



DETERMINING AN EFFECTIVE APPROACH FOR CAPTURING NEWBORN MOOSE CALVES AND MINIMIZING CAPTURE-RELATED ABANDONMENT IN NORTHEASTERN MINNESOTA

Glenn D. DelGiudice, William J. Severud¹, Tyler R. Obermoller, Katie J. Foshay, and Robert G. Wright²

SUMMARY OF FINDINGS

The goal of our recently initiated moose (*Alces alces*) calf research in northeastern Minnesota is to enhance our understanding of calf survival and specific causes of mortality and to assess their quantitative impact on the population's performance. The instantaneous probability of death (hazard) for northern ungulates is highest at birth, and although it declines sharply during the first 12 months, it is markedly higher than during the subsequent prime years of life. Fulfilling the primary goal of the calf study requires 3 things, the ability to: 1) capture and Global Positioning System (GPS)-collar a sample of newborn moose calves representative of the population, 2) closely monitor their movements and survival, and 3) conduct rapid investigative responses to calf mortalities. Unexpectedly, 2013's capture operations resulted in a relatively high number of neonates being abandoned within 48 hours post-capture (DelGiudice et al. 2013). This was unacceptable and prompted our team to examine closely all of the associated data collected to better understand capture-related abandonment and adopt a new approach for this year (2014) which would limit this undesirable side-effect of capture operations. We expected hourly fixes from our GPS-collared neonates and their dams to facilitate our accomplishment of this task. Our objective was to further evaluate the helicopter-assisted captures of moose neonates in 2013 and present preliminary analyses comparing the 2 all-ground calf capture approaches used in spring 2014. During 8-17 May 2013 we captured 49 newborn calves of 31 dams. The adult fled with the approach of the helicopter in most cases. Ultimately, 7 of 31 (23%) dams abandoned 9 of 49 (18%) calves, and twins (8 versus 1 singleton) appeared to be predisposed to abandonment. Hourly location data of neonates and dams indicated that capture-related abandonment involves movement behavior which is highly variable and complex. During the 1-6-hour interval post-capture, abandoning and non-abandoning dams were similar distances from calf capture-sites and their calves, but by the 7-12-hour interval mean distances of the 2 groups were diverging. By the 13-18-hour interval post-capture the non-abandoning dams were returning to their calves, whereas the abandoning dams were moving farther away. Paradoxically, movements of most of the abandoning dams included periodic returns to their calves, just as had occurred for some of the non-abandoning dams within the 48 hours post-capture. While our intense monitoring of movement behavior has illuminated much about capture-related abandonment, additional study should provide a greater understanding of its effects on the survival of neonates. During 8-15 May 2014, our initial approach to ground captures resulted in 7 of 12 (58%) neonates ultimately being abandoned by 5 of 9 dams (56%) at least in partial response to capture operations. Adhering to our Abandonment Contingency Plan, we successfully recovered 6 of the 7 calves and transported them to the Minnesota Zoo. Beginning 21 May, we reduced our capture teams to 2 people and limited our handling of calves to fitting the GPS collar quickly over the head to the neck and determining sex. By 19 June, we captured an additional 13 (8 males, 5 females) calves from 10 dams with no capture-related abandonment. These captures included 3 sets of twins and 7 singletons. While it is difficult to ascertain that any one factor was most important, we think that reducing the capture team to 2 people and mean handling time to <60 seconds likely had the most positive impacts.

¹Graduate Research Assistant, University of Minnesota, Department of Fisheries, Wildlife, and Conservation Biology, 2003 Upper Buford Circle, Ste. 135, St. Paul, MN 55108, ²Wildlife GIS Specialist, Minnesota Information Technology Services @ Department of Natural Resources, Section of Wildlife, 5463-C West Broadway, Forest Lake, MN 55025.

INTRODUCTION

The winter moose population in northeastern Minnesota has decreased 51% from an estimated 8,840 moose in 2006 to the current (2014) estimate of 4,350 (Lenarz et al. 2009, 2010; DelGiudice 2014). Climate change (i.e., warming temperatures) has been implicated as an underlying factor in the declines of the state's northeastern and northwestern populations (Murray et al. 2006; Lenarz et al. 2009, 2010). In the latter, malnutrition and pathogens additionally were identified as contributing factors; now recent aggressive study in the northeast is accumulating evidence of similar impacts of such factors (Butler et al. 2013). Mean annual natural mortality rates of adults were similarly high in the northwest and northeast (21%, Murray et al. 2006; Lenarz et al. 2009; Butler et al. 2013; R. A. Moen, Natural Resources Research Institute [NRRRI], personal communication), and have remained elevated in the northeast up until this year (Carstensen et al. this issue).

Adult survival has a greater impact on ungulate population dynamics than that of juveniles; however, high annual variability in juvenile survival also can have a pronounced influence on a population's growth rate (Gaillard et al. 1998, 2000). Across much of moose range in Ontario, Canada, declining moose numbers and winter calf:cow ratios have been a cause for concern since the 1990s. Overall, natural causes were the leading mortality forces on calves. In Algonquin Provincial Park it was primarily predation by black bears (*Ursus americanus*) and wolves (*Canis lupus*), whereas in a Wildlife Management Area where hunting was permitted and accounted for 16% of calf mortality it was malnutrition, exposure, and tick-related mortality (Patterson et al. 2013).

Average pregnancy rates have been relatively high (83%) in northeastern Minnesota, but annually it has been variable (range of 55-100%, M. S. Lenarz, Minnesota Department of Natural Resources [MNDNR], unpublished data; Butler et al. 2013). Recently, Lenarz et al. (2010) reported an average annual survival rate of 0.40 for calves in the northeastern population. These crude estimates were based on fixed-wing flights conducted during May-June to determine whether radiocollared cows had newborn calves present, and again in April-May of the following year to determine if calves were still present. Further, based on the MNDNR's annual aerial moose survey conducted in January, the calf:cow ratio has declined from 0.52 in 2005 to 0.36 in 2012, and was as low as 0.24 in 2011 (Lenarz 2012).

The average annual survival rate of northeastern Minnesota moose was consistent with estimates from moose populations elsewhere where black bears and wolves were common (Hauge and Keith 1981), yet black bear predation on moose calves can be highly variable across North America (see Ballard's 1992 review). Determination of cause-specific mortality of calves was not part of the Lenarz et al. (2009, 2010) study design, consequently very little is known about the specific mortality forces or potential contributing factors.

The goals of our recently initiated moose calf research in northeastern Minnesota, a companion study to the MNDNR's adult moose study, are to enhance our understanding of the seasonal and annual survival of calves, specific causes of mortality and associated factors, and to quantify their impacts on the population's performance. The hazard, or instantaneous probability of death, for northern ungulates is highest at birth, and although it declines sharply during the first 12 months, it is markedly higher than during the subsequent prime years of life (DelGiudice et al. 2002, 2006; Lenarz et al. 2010). Fulfilling the primary goal of the calf study requires the ability to: 1) capture and GPS-collar a sample of newborn moose calves representative of the population in northeastern Minnesota, 2) closely monitor the movements and survival of moose calves, and 3) mount a rapid investigative response to calf mortalities.

Currently, this study is funded to support 3 field seasons. To efficiently and cost-effectively obtain a sample size of 50 newborn calves during the first spring (2013), we opted for capture and handling by an experienced helicopter capture crew (Quicksilver Air, Inc., Fairbanks, Alaska, and Peyton, Colorado). An inherent goal of any wild animal capture operation is to minimize capture-related injury or mortality of the study subjects, and additionally, in the case of calves, to minimize capture-related abandonment. Livezey (1990:193) defined abandonment as

“the permanent separation of mother and young causing the death of the young” and tried to focus increased attention on this potentially serious side-effect of such field operations so that more might be done to minimize associated losses. What became evident from his literature review and communication with many ungulate researchers was that the reported capture-related abandonment rates were highly variable, sorely underestimated, and rarely the primary focus of wildlife research.

Having captured more than 600 newborn moose calves, as well as neonates of other ungulate species, the company awarded our capture contract is considered one of the leading helicopter capture companies. However, 2013’s capture operations resulted in an unexpected and unacceptable number of captured neonates being abandoned. We carefully reviewed the data generated from those operations to better understand capture-related abandonment and to develop a more acceptable protocol for this year (2014). Having both the neonates and their respective dams fitted with GPS collars would be a significant aid to accomplishing this.

OBJECTIVE

1. To further evaluate the helicopter capture operations of newborn moose calves in 2013 and to compare the 2 all-ground calf capture approaches used in spring 2014 in northeastern Minnesota. In a companion research summary (please see Severud et al. this issue), we expand on our description and evaluation of monitoring the GPS-collared calves and dams associated with the new capture approaches and mortality investigations, and we present preliminary findings related to cause-specific mortality.

STUDY AREA

The 6,068-km² study site for this calf research is the same as that of the Environmental and Natural Resources Trust Fund (ENRTF)-supported research addressing survival and cause-specific mortality of adult moose in northeastern Minnesota (Figure 1). This area has been classified as the Northern Superior Upland region (MNDNR 2007) and is characterized by a variety of wetlands, including bogs, swamps, lakes, and streams; lowland stands of northern white cedar (*Thuja occidentalis*), black spruce (*Picea mariana*), and tamarack (*Larix laricina*); and upland conifers of balsam fir (*Abies balsamea*) and jack (*Pinus banksiana*), white (*P. strobus*), and red pines (*P. resinosa*). Trembling aspen (*Populus tremuloides*) and white birch (*Betula papyrifera*) occur on the uplands, often intermixed with conifers. Open lands included lowland and upland deciduous shrub and sedge meadows.

Potential predators of adult moose and their calves include gray wolves and black bears (Fritts and Mech 1981, Erb 2008, Lenarz et al. 2009, Garshelis and Noyce 2011). White-tailed deer (*Odocoileus virginianus*) share most of the study area with moose; their pre-fawning densities are managed at ≤ 10 deer per square mile (MNDNR 2011). The State moose hunt in northeastern Minnesota has been restricted to adult bulls-only since 2007 and accounted for 1.1-1.9% of the overall population (Lenarz 2011). A total of 87 licenses were purchased in 2012 for the State moose hunt, and 46 adult bulls were harvested. Sixteen moose (11 bulls, 5 cows) were harvested during the 1854 Treat Authority’s hunt and 20 bulls were harvested by members of the Fond du Lac Indian Band. Due to rapidly declining numbers, the State moose hunting season was cancelled in 2013 until further notice.

METHODS

Beginning 1 May 2013, we began monitoring closely the locations and movements of 52 GPS-collared (Iridium GPS collars, Vectronic Aerospace, Berlin, Germany) adult female moose, which were determined to be pregnant during the previous winter by serum progesterone concentrations. Additionally, we similarly monitored 7 collared adult females not blood-sampled

during winter capture; their pregnancy status was unknown (see DelGiudice et al. 2013 and Severud et al. 2013 for additional details). Our primary monitoring objective was to record when and where pregnant females made their “pre-calving move” (Bowyer et al. 1999, Severud et al. 2013, McGraw et al. 2014). This is an atypical, long distance move that often occurs just prior to localization and calving (Figure 2). During early May 2014, we began similar monitoring of 16 known pregnant GPS-collared females and 50 collared females of unknown pregnancy status (see Severud et al. this issue); 1 of the pregnant and 6 of the “unknowns” abandoned calves in 2013, consequently their calves were not selected for capture this year.

The Iridium collars of the adults were programmed to record hourly fixes during May 2013 and May-June 2014 when most of the calving occurred. Adult location fixes, and subsequently calf fixes, were transmitted 4 and 8 times per day, respectively, to our base station located about 59 km north of the Twin Cities. We had continuous computer access to the base station. Additional details of our monitoring process are presented by Severud et al. (2013, and this issue).

We assumed that once cows made their pre-calving move then localized, they calved within 12 hours (R. A. Moen, NRRI, personal communication). We then allowed an additional 24 hours for bonding between the dam and her calf or calves for an estimated minimum total bonding time of 24-36 hours. Once monitored females had calved and were allowed this minimum bonding time, the calves were identified as “eligible” for capture and handling. Typically, by capture, actual bonding times were markedly longer than 24-36 hours. Each morning during the May 2013 operations our team provided the commercial capture crew with a list of females (identification numbers, VHF frequency) and their most recent GPS coordinates. Typically, in May 2014, we similarly identified cows eligible to be approached from the ground for calf capture.

In May 2013, the helicopter capture crew located the target dam from the air and then landed some distance away to allow the handler(s) to approach calves on foot. The handling protocol included fitting an expandable Globalstar GPS Calf Collar (440 g, Vectronic Aerospace, Berlin, Germany) over the head; fixing ear-tags; collecting blood by syringe for various analyses; weighing the calf; recording morphological measurements and a rectal temperature; and a physical examination to record any noteworthy injuries or abnormalities. We planned the complete handling protocol to require about 4-6 minutes per calf to limit separation from the dam (Keech et al. 2011), and in the case of twins, an attempt was made to handle both calves. Capturing and collaring both members of a twin set limited the risk of the dam abandoning the twin being handled with the one not being handled (M. A. Keech, Quicksilver Air, Inc., personal communication) and would provide a more thorough understanding of calf survival. Further, an important field objective was to capture, handle, and release twins at the same time and in close proximity (Keech et al. 2011). When twins were identified at captures in 2013 and 2014, handling crews achieved this objective with 100% success.

Initially, in May 2014, our plan was to have 3-person capture teams use the 2013 calf-handling protocol during ground captures. However, apparent capture-related abandonments of calves during 8-15 May prompted us to reconsider and revise the protocol. We trimmed our capture teams to 2 people and limited our handling to simply fitting the GPS collar and sexing each neonate, while simultaneously noting any injuries or abnormalities. Twins were collared simultaneously, 1 team member per calf. An additional capture would not be attempted until our monitoring documented that the dam had returned to its calf or calves post-capture.

The Globalstar GPS calf collars fitted in 2014 were the same as in 2013, except the collar band material was modified to minimize the risk of potential abrasions to the back of the calves' necks. As part of the revised capture protocol we employed during 21 May-19 June 2014, we removed a plastic sleeve fitted around the top of the expandable calf collars in an attempt to make them appear less obtrusive and conspicuous to the dam. Finally, as a precautionary measure, we used a commercial scent-blocking product on the handlers' clothes, gloves, and on the collars.

