshoe hare (*Lepus americanus*) pellets in a 1- m^2 plot at each sampling station (McCann et al. 2008). During fall, small mammal snap-trapping will occur for 2 consecutive days at the same sampling stations, similar to protocol used on an existing small mammal survey in Minnesota (Aarhus-Ward 2009). During both spring (hare pellet

sampling) and fall (small mammal trapping), we also will count the number of red squirrels (*Tamiasciurus hudsonicus*) observed or heard along each transect. Rather than using 10-min point counts (e.g., Mattson and Reinhart 1996, Bayne and Hobson 2000) with our small mammal/hare pellet stations as the sampling points, we will simply record the number of unique squirrels observed/heard along each transect while checking pellet plots and small mammal traps. Information on white-tailed deer (*Odocoileus virginianus*) 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 128 martens (58 females, 70 males) and 65 fishers (36 females, 29 males) have been captured. Herein we provide a basic summary of data collected to date on denning chronology, den structures, pregnancy status, and litter size. Because tooth aging has not yet been completed for all animals, and some yet-to-be-aged females may be only 1 year of age (i.e., not capable of producing kits), we present results only for animals known to be ≥ 2 years of age during spring den visits, or those of unknown age, but for which we have confirmed parturition at the time of this writing (i.e., until age is known, we do not include animals that we have confirmed to be nulliparous). No reproductive data are yet available for spring 2011.

Treating females that were alive during multiple parturition periods as independent units, and excluding females known to be 1 year of age during the parturition period, a total of 17 female martens have been available for monitoring during the kit-rearing season. However, we have confirmed age and reproductive status for only 12 female martens (Table 1). Three additional females for which we do not yet have age data were confirmed to be nulliparous, and we were unable to confirm birth status for 2 adult females in 2009. Because it has been more difficult to inspect marten natal dens with video equipment, we have had to rely more on remote cameras to obtain litter information when kits are moved by the female, or when they are older and more mobile. Hence, many estimates of marten litter size are reported as minimums. Acknowledging this, average size of 8 litters confirmed to date is 3.4 (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 concentrations have not sufficiently elevated in pregnant animals at that time to allow us to confirm mid-winter pregnancy status.

A total of 15 marten natal or maternal dens have been located to date (Table 2). Based on 11 dens confirmed prior to June 1 of each year, 64% have been in tree cavities, while 36% have been in underground tunnels (Table 2). We have confirmed only 4 maternal dens used after June 1, and all 4 were in underground burrows situated in rock-laden soils (Table 2). Most female martens 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 as independent units. Excluding individuals known to be 1 year of age during the parturition period, a total of 31 female fishers have been available for monitoring during the kit-rearing season. At the time of this writing, we have confirmed both age and reproductive status for 20 female fishers and reproductive status (but not yet age) from 8 additional females (Table 3). In addition, we have confirmed reproductive status for 3 females for which there is currently some uncertainty or discrepancy in age data collected. Pooling all female fishers that produced a litter ($n = 20$), average litter size is 2.7 (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 older females (~2.4 versus 2.9). There also is some indication that birth rates are lower for 2-year-olds compared to older females. Parturition rate for 2 year olds is between 50 and 71% depending on the age assigned to 3 females with uncertain age assignment. Parturition rate for female fishers ≥ 3 years of age is 85–86%. However, the only 2 apparently 'failed' reproductive events detected for adults ≥ 3 years of age were from the same 7+ year old female in 2009 and 2010.

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 appears that the fisher progesterone concentrations are sufficiently elevated in pregnant females at the time of our winter capture operations (i.e., mid-December 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.

A total of 29 fisher natal or maternal dens have been confirmed. With 1 exception (hollow base of a live oak tree), all natal/maternal dens located prior to 1 June of each year (n = 28) have been in elevated tree cavities. Only 1 maternal den has been located after June 1, in a hollow log on the ground (15.7"-diameter sugar maple). Of the 28 natal/maternal dens located prior to June 1, detailed measurements have been completed on only 24. Cavities have been located in both live trees and snags (overall average dbh = 20.6 inches), of which 44% were in live aspen (average dbh = 21.5 inches), 31% in aspen snags (average dbh = 19.9 inches), 17% in live oaks (average dbh = 19.5 inches), and 1 each in a pine snag (dbh = 21.9 inches) and live cedar (dbh = 20.3 inches) (Table 4). While monitoring has not been standardized across animals, many female fishers appear to move kits from their natal den to at least 2 different maternal dens prior to 1 June.

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Table 1. Parturition status and litter size for radiocollared female marten in Minnesota¹.

ID	Year	Age	Litter	Litter size
M09-280	2010	2	Yes	≥3
M09-262	2009	2	Yes	
M09-254	2010	2	Yes	≥3
M09-264	2009	3	No	
M10-290	2010	3	No?	
M09-262	2010	3 or 4	Yes	4
M09-247	2009	5	Yes	4
M08-140	2008	9	Yes	
M09-286	2009	9	Yes	≥3
M08-140	2009	10	Yes	≥2
M09-286	2010	10	Yes	≥4
M09-237	2010	11	Yes	4

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

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

ID	Year	Date confirmed	Den type	Den structure	Den details
M09-254	2010	4/19	natal	tree cavity	15.9" dbh live red maple
M09-237	2010	4/19	natal	tree cavity	16.8" dbh live tamarack
M08-140	2009	4/21	natal	underground burrow	rock-laden soil
M09-280	2010	4/28	natal	underground burrow	rock-laden soil
M08-140	2008	4/30	natal	underground burrow	rock-laden soil
M09-286	2010	5/7	natal	tree cavity	21.5" dbh live cedar
M09-262	2010	5/10	natal	tree cavity	18.8" dbh live cedar
M09-286	2009	5/19	natal	tree cavity	16.1" dbh live cedar
M09-286	2010	5/19	maternal	tree cavity	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 radiocollared female fishers in Minnesota[†].