For our May 2014 operations we had developed an Abandonment Contingency Plan in collaboration with our consulting veterinarian and veterinary staff at the Minnesota Zoo. If our computer-monitoring showed that a dam was away from its newly captured calf/calves overnight or had reunited and then left them again, it was flagged as a potential abandonment. If a calf had been alone for 24 hours, we deployed a retrieval team. Precipitation or below normal May temperatures would hasten our response to recover abandoned calves. Recovered calves were brought to the Minnesota Zoo, which had agreed to act as a staging area before distributing calves to other zoos in the U. S. which had agreed to accept them. The Minnesota Zoo had agreed to accept 4-5 calves for its exhibits. All captures and handling protocols followed requirements of the Institutional Animal Care and Use Committee for the University of Minnesota (Protocol 1302-30328A).

RESULTS AND DISCUSSION

2013 Moose Calf Captures

During 8-17 May 2013 we captured 49 (25 females, 24 males) newborn calves of 31 dams (Figure 3). Our process for monitoring and determining when GPS-collared dams had calved and met our minimum threshold of bonding time with their calves was highly successful (Severud et al. 2013). As reported last year, most of these dams were non-aggressive during the capture and handling of their calves (DelGiudice et al. 2013), particularly compared to dams of captured neonates in Alaska and Ontario (Keech et al. 2011, Patterson et al. 2013).

The twinning rate of the 2013 study cohort was unusually high at 58% (18 of 31 dams); 11, 4, and 3 were female/male, female/female, and male/male sets of twins, respectively. Thirteen adult females had singletons (6 females, 7 males). The high twinning rate likely had much to do with restricting our capture of newborn calves to a period early in the calving season when the birthing of twins is most likely to occur (see DelGiudice et al. 2013). Twinning rates can be highly variable. Patterson et al. (2013) reported an overall twinning rate of 16.7% in a 4-year study of moose calves in central Ontario. Keech et al. (2011) observed an overall average twinning rate of 42% (24-52%) for collared cows ≥ 3 years old during their 7-year study. The long-term average annual twinning rate in northeastern Minnesota may be about 29% (M. Schrage, Fond du Lac Band, unpublished data), whereas in northwestern Minnesota, Murray et al. (2006) reported an average twinning rate of 19%.

During 2013 we unexpectedly documented a high level of capture-related abandonment of calves by dams, but they were distributed at a low frequency throughout our 10-day operation (Figure 3). In almost all captures the adult fled at the approach of the helicopter or handler(s) and did not return to its calf or calves for varying lengths of time. Specifically, 7 of 31 (23%) dams abandoned 9 of 49 (18%) calves, apparently prompted by capture-related activities. Over the decades, rates of capture-induced abandonment of moose neonates have ranged up to 28% of calves captured, and up to 40% for caribou (*Rangifer tarandus*) (see the review of Livezey 1990). In a recent Alaskan study, researchers similarly used helicopters to capture 422 moose neonates and experienced 32 (8%) capture-related abandonments or mortalities (Keech et al. 2011). Because ours was the first study of free-ranging moose neonates fitted with GPS collars, it permitted nearly continuous monitoring of the calves and their GPS-collared dams. Indeed, unlike in other studies employing VHF telemetry, there was almost no way abandonment could be underestimated unless the collars malfunctioned. In a 4-year study involving all ground captures of moose neonates, capture-related abandonment of 5% of the calves was reported (Patterson et al. 2013). As with many of the dams in that study, VHF collars were fitted to the calves, but a high proportion of the dams were not collared. Consequently, the estimate of capture-related abandonment may have been biased.

In our study abandonment behavior was highly variable from day to day, and discernible patterns associated with various independent variables generally were not evident. On only 1 day did more than 1 dam abandon its calf, and on several days no dams abandoned calves,

despite a relatively high number of calves being captured (Figure 3, 16 May). Six of 25 (24%) females and 3 of 24 (13%) males were abandoned by their dams following capture and handling; however, according to a 2-sided Fisher's Exact Test the difference was not significant ($P = 0.464$). As previously reported, there was no difference in mean birth-date, capture-date, minimum bonding time, body mass, hind leg length, or rectal temperature between calves abandoned versus those not abandoned (DelGiudice et al. 2013). However, whether 1 or both twins of a set were abandoned, a 1-sided Fisher's Exact Test showed an apparent ($P = 0.104$) predisposition of twins to abandonment compared to singletons (Figure 4). Prompted by capture and handling, 33% of dams of twins abandoned their calves, whereas only 7% of dams of singletons responded to calf capture by abandoning neonates.

Hourly location data accumulated by GPS collars on the neonates and dams post-capture indicated that capture-related abandonment involves movement behavior which is highly variable and more complex than a dam simply taking flight in response to disturbances associated with the capture operation. Our first hourly locations indicated that on average the dams which did not ultimately abandon their calves actually fled farther than the those that did abandon (257 versus 183 m), although not significantly so (Figure 5). During the 1-6-hour interval post-capture, abandoning and non-abandoning dams were similar distances from the calf capture-sites and their calves, but by the 7-12-hour-interval mean distances of the 2 groups were diverging (Figure 5). By the 8-13-hour interval the non-abandoning dams clearly were returning to their calves, whereas the abandoning dams were steadily moving farther away from the capture-sites and their calves (Figure 5).

Paradoxically, abandoning movements included periodic returns by some of the dams to their calves, just as had occurred for some of the non-abandoning dams within 48 hours post-capture. We used the mean distance non-abandoning dams were away from their calves during the first 48 hours post-capture (256 m) as a threshold distance to indicate that dams had essentially reunited with their calves at varying points. Five abandoning dams and 5 non-abandoning dams returned a mean 1.5 (SE = 0.2, range = 1-2) and 1.3 times (SE = 0.3, range = 1-3), respectively. Interestingly, though they shared similar aspects of this post-capture movement behavior, the abandoning dams were much farther from their calves just before returning (mean = 1,531, SE = 210, range = 1,105-2,223 m) than the non-abandoning dams (mean = 582, SE = 80, range = 402-812 m), and they did not stay with their calves as long (mean = 1.5, SE = 0.2, range = 1-2 hours versus mean = 4.7, SE = 1.1, range = 1-7 hours) once they returned. Additionally, 5 of the abandoning dams stayed with their calves immediately after capture for 1-11 hours before leaving. While our intense monitoring of movement behavior has illuminated much about capture-related abandonment, additional analyses should provide a greater understanding of its direct and indirect effects on the survival of these neonates.

2014 Moose Calf Captures

During 8-15 May 2014, our initial approach to ground captures resulted in 7 of 12 (58%) neonates ultimately being abandoned by 5 of 9 dams (56%) at least in partial response to capture operations (Figure 6). Capture-related abandonments began on the first day of captures with a set of twins, after which we stopped operations for several days to reconsider our protocol. The first day's capture had required a long trek (≥ 6.4 km) under challenging field conditions (e.g., occasional deep snow), and we concluded that perhaps we had disturbed the dam well in advance of our capture of the twins. Consequently, we decided to employ a less disturbing ground approach, which we anticipated would be facilitated by rapidly improving field conditions. We also removed the more invasive blood-sampling from our handling protocol, which also would reduce handling time. Overall, from a field perspective, we viewed our captures as progressing smoothly during this initial interval without any unforeseen problems which would lead to abandonment. Our dam monitoring approach (see Severud et al. this issue), targeting GPS coordinates of the center of the dam's calving localization/cluster, and using a hand-held GPS unit in the field allowed us to locate calving sites and calves efficiently.

Search times for capturing calves typically required a mean of 5 minutes (SE = 3.0, range = 0-17, $n = 6$). In 5 cases we did not find the calves we sought to capture (Severud et al. this issue). Handling required a mean 8.9 minutes (SE = 1.8, range = 5-23, $n = 9$); this included twins, and there was no difference for calves abandoned versus those not abandoned. Having removed the disturbance of the helicopter from our capture and handling protocol this year, we initially were quite sure that a capture-related abandonment effect would be minimal, if not eliminated. However, the capture-related abandonments continued to occur during the 8-15 May interval (Figure 6).

Although our sample size was limited to a total of 12 calves by 15 May, twinning appeared to be less of a factor in predisposing calves to capture-related abandonment than during the 2013 operation, but in both cases of dams abandoning twins during this phase, they abandoned both neonates (Figure 4). Additionally, similar to 2013, there was no apparent potential influence of sex of the calf on capture-related abandonment (3 males, 4 females). Overall, mean bonding time (or calf age) at capture during this interval was 49 hours (SE = 4.7, range = 24-76, $n = 9$); there was no difference between calves which were abandoned (mean = 47, SE = 8.4, range = 24-76 hours, $n = 5$) versus those that were not (mean = 52, SE = 3.3, range = 43-58 hours, $n = 4$). These bonding times were similar to those of the 2013 capture operations (mean = 54, SE = 2.7, range = 31-116 hours, $n = 49$) when there also was no difference associated with calves abandoned versus those that were not (DelGiudice et al. 2013).

Adhering to our Abandonment Contingency Plan, we successfully recovered in good condition 6 of the 7 calves which had been abandoned and transported them to the Minnesota Zoo. There, the veterinary staff and zookeepers examined them thoroughly and have since been caring for them. Mean recovery time was 42 hours (SE = 5.8, range = 24-49, $n = 4$) post-capture. The dam of the calf not recovered alive had been with it for 8 hours immediately post-capture, moved away, and then returned to within 200-300 m. Our intense monitoring had shown that the calf was moving about, and so we had hoped that periodic nursing bouts were occurring during the 48 hours post-capture, but a necropsy had concluded that this was unlikely.

With our seventh capture-related abandonment on 15 May, we considered not resuming captures during this calving season. However, during a week of contemplating our mode of operation, we decided to attempt a final new capture approach as described in the Methods section. Beginning 21 May, employing a 2-person team, we began approaching 1 calving dam daily once the minimum bonding time had elapsed. During 21 May-19 June, we captured an additional 13 (8 males, 5 females) calves from 10 dams with no capture-related abandonments. These captures included 3 sets of twins and 7 singletons. Our current success at eliminating capture-related abandonment may be attributable to several factors. While it is difficult to ascertain that any one factor was most important, reducing mean handling time to <60 seconds (0.9, SE = 0.2, range = 0-2 minutes, $n = 10$), for a singleton or twins, may have been most beneficial. Slipping the expandable collar over the head and sexing the calf constituted "handling," but we also carried the calf or calves back to the nearby calving-site for release. The smaller capture team may have contributed to a quieter approach and an apparently briefer search time (mean = 1.5, SE = 1.0, range = 0-11 minutes, $n = 10$), although during both capture phases search times were rather variable. Our allowed bonding time at capture (mean = 74.6, SE = 5.7, range = 53-109 hours, $n = 10$) tended to be longer than during the initial phase and compared to our 2013 operations. Bonding times have been highly variable among studies. Patterson et al. (2013) reported bonding times before capture of 9.5-58 hours (median = 19 hours) on their WMU49 site and <48 hours (48%) and 48-120 hours (52%) at Algonquin Provincial Park. In Interior Alaska, Keech et al. (2011) estimated mean bonding times of 2.6 days (62 hours) and a range of 0.5-11 days (12-264 hours). Finally, the absence of conspicuous ear-tags on calves and collar modifications during the second phase of 2014 captures may have contributed to limiting potential recognition problems between dams and calves, but this is difficult to quantify.

Considerations for Future Capture Operations

Currently, capture-related abandonment of newborns of moose and other ungulates is poorly understood and likely has been sorely underestimated due to the limitations of conventional VHF telemetry. However, our deployment of recently developed expandable GPS collars on moose neonates of GPS-collared dams has demonstrated enhanced potential for increasing our understanding of those factors which have the greatest influence on abandonment behavior and how to minimize that influence. Our hourly location data also indicated that the operative definition of capture-related abandonment and how to respond to suspected abandonment in the field as it occurs warrants re-examination. As challenging as capture-related abandonment has made our calf capture operations during 2013 and 2014 of this survival and cause-specific mortality study, what we have learned and our ability to adapt reflect a significant success with respect to our calf capture protocol. We now will be able to apply what we have learned towards next year's capture operations with the promise of obtaining a more rigorous seasonal sample size of collared calves to add to the 44 calves we have been able to study thus far for survival and natural causes of mortality (Severud et al. this issue).

ACKNOWLEDGMENTS

We thank L. Ross, M. Hass, T. Enright, J. Lodel, B. Betterly, A. Jones, K. Miedtke, and S. Larson, the adult moose study team (E. Hildebrand, D. Pauly, M. Carstensen, M. Dexter), L. Cornicelli, M. Larson, J. Forester, B. Patterson, S. Moore, R. Moen, and M. Schrage for varying combinations of technical input and assistance, logistical support, or other cooperative efforts throughout the planning and operational phases of this study thus far. We are grateful for the flying, capture, and calf-handling skills and efforts of R. Swisher and M. Keech (Quicksilver Air, Inc.) during the 2013 capture operations. We appreciate the statistical consultation of V. St-Louis. This project was funded in part by the Wildlife Restoration (Pittman-Robertson) Program.

LITERATURE CITED

- Ballard, W. B. 1992. Bear predation on moose: a review of recent North American studies and their management implications. *Alces Supplement* 1:162-176.
- Bowyer, R. T., V. Van Ballenberghe, J. G. Kie, and J. A. K. Maier. 1999. Birth-site selection by Alaskan moose: maternal strategies for coping with a risky environment. *Journal of Mammalogy* 80:1070-1083.
- Butler, E. A., M. Carstensen, E. C. Hildebrand, and D. C. Pauly. 2013. Determining causes of death in Minnesota's declining moose population: a progress report. Pages 97-106 *in* L. Cornicelli, M. Carstensen, M. D. Grund, M. A. Larson, and J. S. Lawrence, editors. *Summaries of wildlife research findings, 2012*. Minnesota Department of Natural Resources, Wildlife Populations and Research Unit, St. Paul, Minnesota, USA.
- DelGiudice, G. D. 2013. 2013 Aerial moose survey. Unpublished report. Minnesota Department of Natural Resources, St. Paul, Minnesota, USA. 6pp.
- DelGiudice, G. D. 2014. 2014 Aerial moose survey. Unpublished report. Minnesota Department of Natural Resources, St. Paul, Minnesota, USA. 6pp.
- DelGiudice, G. D., M. R. Riggs, P. Joly, and W. Pan. 2002. Winter severity, survival and cause-specific mortality of female white-tailed deer in north central Minnesota. *Journal of Wildlife Management* 66:698-717.
- DelGiudice, G. D., J. Fieberg, M. R. Riggs, M. Carstensen Powell, and W. Pan. 2006. A long-term age-specific survival analysis of female white-tailed deer. *Journal of Wildlife Management* 70:1556-1568.

- DelGiudice, G. D., W. J. Severud, and R. G. Wright. 2013. Helicopter capture of newborn moose calves in northeastern Minnesota: an evaluation. Pages 40-51 *in* L. Cornicelli, M. Carstensen, M. D. Grund, M. A. Larson, and J. S. Lawrence, editors. Summaries of wildlife research findings, 2012. Minnesota Department of Natural Resources, Wildlife Populations and Research Unit, St. Paul, Minnesota, USA.
- Erb, J. 2008. Distribution and abundance of wolves in Minnesota, 2007-2008. Unpublished report. Minnesota Department of Natural Resources, St. Paul, Minnesota, USA. 11pp.
- Fritts, S. H., and L. D. Mech. 1981. Dynamics, movements, and feeding ecology of a newly protected wolf population in northwestern Minnesota. *Wildlife Monographs* No. 80.
- Gaillard, J.-M., M. Festa-Bianchet, and N. G. Yoccoz. 1998. Population dynamics of large herbivores: variable recruitment with constant adult survival. *Trends in Ecology and Evolution* 13:58-63.
- Gaillard, J.-M., M. Festa-Bianchet, and N. G. Yoccoz, A. Loison, and C. Toigo. 2000. Temporal variation in fitness components and population dynamics of large herbivores. *Annual Review of Ecology and Systematics* 31:367-393.
- Garshelis, D., and K. Noyce. 2011. Status of Minnesota black bears, 2010. Final Report to Bear Committee. Minnesota Department of Natural Resources, St. Paul, USA. 24pp.
- Gasaway, W. C., R. D. Boertje, D. V. Granaard, D. G. Kelleyhouse, R. O. Stephenson, and D. G. Larsen. 1992. The role of predation in limiting moose at low densities in Alaska and Yukon and implications for conservation. *Wildlife Monographs* 120.
- Hauge, T. M., and L. B. Keith. 1981. Dynamics of moose populations in northeastern Alberta. *Journal of Wildlife Management* 45:573-597.
- Keech, M. S., M. S. Lindberg, R. D. Boertje, P. Valkenburg, B. D. Taras, T. A. Boudreau, and K. B. Beckmen. 2011. Effects of predator treatments, individual traits, and environment on moose survival in Alaska. *Journal of Wildlife Management* 75:1361-1380.
- Lenarz, M. S. 2011. 2011 Minnesota moose harvest. Unpublished report. Minnesota Department of Natural Resources, St. Paul, Minnesota, USA. 4pp.
- Lenarz, M. S. 2012. 2012 Aerial moose survey. Unpublished report. Minnesota Department of Natural Resources, St. Paul, Minnesota, USA. 6pp.
- Lenarz, M. S., M. E. Nelson, M. W. Schrage, and A. J. Edwards. 2009. Temperature mediated moose survival in northeastern Minnesota. *Journal of Wildlife Management* 73:503-510.
- Lenarz, M. S., J. Fieberg, M. W. Schrage, and A. J. Edwards. 2010. Living on the edge: viability of moose in northeastern Minnesota. *Journal of Wildlife Management* 74:1013-1023.
- Livezey, K. B. 1990. Toward the reduction of marking-induced abandonment of newborn ungulates. *Wildlife Society Bulletin* 18:193-203.
- McGraw, A. M., J. Terry, and R. Moen. 2014. Pre-parturition movement patterns and birth-site characteristics of moose in northeastern Minnesota. *Alces*. *In press*.
- Minnesota Department of Natural Resources (MNDNR). 2007. Ecological Classification System. Minnesota Department of Natural Resources, St. Paul, USA. <http://www.dnr.state.mn.us/ecs/index.html>.
- Minnesota Department of Natural Resources (MNDNR). 2011. Minnesota moose research and management plan. Division of Fish and Wildlife, St. Paul, Minnesota, USA. 52pp.
- Murray, D. L., E. W. Cox, W. B. Ballard, H. A. Whitlaw, M. S. Lenarz, T. W. Custer, T. Barnett, and T. K. Fuller. 2006. Pathogens, nutritional deficiency, and climate influences on a declining moose population. *Wildlife Monographs* No. 166.
- Patterson, B. R., J. F. Benson, K. Middel, K. J. Mills, M. E. Obbard. 2013. Moose calf mortality in central Ontario, Canada. *Journal of Wildlife Management* 77:832-841.
- Severud, W. J., G. D. DelGiudice, and R. G. Wright. 2013. Evaluating the use of GPS collars to determine moose calving and calf mortalities in northeastern Minnesota. Pages 52-62 *in* L. Cornicelli, M. Carstensen, M. D. Grund, M. A. Larson, and J. S. Lawrence, editors. Summaries of wildlife research findings, 2012. Minnesota Department of Natural Resources, Forest Wildlife Populations and Research Unit, St. Paul, Minnesota, USA.

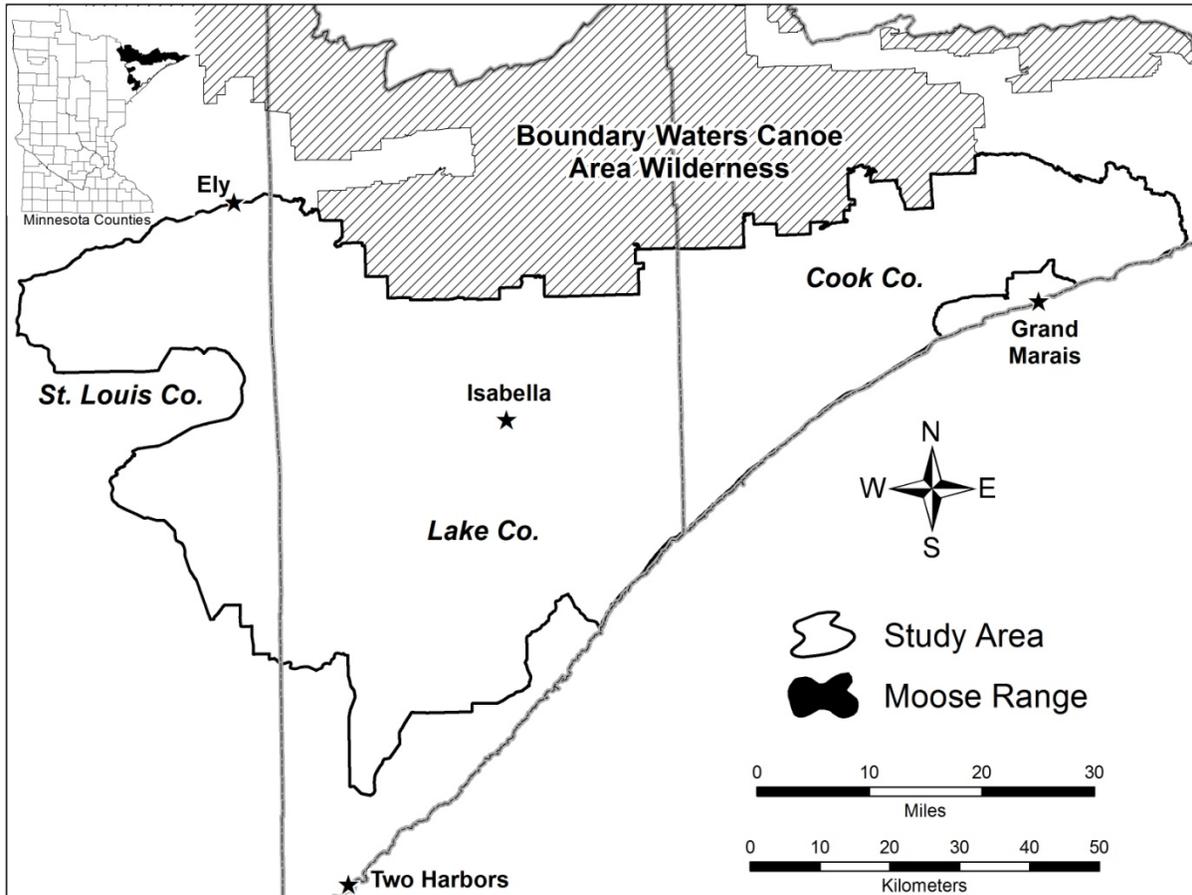


Figure 1. Study area for the study of moose calf survival and cause-specific mortality, northeastern Minnesota, 2013-2017.

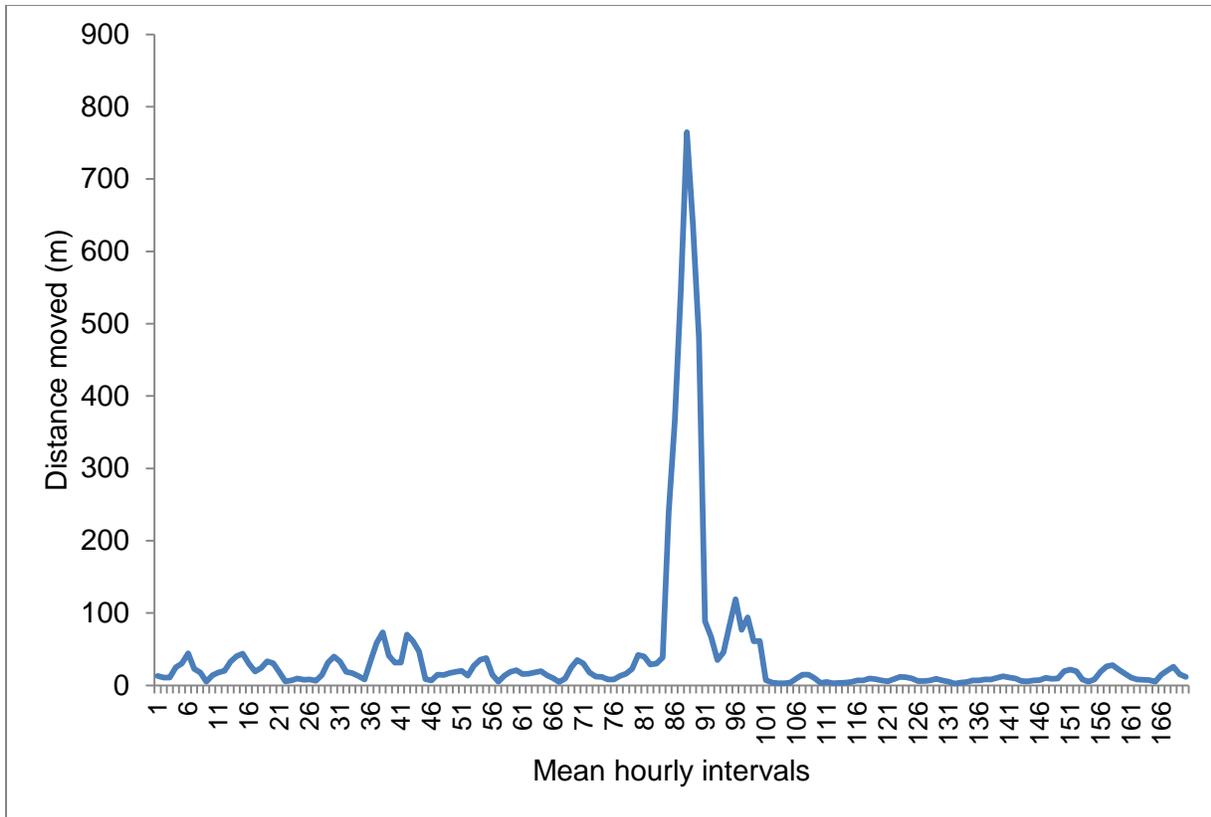


Figure 2. An example of calculated mean hourly distances moved by a pregnant, adult female moose (No. 12500) from 12:04 am, 1 May to 1:42 pm, 8 May 2013. The elevated peak at hour 88 represents the dam’s primary “pre-calving move” (about 800 m), but she didn’t localize completely until after hour 97. We used the latter as indicative of calving so as not to over-estimate bonding time, which was measured during the interval between then and capture time (hour 172). The pre-calving move was used for monitoring calving activity in northeastern Minnesota and for helicopter-assisted capturing of neonates during 8-17 May 2013 and for all ground captures during 8 May-19 June 2014.

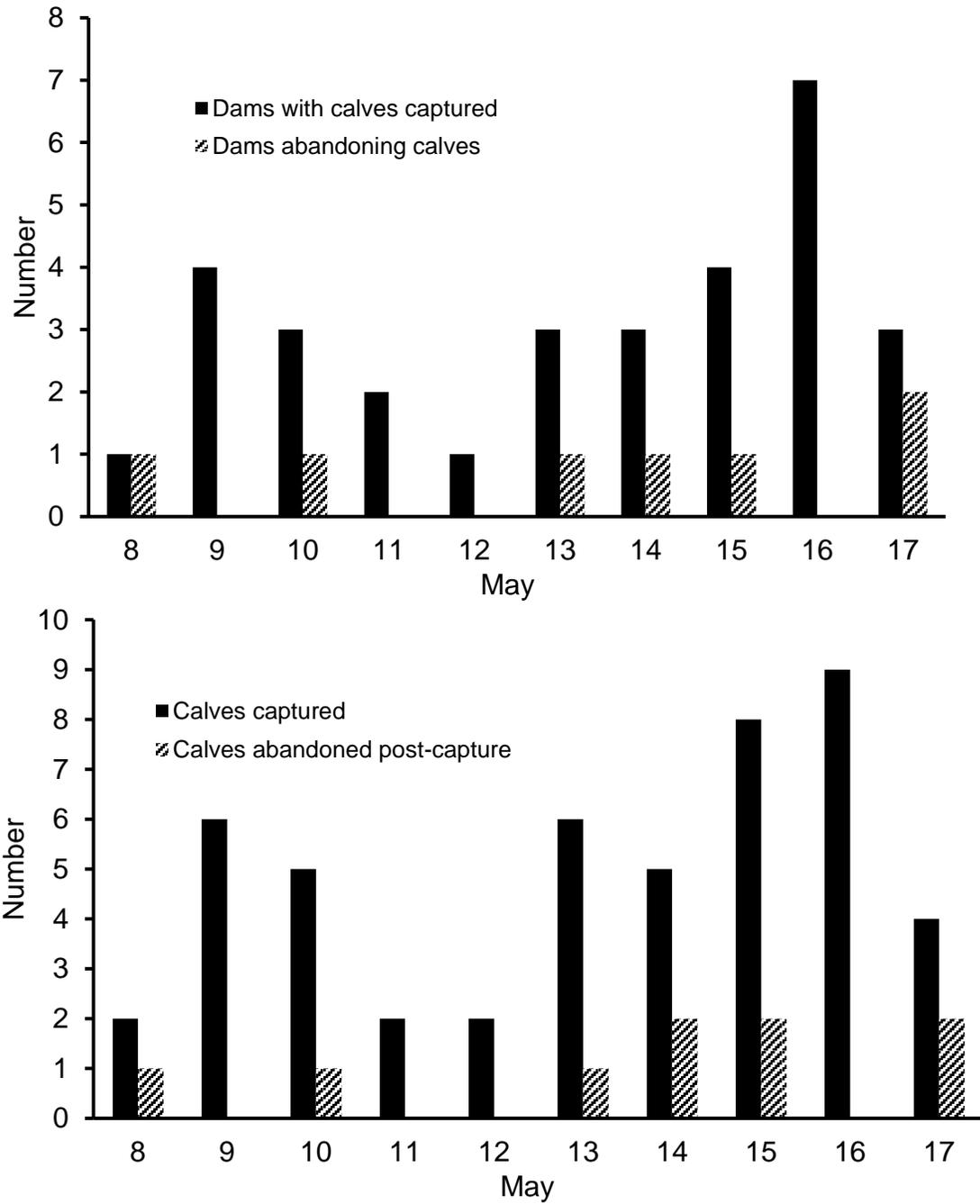


Figure 3. Temporal distributions of the total number of moose dams whose calves were captured, handled, and released compared to those that abandoned their calf or calves post-capture (top), and the total number of moose calves captured, handled, and released compared to those that were abandoned (bottom) apparently in response to capture operations, northeastern Minnesota, 8-17 May 2013.