ID	Year	Age	Litter	Litter size
F10-501	2010	1 or 2	No	
F09-362	2009	1 or 2	No	
F08-375	2008	2	Yes	≥2
F09-360	2009	2	Yes	2
F08-304	2009	2	Yes	2
F08-077	2009	2	Yes	4
F09-362	2009	2	No	
F09-364	2009	2	No	
F10-501	2011	2 or 3	Yes	2
F09-394	2009	3	Yes	3
F08-375	2009	3	Yes	3
F08-353	2009	3	Yes	3
F10-503	2011	3	Yes	2
F09-380	2009	4	Yes	3
F09-394	2010	4	Yes	2
F10-507	2011	4	Yes	3
F08-353	2010	4	Yes	3
F09-394	2011	5	Yes	3
F09-354	2009	7	No?	
F09-354	2010	8	No?	
F09-370	2009	11	Yes	3
F10-328	2010		Yes	2
F09-461	2010		Yes	3
F10-507	2010		Yes	3
F11-316	2011		Yes	1
F11-340	2011		Yes	3

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

Table 4. Natal and maternal den structures used by radiocollared female fishers in Minnesota.

ID	Year	Date confirmed	Den type	Den structure	Den details
F10-507	2011	3/9	Natal	Tree cavity	16.9" dbh aspen snag
F08-353	2010	3/24	Natal	Tree cavity	15.1" dbh live aspen
F10-507	2010	3/26	Natal	Tree cavity	25.6" dbh live oak
F09-394	2010	3/26	Natal	Tree cavity	24.9" dbh live aspen
F09-394	2011	3/28	Natal	Tree cavity	15.8" dbh live aspen
F10-501	2011	3/30	Natal	Tree cavity	20.3" dbh live cedar
F08-375	2009	4/7	Natal	Tree cavity	21.9" dbh w. pine snag
F09-360	2009	4/8	Natal	Tree cavity	15.3" dbh aspen snag
F08-353	2009	4/8	Natal	Tree cavity	23.2" dbh live aspen
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
F10-507	2011	4/12	Maternal	Tree cavity	15.1" dbh live oak
F10-507	2010	4/13	Maternal	Tree cavity	22.1" dbh aspen snag
F09-380	2009	4/14	Natal	Tree cavity	23.6" dbh aspen snag
F09-370	2009	4/15	Natal	Tree cavity	23.5" dbh aspen snag
F09-394	2009	4/18	Natal	Tree cavity	21.5" dbh live aspen
F09-394	2010	4/20	Maternal	Tree cavity	26.1" dbh live aspen
F08-353	2010	4/22	Maternal	Tree cavity	24.3" dbh aspen snag
F09-394	2011	5/4	Maternal	Tree cavity	19.8" dbh live aspen
F09-461	2010	5/18	Maternal	Tree cavity	22.3" dbh live aspen
F09-360	2009	5/29	Maternal	Hollow tree base	19.1" dbh live oak
F08-375	2008	6/25	Maternal	Hollow log	15.7" diam. sugar maple

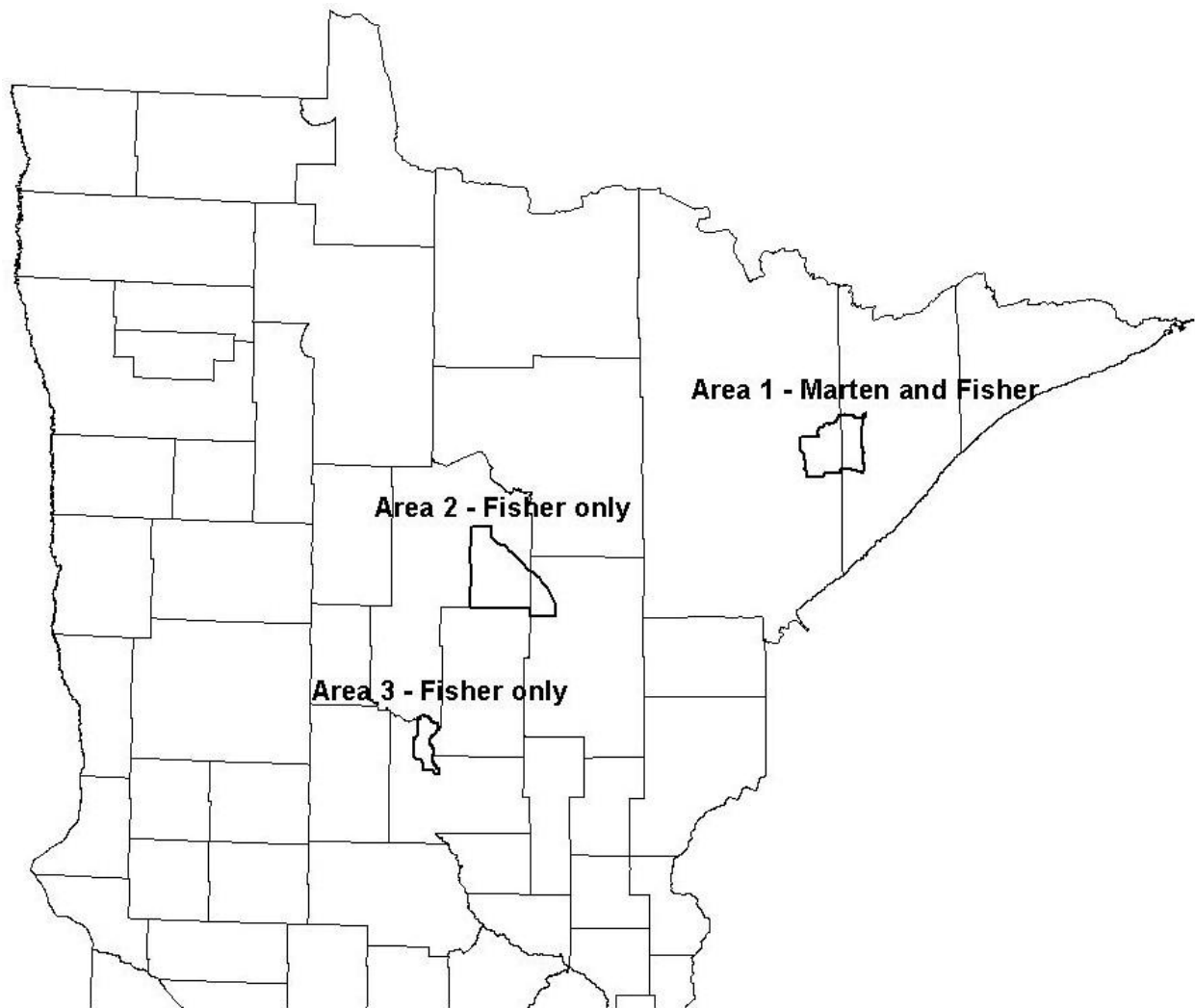


Figure 1. Fisher and marten study areas in Minnesota, 2008–2010.

HABITAT SELECTION BY MALE RUFFED GROUSE AT MULTIPLE SPATIAL SCALES

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

SUMMARY OF FINDINGS

We collected data about ruffed grouse drumming structures during 2 spring field seasons. During 2009, we located 454 used drumming structures within 200 m of survey transects, and we sampled vegetation characteristics at 434 of them and at 434 nearby unused structures. During 2010, we located 449 individual drumming structures and sampled vegetation characteristics at all of them and at 449 nearby unused structures. Data collection is complete, and we have begun analyzing the data. Research results will be available beginning in fall 2011.

INTRODUCTION

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

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

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

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

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

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

OBJECTIVES

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

STUDY AREA

The study area is in portions of Red Lake Wildlife Management Area and adjacent Beltrami Island State Forest in Roseau, Beltrami, and Lake of the Woods counties in north-central Minnesota. The study area encompassed approximately 251,038 ha and was located in the Laurentian Mixed Forest Province (MNDNR 2003). The study area was divided into 2 sampling units with the northern part (116,454 ha) of the study area dominated by conifer and the southern part (134,584 ha) dominated by aspen. We did not include Red Lake Band Tribal Lands in our study.

METHODS

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

Data Collection

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

sample transect. We randomly selected 10 transects from each of the 3 groups to survey for our study.

Each of the 30 selected transects were surveyed on foot beginning 0.5 hours before sunrise during 8 different mornings during an 8-week period ending on the Friday nearest 31 May. When drumming grouse were detected during a survey, the exact location of each one was determined by approaching it and identifying the log or other structure on which it was standing to drum, often indicated by the presence of fresh droppings. Universal Transverse Mercator (UTM) coordinates were taken using a hand held global positioning system (GPS) unit at drumming structures, and the drumming structure's location was confirmed by approaching during subsequent surveys.

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

We measured characteristics of ruffed grouse habitat at 3 spatial scales. The smallest scale was the area immediately surrounding drumming locations identified during surveys. Characteristics at this scale were measured in the field. The same variables were measured at an unused but potential drumming structure (e.g., log or stump with no signs of use by grouse) nearest a randomly selected point within 85 m of each used drumming structure. A circle with a radius of 85 m represents the "core area" (2.3 ha) of a male's home range during the 2-month "drumming season" (6.7 ha, Archibald 1975). An 85-m radius ensured that selected unused locations were within the home range, whereas the 146-m radius of the home range would not have. This information was collected for all used drumming structures that fell within 200 m of the transect line.

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

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

Data Analysis

New field data—We will conduct a separate analysis at each spatial scale of interest. At the scale of specific drumming locations the analysis will follow a case-control logistic regression design in which the response variable is whether the point was used or not used (Keating and Cherry 2004). This may reveal selection for characteristics of drumming locations, given the constraint of occupying a limited home range. At all larger spatial scales we will use regression analyses in which the response variable is the density of drumming males per hectare. For all analyses we will define *a priori* models consisting of explanatory variables that

represent hypothesized habitat relationships. We will use information-theoretic model selection procedures and consider multimodel inference (e.g., Burnham and Anderson 2002).

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

RESULTS

We collected data about ruffed grouse drumming structures during 2 spring field seasons. During 2009, we located 454 used drumming structures within 200 m of survey transects, and we sampled vegetation characteristics at 434 of them and at 434 nearby unused structures. During 2010, we located 449 individual drumming structures and sampled vegetation characteristics at all of them and at 449 nearby unused structures. Data collection is complete, and we have begun analyzing the data. Research results will be available beginning in fall 2011.

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

Michael A. Larson and J. Wesley Bailey

SUMMARY OF FINDINGS

During July and August of 2010, we attempted to locate spruce grouse (*Falci pennis canadensis canadensis*) hens and broods by surveying likely habitat and broadcasting a chick call. We detected a few spruce grouse, but determined that the survey method would not be successful for studying habitat selection by broods.