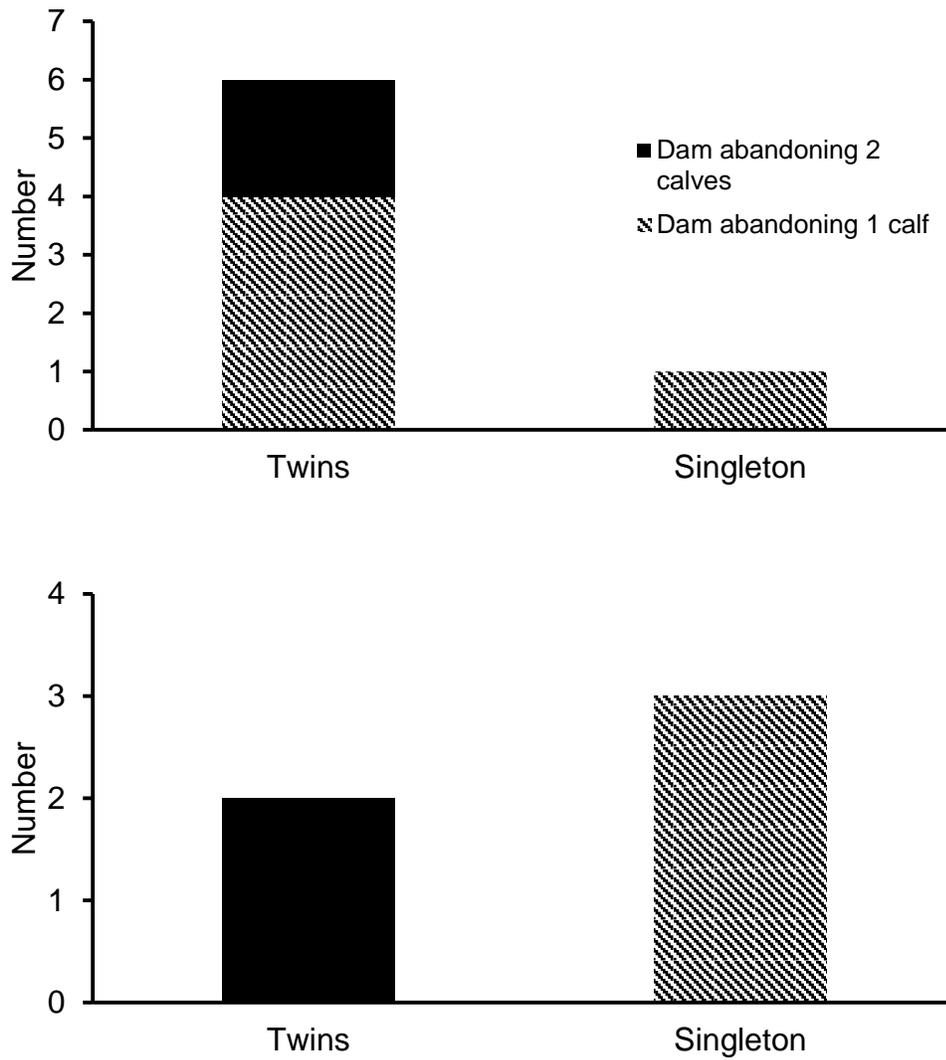


Figure 4. Comparison of dams abandoning twins versus singletons in response to helicopter-assisted captures (8-17 May 2013, top) versus ground captures not assisted by helicopters (8 May-19 June 2014, bottom), northeastern Minnesota.

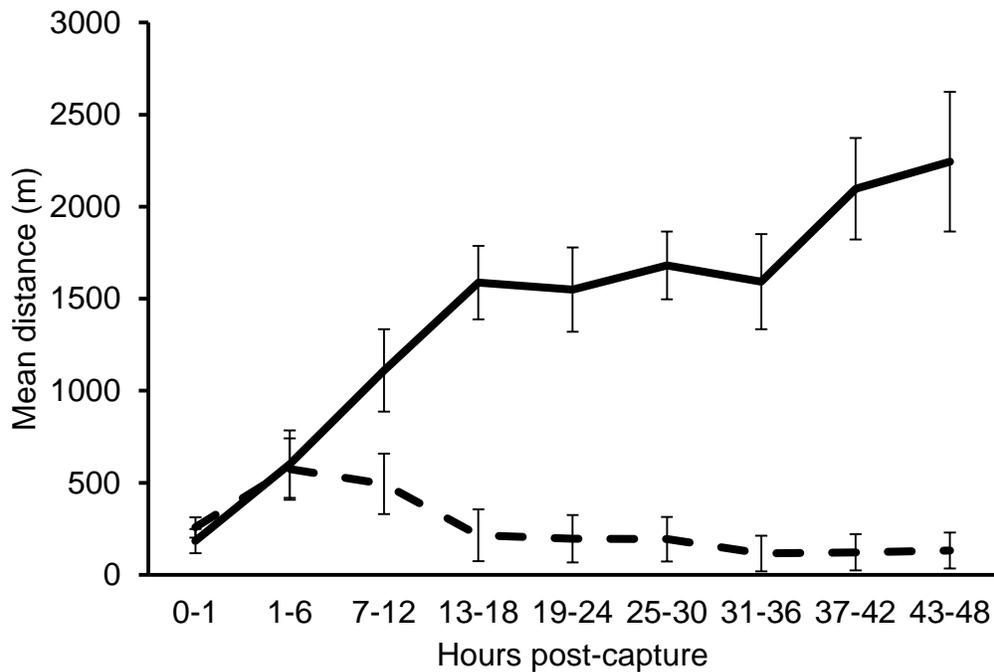
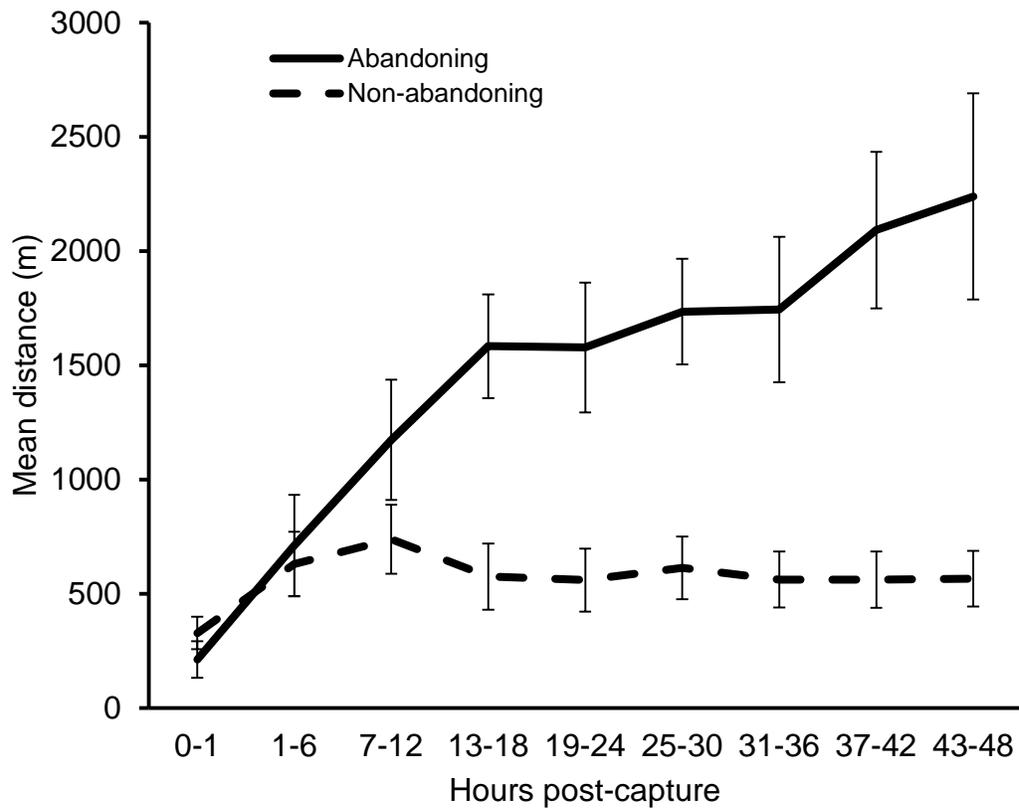


Figure 5. Mean (\pm SE) distance (m) of abandoning and non-abandoning moose dams from the capture site of their newborn calves (top) and from the calves (bottom, 2-4 days old) during 6-hour intervals up to 48 hours post-capture, northeastern Minnesota, 8-17 May 2013.

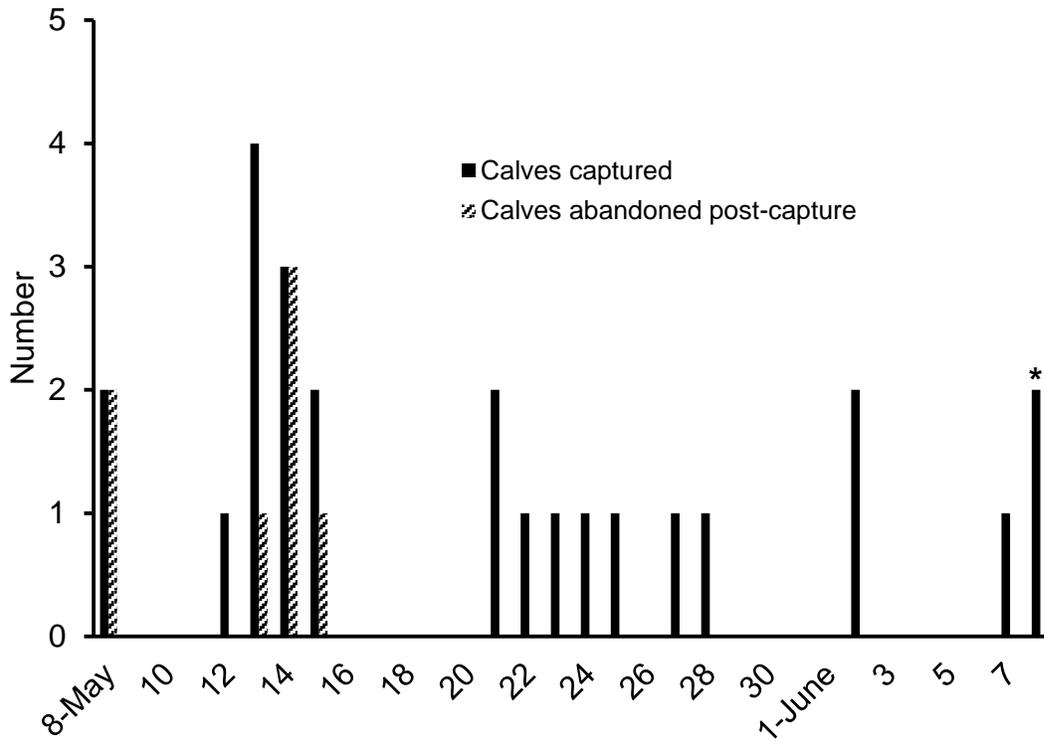
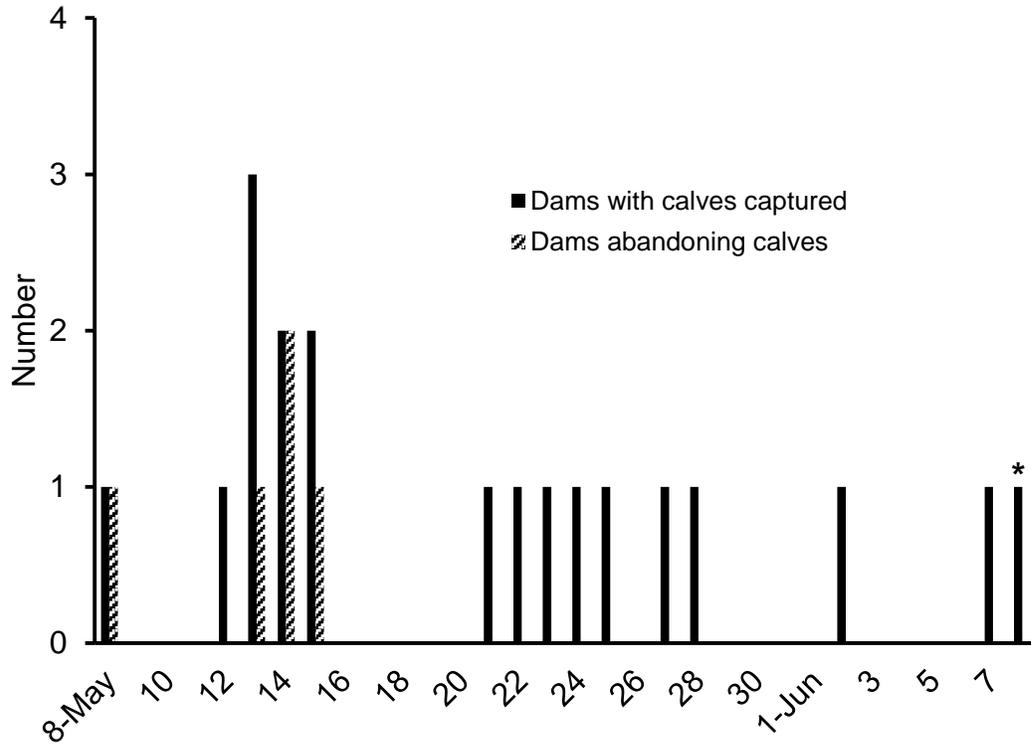


Figure 6. (top), and the total number of moose calves captured, handled, and released compared to those that were abandoned (bottom) apparently in response to capture operations, northeastern Minnesota, 8 May-19 (*) June 2014.



USING GPS COLLARS TO DETERMINE MOOSE CALVING AND CAUSE-SPECIFIC MORTALITY OF CALVES IN NORTHEASTERN MINNESOTA: PROGRESS REPORT ON SECOND FIELD SEASON

William J. Severud¹, Glenn D. DelGiudice, Tyler R. Obermoller, Katie J. Foshay, and Robert G. Wright²

SUMMARY OF FINDINGS

Adult survival is an important driver of large herbivore population dynamics; however, low and variable recruitment also can have a strong influence on population trajectory. The northeastern Minnesota moose (*Alces alces*) population has been exhibiting a downward trend since 2006. Our research was initiated because neonatal and seasonal survival rates and specific causes of mortality (e.g., predation, undernutrition, disease) of calves are largely unknown. Beginning 1 May 2014 we monitored 70 adult female moose fitted with global positioning system (GPS) collars (16 confirmed pregnant at capture during February 2014 by progesterone concentrations, 50 unknown, 4 not pregnant) for long-distance pre-calving movements followed by localization. Our ground crew captured calves from 19 of the 56 cows which had made the pre-calving move then localized. Of these 56 cows, 13 were of the 16 confirmed pregnant and 43 collared in 2013 were of unknown pregnancy status. Twenty-five neonates from 19 dams (32% twinning rate) were fitted with expandable GPS collars during May and June 2014 for subsequent monitoring during their first year. We are retrieving collars from calf mortalities and estimating proximate causes of mortality on site. Mean elapsed time between estimated time of death and mortality investigation ranged from 21.5 to 62 hours; accessibility and functionality of individual collars were influential factors. Seven mortalities (6 natural, 1 capture-related) have occurred, and 7 collars were slipped during 16 May-22 June; 5 calves remained “on air” as of the latter date. After censoring the 7 slipped collars and 7 capture-related abandonments (6 retrieved and placed in zoos, 1 mortality), 6 of 11 calves have died (55%) naturally compared to 17 of 34 calves (50%) during this time period in 2013. A natural abandonment possibly due to umbilical infection, 1 predation by black bear (*Ursus americanus*) and 4 by wolves (*Canis lupus*) are preliminary causes of death. Identifying specific causes of calf mortality and understanding their relations to various landscape and other extrinsic factors should yield insight into mechanisms contributing to the declining moose population in northeastern Minnesota and serve as a basis for an ecologically-sound management response.

INTRODUCTION

The moose is an iconic species of northern Minnesota, which has afforded valuable hunting and viewing opportunities (Minnesota Department of Natural Resources [MNDNR] 2012). The MNDNR has listed moose as a Species of Special Concern (http://files.dnr.state.mn.us/natural_resources/ets/endlist.pdf). Recently, the northwestern population declined precipitously to less than 100 moose due to a variety of natural factors (Murray et al. 2006). The northeastern population is in decline and is experiencing adult mortality rates similar to those of the northwestern population as it decreased (Lenarz et al. 2009, 2010; Butler et al. 2013; Carstensen et al. this issue).

Large herbivore population growth (λ) is most sensitive to variation in adult survival (Gaillard et al. 1998, 2000; Lenarz et al. 2010). Juvenile survival has less of an impact on overall

¹ Graduate Research Assistant, University of Minnesota, Department of Fisheries, Wildlife, and Conservation Biology, 2003 Upper Buford Circle, Ste. 135, St. Paul, MN 55108, 2Wildlife GIS Specialist, Minnesota Information Technology Services @ Department of Natural Resources, Section of Wildlife, 5463-C West Broadway, Forest Lake, MN 55025.

population growth, but differences in temporal variation of juvenile survival may be important in accounting for between-year variation in λ (Gaillard et al. 2000). Fecundity and calf survival ultimately determine recruitment rates which are important to more fully understanding population dynamics (Van Ballenberghe and Ballard 2007). When viable populations of predators are present, predation can be a primary cause of mortality of temperate ungulate neonates (Linnell et al. 1995, Carstensen et al. 2009). Less is known about other specific ultimate or proximate sources of moose calf mortality or factors which may influence predation and other sources of mortality. It also is unclear when predation is compensatory or additive to other sources of mortality (Franzmann et al. 1980, Linnell et al. 1995), although a recent study documented additive effects of predation on moose calves in Alaska (Keech et al. 2011). The degree of predation's impact on calf survival depends on the extant predator guild and relative densities of predator and prey (Eriksen et al. 2011, Patterson et al. 2013).

Particularly after the calves' first summer, the magnitude of mortality from wolves (*Canis lupus*) is variable (Patterson et al. 2013). Wolves are more adept at killing calves in deep snow (DelGiudice et al. 2009), but wolves in an Alaskan study also were responsible for calf mortalities in fall (Keech et al. 2011). Typically, bear-caused (*Ursus* spp.) mortality of calves is greatest closer to their parturition, more immediately following emergence of bears from winter dens (Bastille-Rousseau et al. 2011). Cows in poor nutritional condition may defend calves less vigorously (Patterson et al. 2013). Further, risk of predation is not independent of maternal care and experience (Ozoga and Verme 1986). The importance of natural non-predatory causes of calf mortality likely varies during different times of the year, such as malnutrition and exposure in spring, or malnutrition and tick-related deaths in winter (Patterson et al. 2013). The extent to which diseases drive calf mortality is not well understood, although diseases have led to poor recruitment in moose (O'Hara et al. 2001, Murray et al. 2006). Juvenile animals are more predisposed to parasites than adults, and pathology related to parasite infection may be an important source of mortality for moose calves (Jenkins et al. 2001, Murray et al. 2006). Further, small calves may not be tall enough to efficiently nurse, leading to malnutrition (Murray et al. 2006). Drowning and climate have been known to affect moose calves more than predation in some regions (Crête and Courtois 2009). In winter, temperature and snow depth can be more important causes of mortality than predation (Keech et al. 2011).

Pregnant cow moose tend to move long distances (mean = 6 km) prior to localizing to give birth (McGraw et al. 2014). These distances are typically much longer than movements between foraging and bedding sites. Following a long movement, calving localizations as measured by GPS collars resemble mortality localizations. A cow with calves may stay within a 1.7-ha area for up to 7 days (range = 1-18 days; McGraw et al. 2014).

Until the current study, expandable GPS collars have not been fitted to moose neonates and have only recently been used on other wild ungulate neonates in small numbers (fallow deer [*Dama dama*], $n = 3$, Kjellander et al. 2012) or in captivity (domestic horse [*Equus caballus*], $n = 4$, Hampson et al. 2010). Observable fine-scale movement and habitat use patterns of moose calves, made possible by GPS collars, are facilitating examination of landscape features important for their survival and rapid investigation of mortality events. Having dams and calves fitted with GPS collars also allows us to study the importance of their proximity to juvenile survival.

OBJECTIVES

1. Evaluate monitoring of movement behavior of GPS-collared adult female moose to determine timing and location of calving; and
2. Evaluate remote tracking of GPS-collared calves and dams to determine and investigate calf mortalities and assign cause.

METHODS

Our study area is the same as that of the Environmental and Natural Resources Trust Fund (ENRTF)-supported study focused on survival and cause-specific mortality of adult moose in northeastern Minnesota (see Figure 1, DelGiudice et al. this issue). As part of the adult moose mortality study, 111 (84 females, 27 males) and 37 (25 females, 12 males) moose were captured and fitted with Iridium GPS collars (Vectronic Aerospace, Berlin, Germany) during January 2013 and February 2014, respectively (Butler et al. 2013, Carstensen et al. this issue). Blood was collected and analyzed for pregnancy; ≥ 2.0 ng/mL was the progesterone concentration threshold indicative of pregnancy. We monitored cow movements during pre-parturition and calving, with particular attention afforded to pregnant cows. We looked for movement patterns indicative of calving, including a long-distance movement followed by localization (Bowyer et al. 1999, McGraw et al. 2014).

We began monitoring 70 collared adult female moose (16 confirmed pregnant at capture by progesterone concentrations, 50 unknown [captured and collared in 2013], 4 not pregnant) on 1 May 2014. Based on a 77% pregnancy rate of total tested cows in 2014, we assumed ~55 (39 unknown + 16 known) cows to be pregnant in 2014. Cow collars were programmed to collect hourly locations during May and early-June and to transmit these locations 4 times per day. An automated R program (J. D. Forester, University of Minnesota, unpublished data) generated emailed reports 2 times daily (0500, 1700 hours). Reports contained a file (pdf) displaying various movement and location metrics for each collared cow, and table (csv format) and map (kml format) files with all recent locations of each animal. The pdf reports contained a rough map of northeastern Minnesota with all current cow locations displayed and a summary table of all animal locations and distances moved in the last 24 and 48 hours. The metrics for each cow included the date and time of the last location, movement path of the last 5 days, movement path of the last 24 hours overlaid on Google Earth imagery, a plot showing 3-hour average distances moved, and each cow's data on a single page (Figure 1). The distance plot showed peaks in movements that we then monitored for possible dampening of movements (localization). If the cow moved < 100 m during 36 hours after making a long-distance movement (dam-calf bonding time), the program flagged that cow as "localized," and that cow's calf was determined eligible for capture. Additionally, a blue line representing predictions from a regression based on 2013 calving movements showed if a calving move may have occurred in the last 12 hours. Larger spikes indicate higher likelihood of a calving event. A gray line shows relatively large or small movements over the past 12 hours. When a cow's calf was eligible for capture, we also checked her movement path on the Vectronic Aerospace website (<https://www.vectronic-wildlife.com>; Figure 2). After capture, dams and calves were paired for the automated reports, and an additional metric was added to the movement plot (proximity between dam and calf, Figure 3). This plot was monitored for possible abandonments. Calves also were added to the report and had a page similar to that of the cows displaying their location and movement metrics.

Once a cow localized for > 36 hours, a ground crew searched the potential calving site for the cow and calf or calves (see DelGiudice et al. this issue). Each captured calf was fitted with an expandable Globalstar GPS Calf Collar (520 g; Vectronic Aerospace, Berlin, Germany) and 2 ear-tags, and was weighed (± 0.5 kg). Collars were programmed to take a fix hourly and transmit every third successful fix. All fixes are stored on board the collar. After observing several calves with abrasions on the dorsal side of their necks in winter 2013-2014, we worked with Vectronic Aerospace to redesign the collar belting to be wider, softer, longer, and to include a sleeve to preclude abrasion. Twins each received a collar and ear-tags. As feasible relative to the dam's behavior, the crew also took morphometric measurements (neck circumference, girth, total body length, hind leg length), collected blood and hair, and measured a rectal temperature. As captures progressed, handling was reduced to simply fitting the collar and determining sex (DelGiudice et al. this issue). All captures and handling protocols followed requirements of the Institutional Animal Care and Use Committee for the University of

Minnesota (Protocol 1302-30328A) and were consistent with guidelines recommended by the American Society of Mammalogists (Sikes et al. 2011).

We monitored each collared calf daily until mortality or until its collar drops off (designed to be ~400 days). We relied upon the collars to send mortality alert notifications to cell phones via text message (i.e., SMS) when mortalities occurred, but after several mortalities went unnoticed last season (Severud et al. 2013), we began using the Vectronic Aerospace website and GPS Plus X software to check if calf collars were far from dam collars or in mortality mode. Each morning all dam and calf groups are checked and monitored closely throughout the day if separated by >500 m.

When we receive a mortality alert or determine a mortality may have occurred, we dispatch an investigative team (goal of within 24 hours of death) to collect the collar and carcass remains and to determine the cause of death (Ballard et al. 1979). To avoid possible investigation-induced abandonment, investigations are delayed if the dam is still in the area, especially if she is with a twin. Our primary field objective is to recover the entire carcass and deliver it to the University of Minnesota's Veterinary Diagnostics Laboratory (VDL) for necropsy. If the carcass cannot be extracted and transported, we perform a detailed field necropsy. If scavenged or fed upon, fresh organ and tissue samples are collected and shipped or transported to the VDL as feasible (Butler et al. 2011). Care is taken to haze off predators and scavengers when approaching a potential mortality site; bear repellent spray and firearms are available as a last resort for protection, but their use is not necessarily anticipated (Smith et al. 2008, 2012). We postpone the investigation when predators are sighted on the carcass; return is dependent on the age and size of the carcass as an indication of how long the predator or scavenger may feed.

Once we begin a thorough investigation of the site, we are careful not to disturb potential evidence. We photograph tracks and scat and collect scat when identification is uncertain. We note the presence of puncture wounds on the neck, skull, or hind quarters and claw marks across the body and take photographs of all wounds. When the hide is present, we note if it is inverted, which may indicate a bear was feeding on the carcass. We document the consumption of viscera, the rumen or its contents. Wolves may chew on ribs and ends of long bones, whereas bears are more likely to cache pieces of the carcass. To determine if injuries occurred ante mortem, we look for subcutaneous hemorrhaging or sprays of blood on the collar or surrounding vegetation. Signs of a struggle (broken or matted vegetation, disturbed ground) also are indications of predation. We take note of the position of the carcass (lateral or sternal), and the distribution of body parts (scattered or near the carcass). An odor of decomposition or many fecal pellets in the area may indicate scavenging versus predation.

If we find a GPS collar without a carcass or other evidence of predation, we backtrack to the last known locations of the calf and its dam to examine a larger area in an expanded search. The adult Iridium collars are more accurate than the calf Globalstar collars (W. J. Severud, University of Minnesota, unpublished data), so we use the cow's locations from the approximate time of death of the calf to look for a kill-site or evidence of the cause of mortality. We conclude a collar has been slipped rather than indicating a mortality if the breakaway section is frayed or the bolts holding the breakaway section are loose, coupled with both an absence of blood on the collar and lack of evidence within a 30-m radius of the collar. Additionally, the dam's behavior, large movements away or localization around the estimated time of death, can be used as an indication predation or a slipped collar, respectively.