The current study focuses on habitat selection by adult spruce grouse during spring. During spring of 2011, we surveyed at least 38 plots centered on stands of jack pine (*Pinus banksiana*) and 18 plots centered on stands of lowland black spruce (*Picea mariana*). We detected 44 spruce grouse in those plots; approximately half were males and half were females. The 2011 field season was not complete when this summary was written, and we plan to continue collecting data during the spring of 2012.

INTRODUCTION

The spruce grouse is listed as a Species of Greatest Conservation Need (SGCN) by the Minnesota Department of Natural Resources (MNDNR), which cited its dependence on a potentially vulnerable habitat type (i.e., boreal forest) and a lack of population trend data (MNDNR 2006). Due to the unknown or tenuous status of spruce grouse along the southern edge of its range and the existence of several threats to the viability of their populations, there is interest in learning more about their status and ecology. The Association of Fish and Wildlife Agencies (Williamson et al. 2008) recommended developing formal surveys for monitoring population change and conducting research on the impacts of habitat change and hunting on spruce grouse.

All 3 of the previous studies of spruce grouse in Minnesota were conducted in rather unique study areas (i.e., either entirely black spruce lowlands or primarily peatlands; Anderson 1973, Haas 1974, Pietz and Tester 1979), so it is difficult to apply their results broadly. They were similar to other habitat studies in focusing on the associations between the presence of spruce grouse and certain forest cover types and traditional metrics of forest structure (e.g., tree density and height). However, these studies did not address some of the important questions that are relevant to how we currently manage forests. For example, we do not know whether the density and species of residual trees (i.e., those left after logging) are important, what size and shape of forest stands are optimum, what proportions of different cover types on a landscape are best, or if spatial juxtaposition of cover types or other habitat characteristics are important. In short, wildlife managers want to know how to manage forests with suitable cover types for the most benefit to spruce grouse.

The habitat needs or preferences of spruce grouse may vary by seasonally (Stenlund and Magnus 1951, Pietz and Tester 1982). However, the only time of year when there is a reasonable chance of detecting them is during spring when males display with a flutter flight (Keppie 1992). Investigating habitat selection during other times of year will require radio-tracking. We attempted to survey hens and broods during summer 2010 using a chick call, but we were unsuccessful, so studying habitat selection by broods also will require radio-tracking. This study will focus on the spring display period, so we can learn about habitat selection with minimal costs using auditory surveys, while also learning basic information about the densities of spruce grouse and the feasibility of encountering a sufficient number of grouse for a potential radio-tracking study in the future.

OBJECTIVES

1. To estimate parameters in regression models relating habitat characteristics at the landscape scale to the density of spruce grouse during spring.
2. To test whether specific measures of the structure of forest stands that were associated with the presence of spruce grouse during previous studies are associated with the presence of spruce grouse during spring in a new data set.
3. To estimate parameters in capture–recapture models of time-of-detection data (Allredge et al. 2007) for estimating the probability of detection and densities of spruce grouse.

STUDY AREA

The study will be conducted in Red Lake WMA and adjacent portions of Beltrami Island State Forest, which are in Lake of the Woods, Beltrami, and Roseau counties in northwestern Minnesota (Figure 1). Coniferous forests and mixed coniferous-deciduous forests cover much of the study area, and spruce grouse are often seen and hunted there.

METHODS

Spatial Sampling Design

We are interested in landscapes containing 1 of 2 focal cover types associated with spruce grouse—jack pine (JP) and black spruce (BS). Spruce grouse also may be found in stands of tamarack, balsam fir, and white cedar, but the associations with those cover types are not as strong. To further reduce variation from sources in which we are not interested, we will screen the population of landscapes from which we will draw a sample. The population of landscapes will include square plots of 64 ha (i.e., 800 m x 800 m) containing near the center a forest stand that meets the following criteria: (1) the cover type is JP or lowland BS, (2) it is not classified as stagnant (i.e., not commercially productive), (3) it is 4–16 ha in size, (4) it is within 1.6 km (1 mile) of an accessible road in the study area, and (4) it is on public land. Furthermore, if the cover type is JP, the stand will be 11–50 years old (Szuba and Bendell 1983: 203) and the site index (i.e., a relative measure of productivity for growing trees based on soil type, hydrology, and other factors) will be 45–65; and if the cover type is BS, the stand will be 19–120 years old (Szuba and Bendell 1983: 203, Lycke et al. 2011) and the site index will be 25–50.

We will draw a random sample of points from the area defined by all forest stands meeting the criteria listed above. Each stand containing a random point (i.e., a focal stand) will be the center of a landscape that will be sampled. From a grid of points regularly spaced 200 m apart across the study area we will select the ones nearest the centroid of the randomly selected focal stands. Each of these focal grid points will be the northeastern 1 of 4 grid points in the middle of a 4- × 4-point grid that will define a plot (Figure 2). Circles of 100-m radius from the 16 grid points will completely fill the 64-ha plot without any overlap.

Estimating Grouse Density

In Koochiching County during the early-1970s the breeding display period of spruce grouse lasted 26 days, from within 2 days of snowmelt being complete (e.g., late-April) until late-May, with less intense display continuing until mid-June (Anderson 1973: 27-28, 97, 105). However, more recently in Wisconsin (WI), most spruce grouse quit responding to the cantus by early May (Nick Anich, WIDNR, personal communication). The flutter flight displays of males are detectable from at least 50 m away and perhaps up to 100 m away (Keppie 1992, Worland et al. 2009). The cantus, or song, of females is not performed as regularly as the flight displays of males, but females may be recorded during surveys of spruce grouse (Worland et al. 2009).