RESULTS

We observed 56 of 70 cows (80%) display calving movement behavior (pre-calving movement followed by localization for >36 hours). Mean localization date was 19 May 2014 (median = 18 May, range = 5 May-16 June; Figure 4), with 75% of the localizations occurring during 11-22 May 2014.

We deployed 25 expandable GPS collars on neonates (11 females, 14 males) captured from 19 dams (32% twinning rate) during 8 May-19 June 2014 (Figure 5; see DelGiudice et al.

this issue for additional details). Twin sex composition was 3 male/female, 2 male/male, and 1 female/female. Of the 19 dams, 4 were confirmed pregnant by progesterone and 15 were unknown. Once we deployed 2 collars on a set of twins, we ceased capture operations due to abandonment concerns possibly resulting from late deep snow cover (DelGiudice et al. this issue). Once conditions improved, we began collaring again, but ceased operations when abandonments continued. We then decided to reduce handling to solely fitting the collar and determining sex. We observed no capture-related abandonment after this final modification of our methods and collars (DelGiudice et al. this issue).

We weighed, measured hind leg length, neck circumferences (upper and lower), and took temperature for 12 calves (6 F, 6 M, 6 singletons, 3 sets of twins) during the first phase of captures (8-15 May 2014). We measured chest girth for 2 calves (1 F, 1 M, 1 singleton, 1 twin), and measured total body length of 1 calf (M, twin; Table 1). Mean rectal temperature was 101.4 °F (SE = 0.7, range = 97.7-107, $n = 12$).

We visited 5 cows (2 pregnant, 3 unknown) which exhibited movement patterns indicative of calving, yet no calves were found. We often found evidence of a calving site (bare ground, many pellets and beds) or calves (pellets, tracks), but could not locate the calves. We limited search time to ~5 minutes to lessen disturbance. Often the cows returned to these sites after we departed, indicating a calf or calves may have been present, but undetected by our capture team.

As of 16 June 2014, we have documented 6 natural mortalities (Figure 6) and 7 slipped collars; 5 collared calves remain “on air.” Capture-related activities accounted for 6 abandonments and 1 mortality associated with capture-related abandonment (DelGiudice et al. this issue). Of the remaining 6 mortalities, there was 1 natural abandonment associated with an umbilical infection, 1 bear-kill, and 4 wolf-kills. Histological and disease-screening results from the VDL are pending. After censoring the capture-related abandonments and slipped collars, 6 of 11 calves have died (55%) as of 22 June 2014, with 5 of those preyed upon by wolves or bears (Figure 7).

Of the 6 mortalities we investigated on site, 1 of the collars failed to send a mortality alert text message. We investigated the site because the collar was not transmitting and the dam had departed from the last known calf location. This collar was found in a shallow puddle and never transmitted a mortality message to the satellite base station (and stopped collecting and sending GPS fixes).

Mean elapsed time between estimated time of death (when the collar entered mortality mode) and mortality investigation was 46 hours (range = 0-239 hours, $n = 17$). A collar that was submerged in a shallow puddle did not collect locations or transmit for 7 days and took 158 hours to investigate. The mean response-time was 62 hours (range = 9.5-239 hours, $n = 10$) when we received a mortality alert text message, although in 7 of these cases the collar was slipped (mean = 85, range = 13-239 hours). For natural mortalities (omitting slipped collars), mean response time was 21.5 hours (range = 9.5-52.5, $n = 6$).

All collared dams were tracked for calving movements from 1 May to 20 June. We found 56 cows made calving movements and localizations indicative of calving. We documented several mortalities of uncollared calves which were incidental to mortalities of collared adult moose (Carstensen et al. this issue). One adult was killed by wolves and we found 2 partial calf skulls. Calving movements suggested that this cow calved and then she and her twins were killed by wolves. Another cow died of malnutrition and its calf subsequently died within 30 m. In another case, a cow was investigated due to collar malfunction. The investigation revealed cow and calf beds and tracks, but no evidence of predation or scavenging.

DISCUSSION

Tracking GPS-collared cow movements was again this year a highly reliable way to estimate whether or not they had calved. Of the 24 dams suspected of calving and subsequently visited, we collared calves from 19 (79% success rate). We do not know for certain whether the 5 dams visited without collared calves had given birth, but site evidence and dam behavior often

suggested that dams had calved. In one case we visited a dam that was later determined to not have localized once more GPS fixes were transmitted. Our study objective was to fit GPS collars to 50 newborns over the entire calving period (May to early-June); we fit 25 collars over 13 days of captures (several periods of inactivity due to method refinement; DelGiudice et al. this issue). We decided to track cows during May and June to look for movement patterns indicative of calving rather than fitting vaginal implant transmitters (VITs) to pregnant cows for several reasons. Fitting VITs would have required determining pregnancy status during winter captures, which would have added significant expense and time to the handling of the captured adult females, and most of the cows we monitored for calving were collared the previous year. Monitoring known pregnant cows for a “calving move” did not limit us to only those 16 pregnant females which would have been fitted with a VIT; the latter also would have required the expense of monitoring from a fixed-wing aircraft. Monitoring calving movements has been invaluable as we plan to capture calves from collared cows that we will not need to recapture during winter to determine pregnancy.

We observed and handled 6 sets of twins throughout calving, as opposed to last year when most twins were observed and captured at the beginning of the calving window (Figure 5). To more accurately represent the northeastern Minnesota population we attempted to spread our capture efforts throughout the calving season. In a previous study in northeastern Minnesota, the mean calving date was 14 May 2011 (range = 3-27 May), with 70% of births occurring during 9-20 May (McGraw et al. 2014). In 2013, we observed the mean date a cow localized (assumed to have calved within 12 hours) was 12 May (median = 13 May, range = 5-16 May), while in 2014 mean localization date was 19 May (median = 18 May, range = 5 May-16 June). The later calving dates in 2014 may be a result of the severe winter of 2013-2014 or of more intense monitoring into June 2014.

We have documented 7 mortalities to date, 6 of them due to natural causes (55% natural mortality rate). One resulted from capture-related abandonment (DelGiudice et al. this issue). Another calf died of a severe umbilical infection. Its dam was spending time between the sick calf and its healthy twin, but ultimately left the sick calf and remained with the healthy calf. Wolf predation claimed 4 calves, and a bear killed 1 calf. During 5 May to 8 June 2013, we documented 17 natural mortalities (50% mortality rate); causes included drowning, natural abandonment, and predator-kills. During May 2013-February 2014, we documented 25 natural mortalities of 34 collared calves (74% mortality, Figure 8). Predation accounted for 22 of these mortalities (88%, 4 bear-kills, 18 wolf-kills).

To date we have had 7 collars slip off in 2014. In each instance the collar was stretched and the sewn expansion loops broke prematurely. There was no tearing or blood on the collars or sign of a struggle at the collar location. Additionally, the dam did not flee the site at the time the collar entered mortality mode. This is likely a design flaw that will need to be addressed before next year’s captures. We are currently testing the resilience of the collar banding.

Body mass measurements from calves in northeastern Minnesota (mean = 16.2 kg, SE = 0.5) agree with data from healthy moose calves throughout North American moose range (13-18 kg; Jensen et al. 2013). Measurements taken in 2013 and 2014 are comparable, although the 2014 sample size was limited ($n = 43$ versus 12). We did not detect any sexual dimorphism in calf measurements, but singletons tended to be heavier than twin calves, as reported elsewhere. However, northeastern Minnesota calves tended to be heavier than Alaskan calves (Schwartz and Hundertmark 1993).

ACKNOWLEDGMENTS

We would like to thank J. Forester, T. Kreeger, volunteers L. Ross and M. Haas, the adult moose mortality study team (M. Carstensen, E. Hildebrand, D. Pauly, M. Dexter, N. Hansen, J. Giudice, D. Ingebritsen, and D. Plattner), A. Wunschmann, A. Armien, J. Rasmussen, A. McGraw, J. Terry, and R. Moen. This project was funded in part by the Wildlife Restoration (Pittman-Robertson) Program.

LITERATURE CITED

- Ballard, W. B., A. W. Franzmann, K. P. Taylor, T. H. Spraker, C. C. Schwartz, and R. O. Peterson. 1979. Comparison of techniques utilized to determine moose calf mortality in Alaska. *Alces* 15:362–387.
- Van Ballenberghe, V., and W. B. Ballard. 2007. Population dynamics. Pages 223–245 in A. W. Franzmann and Schwartz, Charles C., editors. *Ecology and Management of the North American Moose*. Second Edition. University Press of Colorado, Boulder.
- Bastille-Rousseau, G., D. Fortin, C. Dussault, R. Courtois, and J. P. Ouellet. 2011. Foraging strategies by omnivores: are black bears actively searching for ungulate neonates or are they simply opportunistic predators? *Ecography* 34:588–596.
- Bowyer, R. T., V. Van Ballenberghe, J. G. Kie, and J. A. K. Maier. 1999. Birth-site selection by Alaskan moose: maternal strategies for coping with a risky environment. *Journal of Mammalogy* 80:1070–1083.
- Butler, E., M. Carstensen, E. Hildebrand, and D. Pauly. 2013. Determining causes of death in Minnesota's declining moose population: A progress report. Pages 97–105 in L. Cornicelli, M. Carstensen, M. D. Grund, M. A. Larson, and J. S. Lawrence, editors. *Summaries of Wildlife Research Findings 2012*. Minnesota Department of Natural Resources, St. Paul, USA.
- Carstensen, M., G. D. Delgiudice, B. A. Sampson, and D. W. Kuehn. 2009. Survival, birth characteristics, and cause-specific mortality of white-tailed deer neonates. *Journal of Wildlife Management* 73:175–183.
- Crête, M., and R. Courtois. 2009. Limiting factors might obscure population regulation of moose (*Cervidae: Alces alces*) in unproductive boreal forests. *Journal of Zoology* 242:765–781.
- DelGiudice, G. D., K. R. McCaffery, D. E. Beyer, and M. E. Nelson. 2009. Prey of Wolves in the Great Lakes Region. Pages 155–173 in A. P. Wydeven, T. R. Deelen, and E. J. Heske, editors. *Recovery of Gray Wolves in the Great Lakes Region of the United States*. Springer New York, New York, NY.
- Eriksen, A., P. Wabakken, B. Zimmermann, H. P. Andreassen, J. M. Arnemo, H. Gundersen, O. Liberg, J. Linnell, J. M. Milner, H. C. Pedersen, H. Sand, E. J. Solberg, and T. Storaas. 2011. Activity patterns of predator and prey: a simultaneous study of GPS-collared wolves and moose. *Animal Behaviour* 81:423–431.
- Franzmann, A. W., C. C. Schwartz, and R. O. Peterson. 1980. Moose calf mortality in summer on the Kenai Peninsula, Alaska. *Journal of Wildlife Management* 44:764–768.
- Gaillard, J. M., M. Festa-Bianchet, N. G. Yoccoz, A. Loison, and C. Toigo. 2000. Temporal variation in fitness components and population dynamics of large herbivores. *Annual Review of Ecology and Systematics* 31:367–393.
- Gaillard, J.-M., M. Festa-Bianchet, and N. G. Yoccoz. 1998. Population dynamics of large herbivores: variable recruitment with constant adult survival. *Trends in Ecology and Evolution* 13:58–63.
- Hampson, B., J. Morton, P. Mills, M. Trotter, D. Lamb, and C. Pollitt. 2010. Monitoring distances travelled by horses using GPS tracking collars. *Australian Veterinary Journal* 88:176–181.
- Jenkins, A., K. Handeland, S. Stuen, L. Schouls, I. van de Pol, R. T. Meen, and B. E. Kristiansen. 2001. Ehrlichiosis in a moose calf in Norway. *Journal of Wildlife Diseases* 37:201–203.
- Jensen, W. F., J. R. Smith, J. J. Maskey Jr, J. V. McKenzie, and R. E. Johnson. 2013. Mass, morphology, and growth rates of moose in North Dakota. *Alces* 49:1–15.
- Keech, M. A., M. S. Lindberg, R. D. Boertje, P. Valkenburg, B. D. Taras, T. A. Boudreau, and K. B. Beckmen. 2011. Effects of predator treatments, individual traits, and environment on moose survival in Alaska. *Journal of Wildlife Management* 75:1361–1380.

- Kjellander, P., I. Svartholm, U. A. Bergvall, and A. Jarnemo. 2012. Habitat use, bed-site selection and mortality rate in neonate fallow deer *Dama dama*. *Wildlife Biology* 18:280–291.
- Lenarz, M. S., J. Fieberg, M. W. Schrage, and A. J. Edwards. 2010. Living on the edge: Viability of moose in northeastern Minnesota. *Journal of Wildlife Management* 74:1013–1023.
- Lenarz, M. S., M. E. Nelson, M. W. Schrage, and A. J. Edwards. 2009. Temperature mediated moose survival in northeastern Minnesota. *Journal of Wildlife Management* 73:503–510.
- Linnell, J. D. C., R. Aanes, and R. Andersen. 1995. Who killed Bambi? The role of predation in the neonatal mortality of temperate ungulates. *Wildlife Biology* 1:209–223.
- McGraw, A. M., J. Terry, and R. A. Moen. 2014. Pre-parturition movement patterns and birth site characteristics of moose in northeast Minnesota. *Alces* 50:in press.
- Murray, D. L., E. W. Cox, W. B. Ballard, H. A. Whitlaw, M. S. Lenarz, T. W. Custer, T. Barnett, and T. K. Fuller. 2006. Pathogens, nutritional deficiency, and climate influences on a declining moose population. *Wildlife Monographs* 166:1–30.
- O'Hara, T. M., G. Carroll, P. Barboza, K. Mueller, J. Blake, V. Woshner, and C. Willetto. 2001. Mineral and heavy metal status as related to a mortality event and poor recruitment in a moose population in Alaska. *Journal of Wildlife Diseases* 37:509–522.
- Ozoga, J. J., and L. J. Verme. 1986. Relation of maternal age to fawn-rearing success in white-tailed deer. *Journal of Wildlife Management* 50:480–486.
- Patterson, B. R., J. F. Benson, K. R. Middel, K. J. Mills, A. Silver, and M. E. Obbard. 2013. Moose calf mortality in central Ontario, Canada. *Journal of Wildlife Management* 77:832–841.
- Schwartz, C. C., and K. J. Hundertmark. 1993. Reproductive characteristics of Alaskan moose. *Journal of Wildlife Management* 57:454.
- Severud, W. J., G. D. DelGiudice, and R. G. Wright. 2013. Evaluating the use of GPS-collars to determine moose calving and calf mortalities in northeastern Minnesota. Pages 52–62 in L. Cornicelli, M. Carstensen, M. D. Grund, M. A. Larson, and J. S. Lawrence, editors. *Summaries of Wildlife Research Findings 2012*. Minnesota Department of Natural Resources, St. Paul, USA.
- Sikes, R. S., W. L. Gannon, and The Animal Care and Use Committee of the American Society of Mammalogists. 2011. Guidelines of the American Society of Mammalogists for the use of wild mammals in research. *Journal of Mammalogy* 92:235–253.
- Smith, T. S., S. Herrero, T. D. Debruyne, and J. M. Wilder. 2008. Efficacy of bear deterrent spray in Alaska. *Journal of Wildlife Management* 72:640–645.
- Smith, T. S., S. Herrero, C. S. Layton, R. T. Larsen, and K. R. Johnson. 2012. Efficacy of firearms for bear deterrence in Alaska. *Journal of Wildlife Management* 76:1021–1027.