Flight displays and the cantus may be elicited by broadcasting a recording of the female cantus (MacDonald 1968). Although unsolicited breeding displays may occur primarily or only during a few hours near sunrise (Keppie 1992), broadcasting a call may be “effective in locating breeding males throughout the day” (Bouta 1991: 6).

We will visit each randomly selected 64-ha landscape plot once, beginning 60 minutes before sunrise (Keppie 1992:309), and survey for spruce grouse for 8 minutes at each of the 16 regularly spaced grid points. Each survey will be divided into 4 intervals of 2 minutes each. Detections of spruce grouse, including the direction and estimated distance, will be recorded separately for each interval and each grouse. During the last interval we will broadcast a recording of the female cantus for 30-second bursts spaced 30 seconds apart (Jakob et al. 2010). The first 3 intervals will consist of silent observation, because we want to estimate probabilities of detection based on spontaneous grouse behaviors, rather than just behaviors that are elicited.

We will use time-of-detection methods (Aldredge et al. 2007) to estimate probabilities of detection and densities of spruce grouse. The methods involve treating each 2-minute survey interval as a separate occasion, so each grouse detected during an 8-minute survey has a 4-digit detection history consisting of 1s (i.e., detected during the interval) and 0s (i.e., not detected). The data are analyzed in a capture–recapture framework in which we will estimate the potential effects of detection distance, broadcasting the cantus, observer behavior after first detecting a grouse at a survey point (Riddle et al. 2010), wind speed, presence of precipitation, time of day, day of the year, sex of the bird, and forest cover type on probabilities of detection. We will also estimate grouse abundance with these data and calculate grouse density by applying a fixed-distance radius to define an area in which probabilities of detection may be assumed to be similar (e.g., 50 m). However, reasonable precision of estimates of density may require sample sizes greater than we will generate during this study. If so, we will use plot-specific probabilities of detection or simply the number of spruce grouse detections as the response variable in the landscape-scale habitat analysis (see **Data Analysis** below).

Habitat Characteristics and Vegetation Sampling

Characteristics of the landscape (i.e., plot) that are potentially related to spruce grouse habitat will be collected from relevant GIS databases. Such characteristics of landscapes include the proportions, juxtaposition, and diversity of different cover types; densities of roads and other edges between cover types; and presence or proportions of native plant community types. Many habitat characteristics at the scale of the forest stand, including age, species composition, size, shape, and management history, will be collected from forest inventory data. However, previous studies have indicated that certain structural characteristics of forest stands that are not available from inventory data also may affect use by spruce grouse. Therefore, we will measure those characteristics in the focal stands (i.e., the randomly selected stands around which the landscape plots were placed).

We will measure vegetation characteristics at the focal grid point (i.e., the point nearest the centroid of the focal stand) for use in addressing Objective 2. We will measure height to the top and bottom of the live forest overstory canopy 10 m away in the 4 cardinal compass directions. We will estimate densities of overstory stems (i.e., >3 m tall) and understory shrubs (i.e., 0.2–3 m tall) using the point-center-quarter method (Higgins et al. 2005: 531). We will estimate vertical cover above 1.5 m with a densiometer. We will confirm the 1–3 most dominant tree species from forest inventory data and record the predominant ground cover as 1 of 10 standard categories from MNDNR Forestry’s CSA Tatum Guide. To account for spatial variation within the stand, we will repeat the point-center-quarter and vertical cover measurements at 2 additional vegetation sampling points 30 m apart along a transect toward the center of the stand. The compass bearing for the transect will be determined from maps before going afield. If the center of the stand is < 60 m from the focal grid point, we will use a randomly selected compass bearing for the transect and place the 2 additional vegetation sampling points 30 m on either side of the focal grid point.

Data Analysis

To address the objectives about habitat relationships, we will specify *a priori*, fit, and rank regression models representing our hypotheses about which combinations of habitat characteristics are most highly correlated with space use by spruce grouse during spring (Burnham and Anderson 2002). We will restrict the number of variables per model to 1/20th of the sample size for the analysis. To help limit the number of variables under consideration, we will focus on variables with the strongest purported influence in the literature, a wide range of observed values, and low correlation with other variables, and we will consider combining similar variables into indices.

For the landscape scale analysis the response variable will be density of spruce grouse, plot-specific probability of detection, or number of grouse detections, depending upon whether or not we attain sufficient precision when estimating density or probability of detection. For stand scale analyses the response variable will be presence or absence of spruce grouse. For Objective 2, the measures of forest structure identified as important during previous studies were tree density and tree height, which are data we will have for only the focal stands. In addition to regression analysis of the sample of focal stands with more detailed vegetation measurements, we will summarize grouse detection data by known characteristics of all stands in the sampled landscapes (e.g., proportions of points with a grouse detection in different overstory cover types).

RESULTS AND DISCUSSION

During spring of 2011, we surveyed at least 38 plots centered on stands of jack pine and 18 plots centered on stands of lowland black spruce. We detected 44 spruce grouse in those plots; approximately half were males and half were females. The 2011 field season was not complete when this summary was written, and we plan to continue collecting data during the spring of 2012.