Table 1. Morphometric measurements (\pm SE) from neonate moose calves captured and collared in northeastern Minnesota, May 2013 and 2014.

2013	Singleton		Twin		Female		Male		Total	
	Mean	<i>n</i>	mean	<i>n</i>	mean	<i>n</i>	mean	<i>n</i>	mean	<i>n</i>
Morphometric										
Mass (kg)	17.2 \pm 0.6	13	15.6 \pm 0.3	30	16.2 \pm 0.4	22	16.2 \pm 0.5	21	16.0 \pm 0.3	43
Hind leg length (cm)	47.2 \pm 0.4	13	45.8 \pm 0.3	36	46.0 \pm 0.3	25	46.4 \pm 0.3	24	46.2 \pm 0.2	49
Chest girth (cm)	61.4 \pm 1.7	5	59.3 \pm 0.9	6	60.3 \pm 1.3	6	60.2 \pm 0.4	5	60.3 \pm 0.9	11
Lower neck circumference (cm)	32.3 \pm 0.5	5	28.8 \pm 0.6	6	30.1 \pm 0.9	6	30.7 \pm 1.0	5	30.4 \pm 0.7	11
Upper neck circumference (cm)	31.0 \pm 0.9	4	28.3 \pm 1.2	6	29.5 \pm 1.3	6	29.3 \pm 1.3	4	29.4 \pm 0.9	10
Body length (cm)	106.7 \pm 1.4	3	-	-	105.5 \pm 1.5	2	109	1	106.7 \pm 1.4	3

2014	Singleton		Twin		Female		Male		Total	
	Mean	<i>n</i>								
Morphometric										
Mass (kg)	16.8 \pm 0.9	6	15.7 \pm 0.4	6	16.0 \pm 0.8	6	16.4 \pm 0.7	6	16.2 \pm 0.5	12
Hind leg length (cm)	43.6 \pm 0.5	6	43.2 \pm 1.0	5	42.8 \pm 0.7	6	44.2 \pm 0.6	5	43.4 \pm 0.5	11
Chest girth (cm)	51.5	1	65	1	51.5	1	65	1	58.3 \pm 6.8	2
Lower neck circumference (cm)	33.6 \pm 0.8	6	34.5 \pm 0.9	6	33.1 \pm 0.8	6	35.0 \pm 0.7	6	34.0 \pm 0.6	12
Upper neck circumference (cm)	29.0 \pm 0.5	6	30.8 \pm 0.8	6	28.7 \pm 0.5	6	31.1 \pm 0.6	6	29.9 \pm 0.5	12
Body length (cm)	-	-	101.0	1	-	-	101.0	1	101.0	1

57 Collar 12629 U // Localized

Last location: 2014-05-25 02:51:04
 Max 3-h Avg. Speed (24h) = 11.7 m/h
 Collar Temp = 20°C // Mortality Status =
 UTM.X = 650291, UTM.Y = 5291498
 Longitude = -90.99453, Latitude = 47.75932

Stat	T2h	T24h	T48h	Total
Disp.	9	12	4	3407
Path	36	156	314	436741
Speed	9	8	8	62

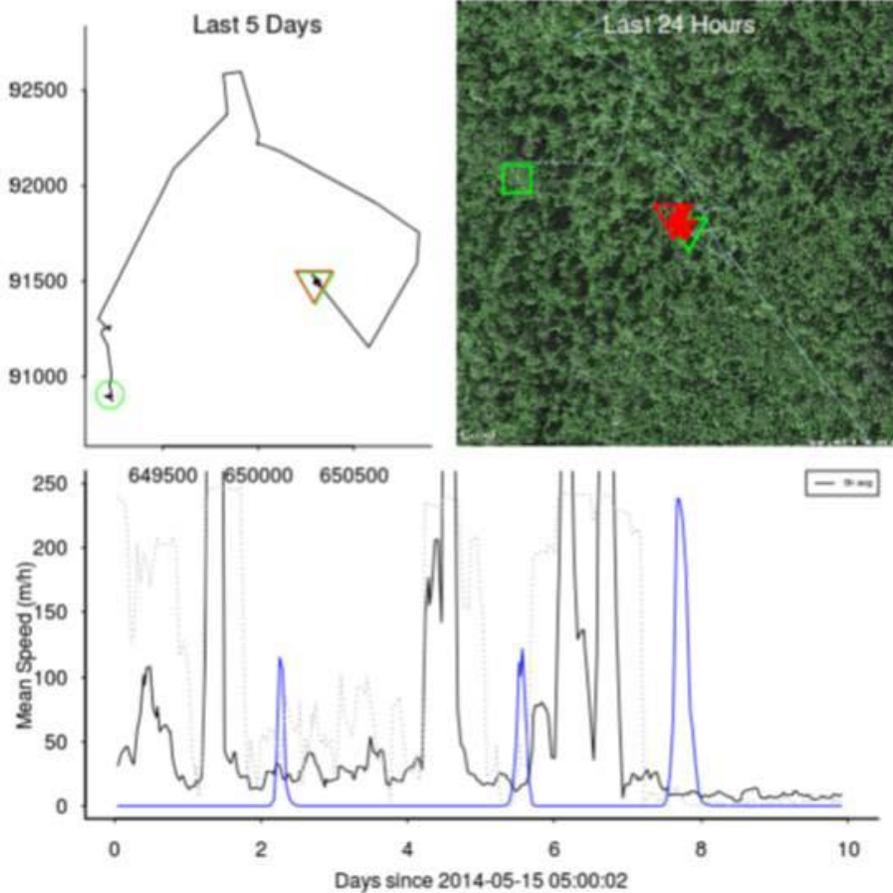


Figure 1. Example report for adult female moose number 12629 from 0500 hours, northeastern Minnesota, 25 May 2014, showing movement paths for the last 5 days and 24 hours, and 3-hour average hourly distances moved. Green circle represents the start of the 5-day period, green triangle the start of the 24-hour period, and red triangle the most recent location. Red dots indicate location when the collar was “localized.” The blue line represents predictions from a regression based on 2013 calving movement data; larger spikes suggest a higher likelihood a calving movement occurred in the past 12 hours. The light gray lines show relatively large or small movements in the past 12 hours. We visited this cow at 10 days since 15 May (25 May).

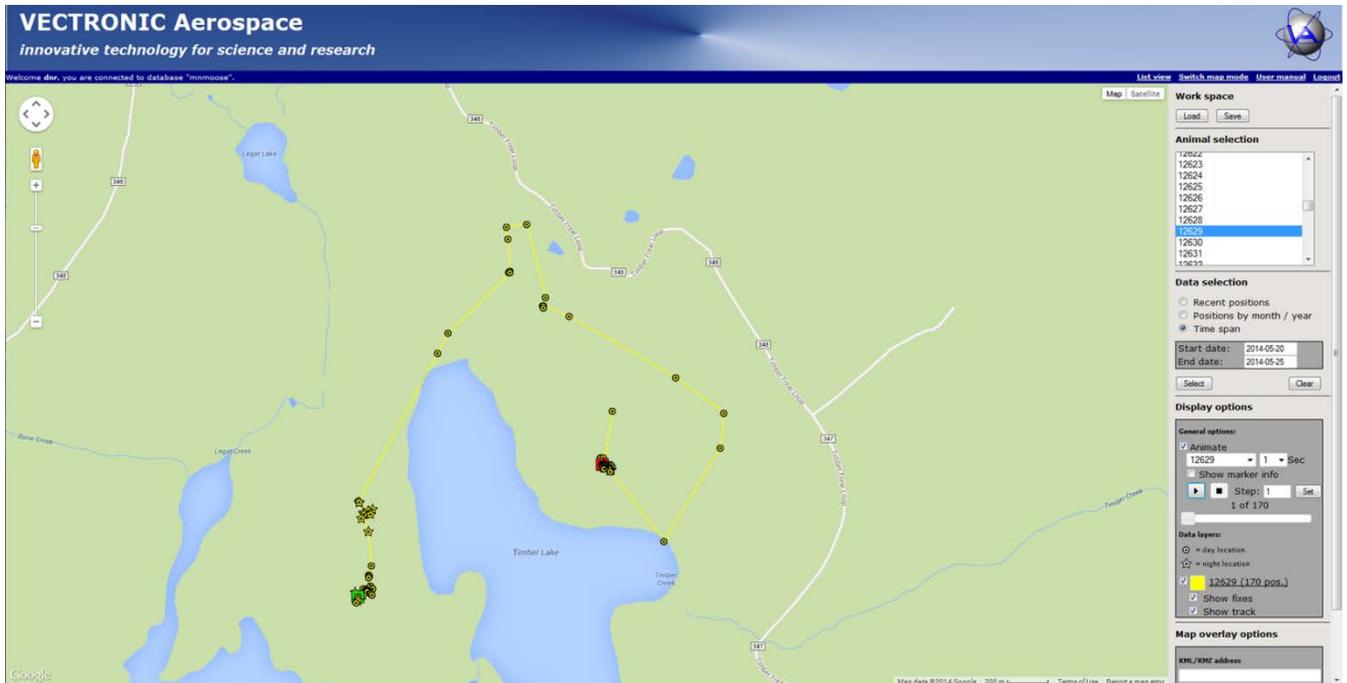


Figure 2. Vectronic Aerospace website (<https://www.vectronic-wildlife.com>) map interface showing the path of adult female moose number 12629, northeastern Minnesota, 20–25 May 2014. The green and red squares represent the start and end of the interval. The cow's movement pattern in the southwestern corner of the map indicates typical bedding and foraging, whereas the cluster in the middle of the map indicates a tight localization which followed a long-distance movement. This cluster is the calving ground and where calf 13253 was collared.

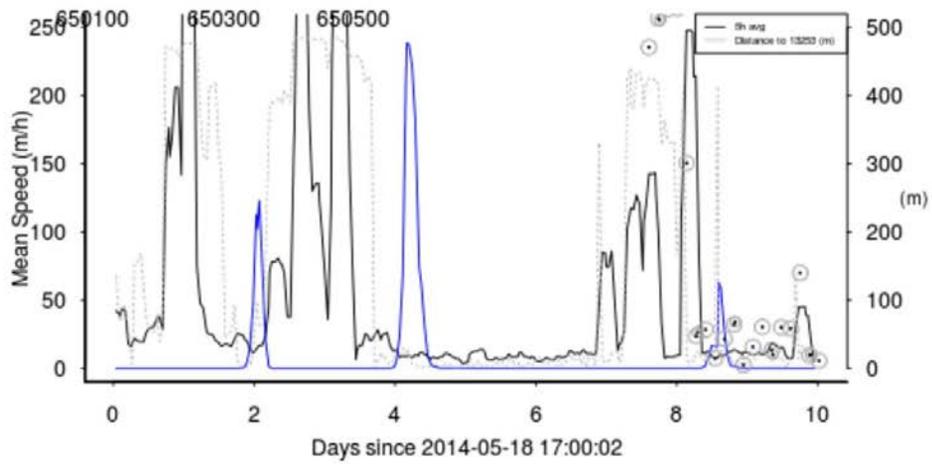


Figure 3. Distance plot displaying both 3-hour average distance moved and proximity of adult female moose number 12629 to calf number 13253, northeastern Minnesota. Black solid line displays the distance the dam has moved; circles with centered dots represent the distance between the dam and calf collar.

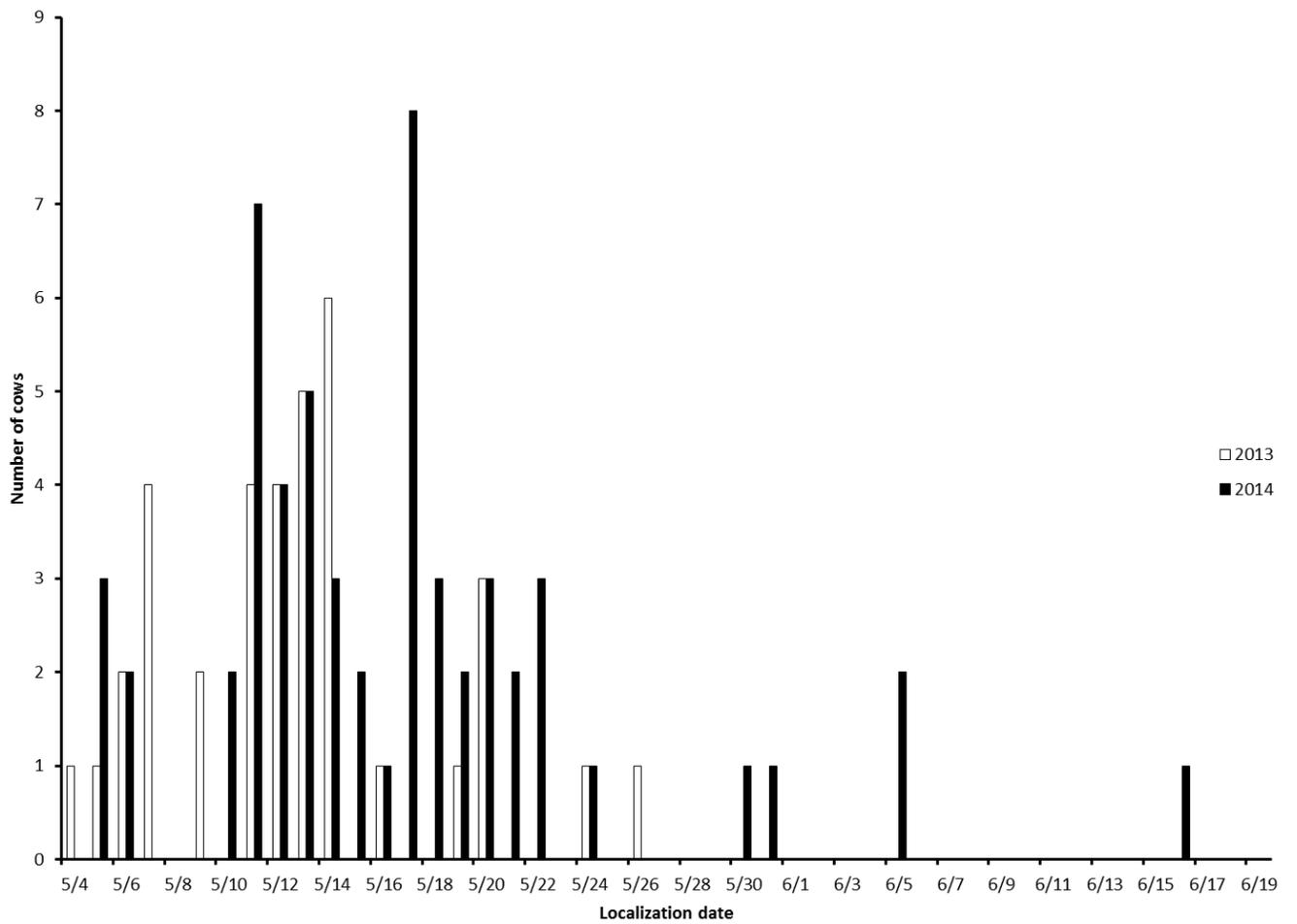


Figure 4. Temporal distribution of calving localizations of collared adult female moose, northeastern Minnesota, May-June 2013 and 2014.