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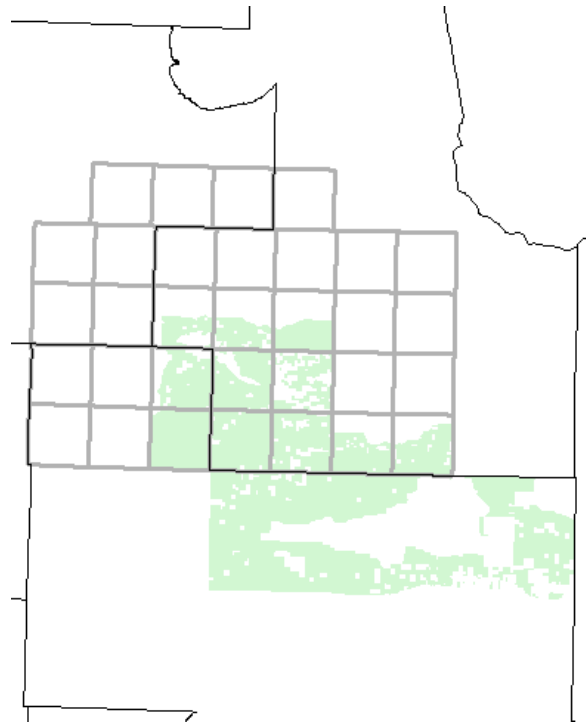


Figure 1. Study area (gray township lines) for spruce grouse research relative to Red Lake Wildlife Management Area (shaded area) and county boundaries (black lines) in northwestern Minnesota.

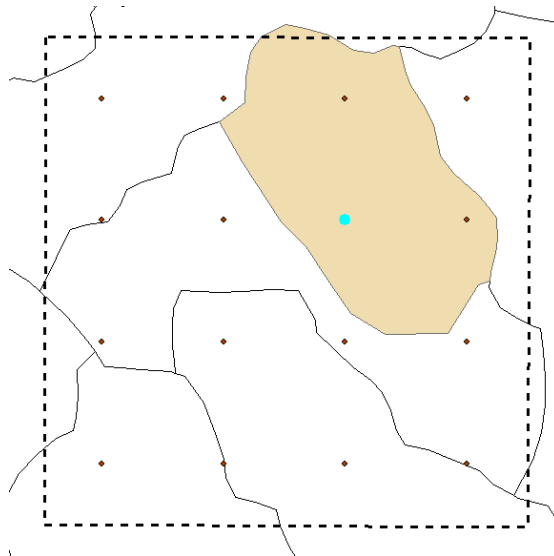


Figure 2. Diagram of a 64-ha spruce grouse survey plot (i.e., landscape; thick dashed line) showing the focal stand (shaded area), other forest stands (solid lines), the grid of 16 survey points spaced 200 m apart, and the survey point nearest the centroid of the focal stand (large blue point).

LANDSCAPE CHARACTERISTICS ASSOCIATED WITH DANCING GROUNDS OF SHARP-TAILED GROUSE

Michael A. Larson and J. Wesley Bailey

SUMMARY OF FINDINGS

We are developing a habitat model to describe the landscape characteristics associated with “dancing grounds” of sharp-tailed grouse (*Tympanuchus phasianellus campestris*) across their range in Minnesota. We provide preliminary results, including a table and figure, but our analyses are not complete and the results are subject to potential revision.

INTRODUCTION

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

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

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

OBJECTIVES

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

STUDY AREA

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

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

METHODS

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

Use-availability Data

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

We selected the sample of available sites from the spatial extent defined in the Study Area section above. The only other constraint we applied for the area from which available sites were randomly selected was that the forest and non-habitat cover types (defined below) were excluded. The definition of the study area, or spatial extent, is important for use-availability comparisons (Johnson 1980). Using a more restrictive study area (e.g., within a limited-distance buffer of known dancing grounds) would lead to inferences focusing on specific characteristics of patches of open cover types (e.g., area, edge density). Using a broader extent for the study area (e.g., all of northern Minnesota) likely would lead to inferences emphasizing the importance of open lands in general. We sought a balance between those extremes.

Landscape Data

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

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

type, regardless of classification by the MN-GAP. Then we added areas with herbaceous vegetation cover practices from the 1997 Conservation Reserve Program (CRP, Minnesota Natural Resources Conservation Service 2010) to our undisturbed grass cover type, regardless of the MN-GAP or NWI classifications.

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

Model Set

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

We used different combinations of the variables to define 73 *a priori* models (Table 2). Thirty, 10, 9, 10, 2, 4, 2, 3, and 2 of the models had 3, 4, 5, 6, 7, 8, 9, 10, and 11 parameters, respectively, including an intercept term. The 73rd model was the global model, which had 19 parameters, because it did not include the OPEN covariate, which was the sum of the area in 3 other cover types that were included in the global model. Several of the models were formulated to be similar to the best models of Hanowski et al. (2000) and Niemuth and Boyce (2004). This is a relatively large set of *a priori* models, because there are relatively few previous studies and there is still much uncertainty about the importance of different landscape characteristics.

With a second phase of analysis we are investigating models that contain variables measured at different spatial scales, not just the 800-m scale. During future phases of analysis we plan to investigate separate models for the eastern and northwestern portions of the study area, which differ in their proportions of different land-use/land-cover categories (Figure 1).

Model Fitting

The most appropriate way to analyze and interpret data from a use-availability study design is still debated in the literature (Keating and Cherry 2004, Johnson et al. 2006). We found the approach advocated by Lele and Keim (2006), which is a form of logistic regression, to be the most appealing, because it addressed potential concerns about logistic regression that were raised by Keating and Cherry (2004), and the concept of weighted distributions upon which it is based is more intuitive than alternative approaches to the analysis. We fit our models using scripts for programs R and WinBUGS provided by S. Lele (personal communication), which were based on partial likelihood and data cloning methods described by Lele (2009). We used AIC values to rank the *a priori* models based on how well they fit the data. We assessed the fit of the global model to the data using *k*-fold cross validation (Johnson et al. 2006).

RESULTS AND DISCUSSION

We used 1,245 randomly selected available sites and 249 used sites in our analyses. Our sample of used sites excluded 3 of the 252 dancing grounds observed at least once during 1991–1993, because they were outside the 85% kernel boundary (Figure 1). Both methods used to generate initial values resulted in the same AIC rankings for the best 5 models, which

had 9–19 parameters. The global model was the best-fitting model in the *a priori* set, because it was the only model that contained all 7 of the parameters that were most highly correlated with the presence of a dancing ground (i.e., had estimates whose confidence intervals did not include 0; Table 2, Figure 2). We did not include distance to road (DIRD) in Figure 2, because that parameter confirmed that dancing grounds closer to roads were more likely to be documented during surveys rather than providing insight about habitat selection. The second best model ($K = 12$, $\Delta AICc = 1,311$) included DILK, SHRB, RDDN, FRST, and OPEN, but not SIMP or DIRD. Looking at the best model with a given number of parameters for models with 3–8 parameters ($n = 6$ models), the distance to nearest lek (DILK) variable occurred in all of them and the area of the shrub cover type (SHRB) occurred in 4 of them.

Results are preliminary and are subject to revision based on continuing work on this project. When our analyses are complete we will compare results to those of Hanowski et al. (2000) and Niemuth and Boyce (2004), who have developed similar models of landscape characteristics associated with the dancing grounds of sharp-tailed grouse.

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

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

^a McGarigal et al. (2002).

Table 2. Estimates of parameters for the global logistic regression model of the presence of sharp-tailed dancing grounds in Minnesota.

Variable	Beta	SE	95% LCL ^a	95% UCL ^a
Intercept	-4.13	0.49	-5.09	-3.18
GRSU	0.02	0.27	-0.50	0.54
GRSD	0.52	0.26	0.01	1.03
SEDG	0.14	0.20	-0.25	0.52
CROP	-0.13	0.48	-1.06	0.80
SHRB	-0.84	0.36	-1.55	-0.13
BOG	-0.58	0.39	-1.35	0.20
FRST	-0.89	0.40	-1.67	-0.11
SIMP	0.33	0.15	0.03	0.62
DILK	-1.67	0.25	-2.15	-1.18
DIGR	0.13	0.33	-0.52	0.77
DIFO	0.17	0.17	-0.16	0.50
DIRD	-1.05	0.46	-1.96	-0.14
RDDN	-1.00	0.23	-1.45	-0.55
EDBS	0.27	0.18	-0.07	0.62
EDBO	0.11	0.14	-0.15	0.38
EDOF	-0.22	0.18	-0.58	0.14
PAFO	0.19	0.15	-0.10	0.47
PASH	0.13	0.12	-0.12	0.37

^aLCL= lower confidence limit and UCL = upper confidence limit.

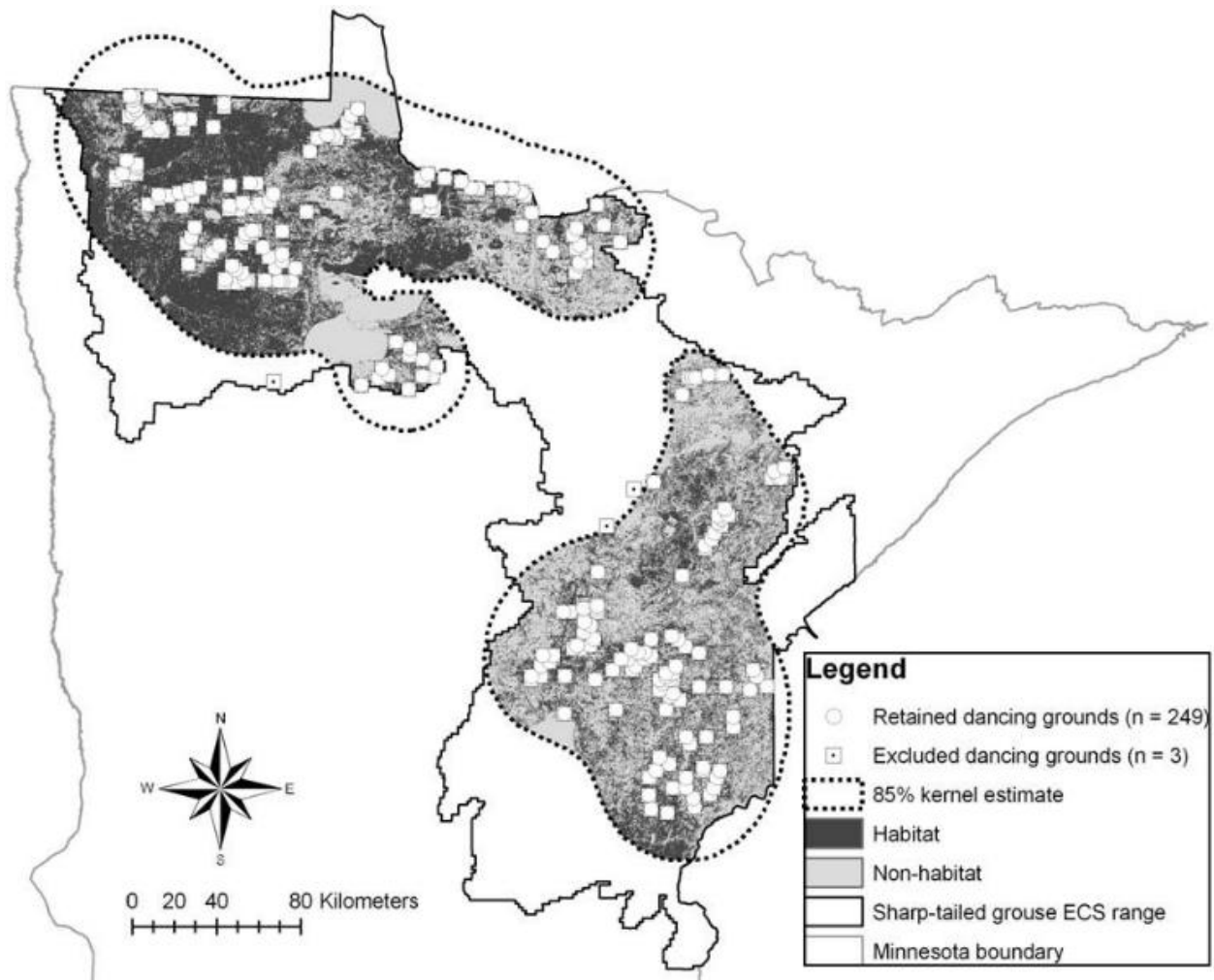


Figure 1. Map of the spatial extent of the habitat selection model for sharp-tailed grouse in northern Minnesota (shaded areas). The extent was defined as occurring within (1) occupied Ecological Classification System subsections and (2) the 85% kernel estimate of space use, based upon the locations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

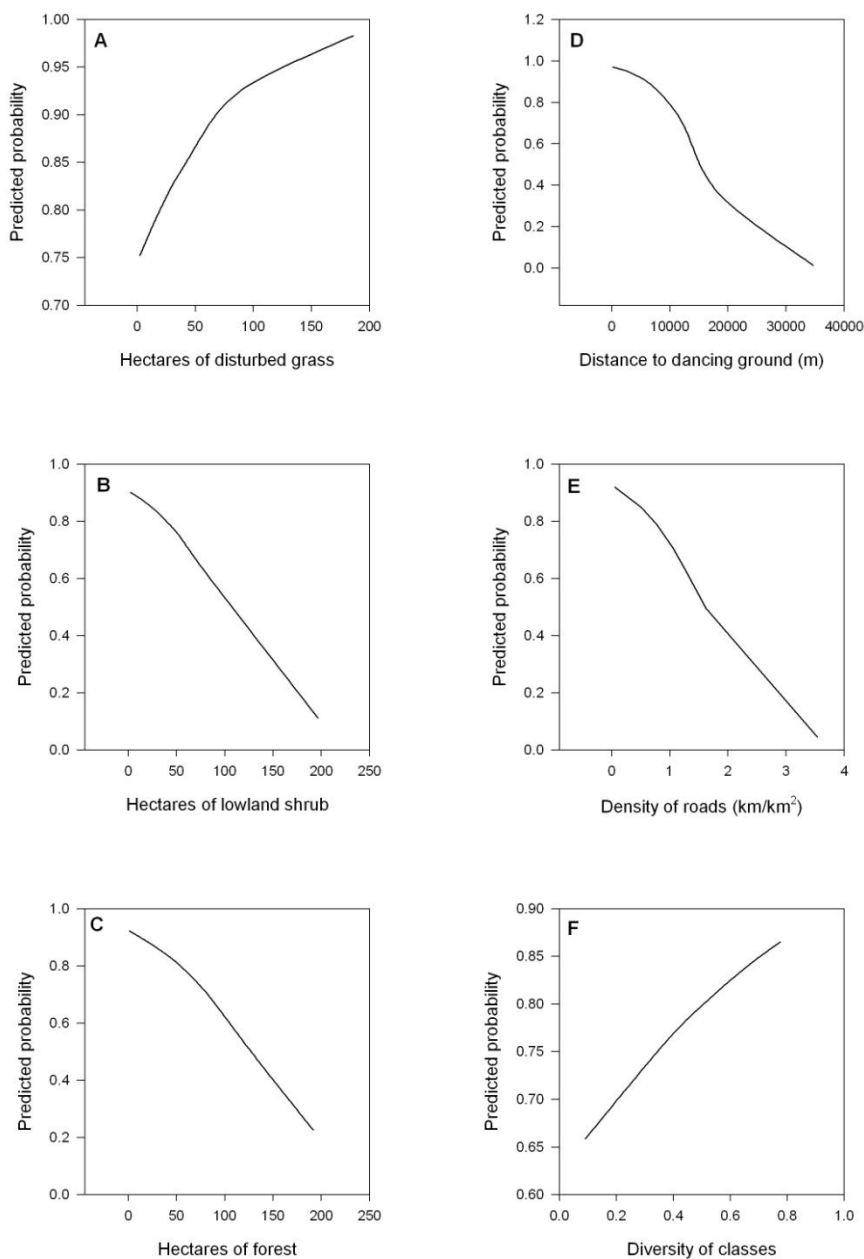


Figure 2. Predicted probabilities of use of dancing grounds of sharp-tailed grouse (points buffered by 800 m, 201-ha circles) relative to landscape characteristics with statistically significant parameter estimates in a use-availability model based on dancing grounds documented during 1991–1993 in Minnesota. The abscissa in panel F is Simpson’s Diversity Index based on land cover classes.