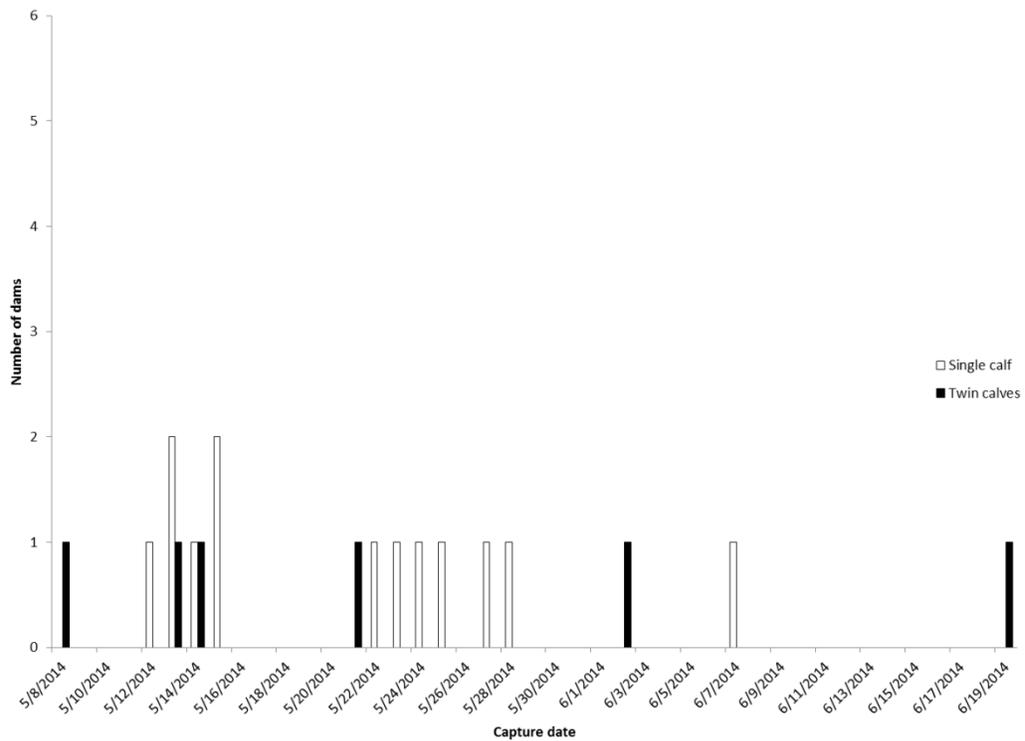
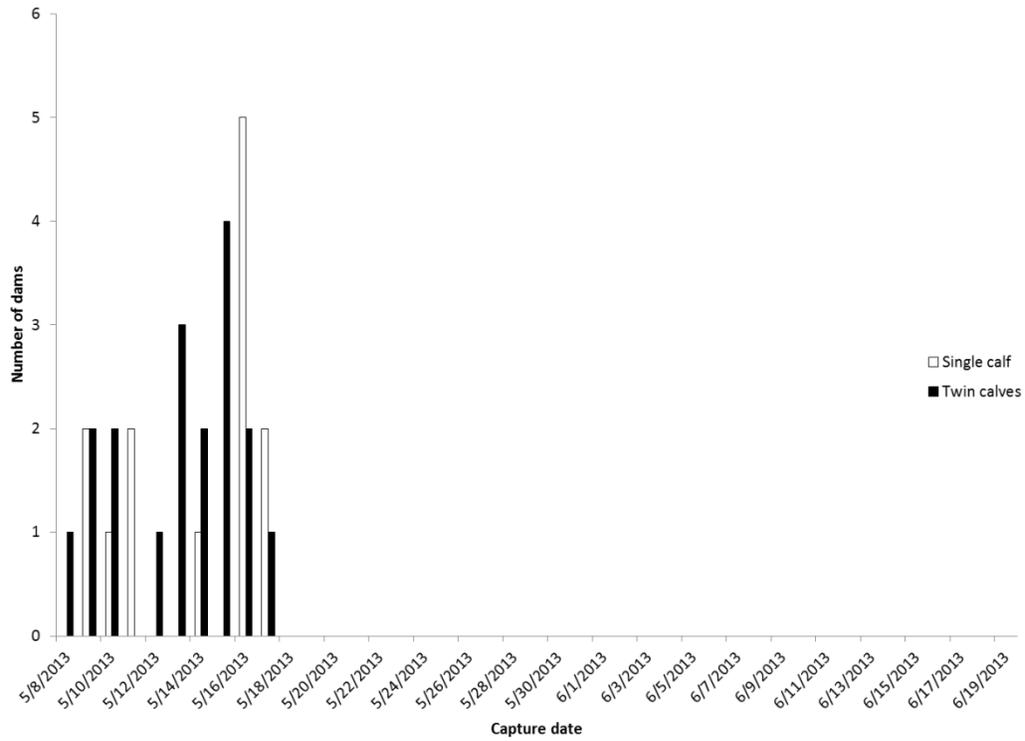


Figure 5. Number of moose dams with single and twin calves captured and handled, northeastern MN, 8-17 May, 2013 (top) and 8 May-19 June 2014 (bottom).

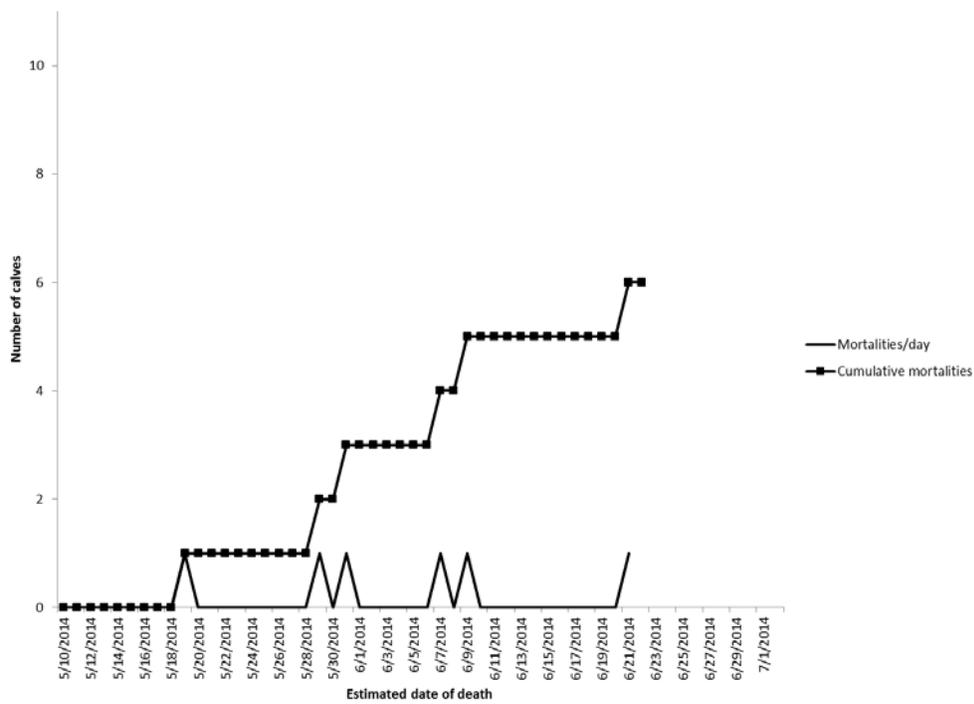
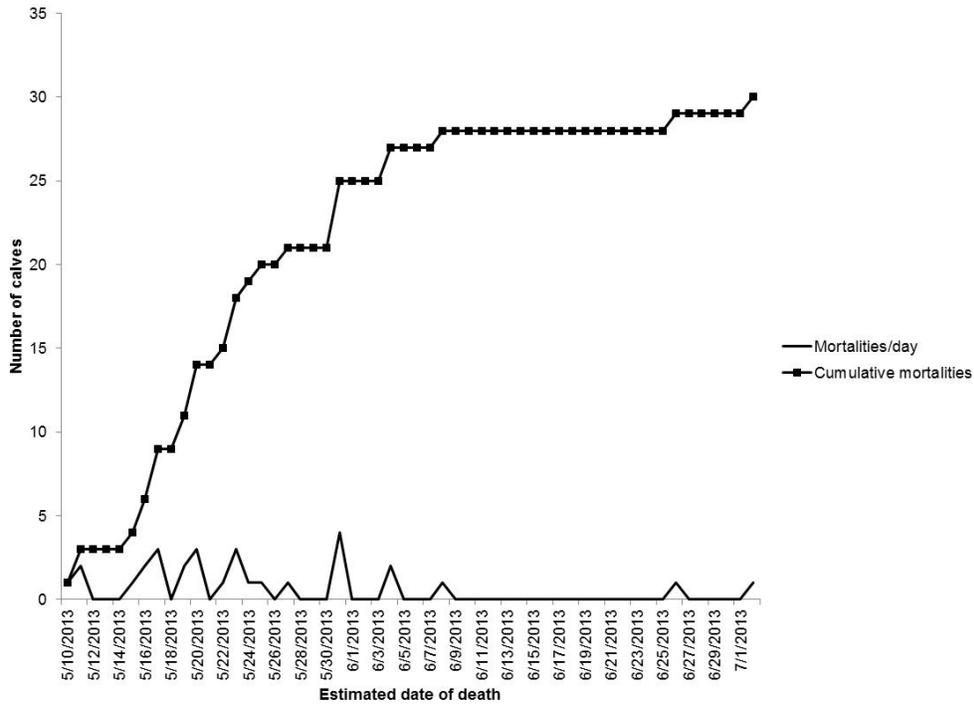


Figure 6. Number of natural mortalities of moose calves by day and cumulative mortality by day, northeastern MN, 10 May – 2 July 2013 (top) and 10 May-9 June 2014 (bottom).

Kaplan-Meier

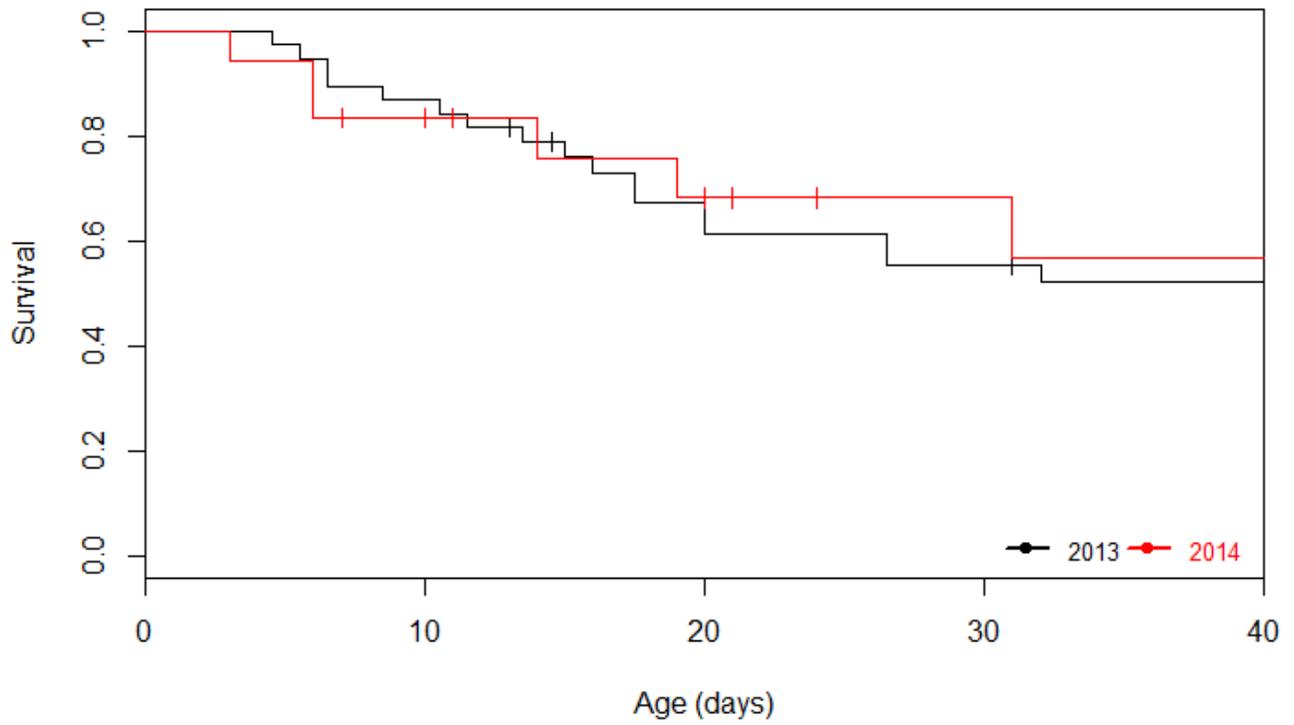


Figure 7. Kaplan-Meier survival curves plotting natural mortality for moose neonates (0-40 days) in northeastern Minnesota, 2013 ($n = 34$) and 2014 ($n = 11$). Plus symbols (+) indicate slipped (censored) collars.

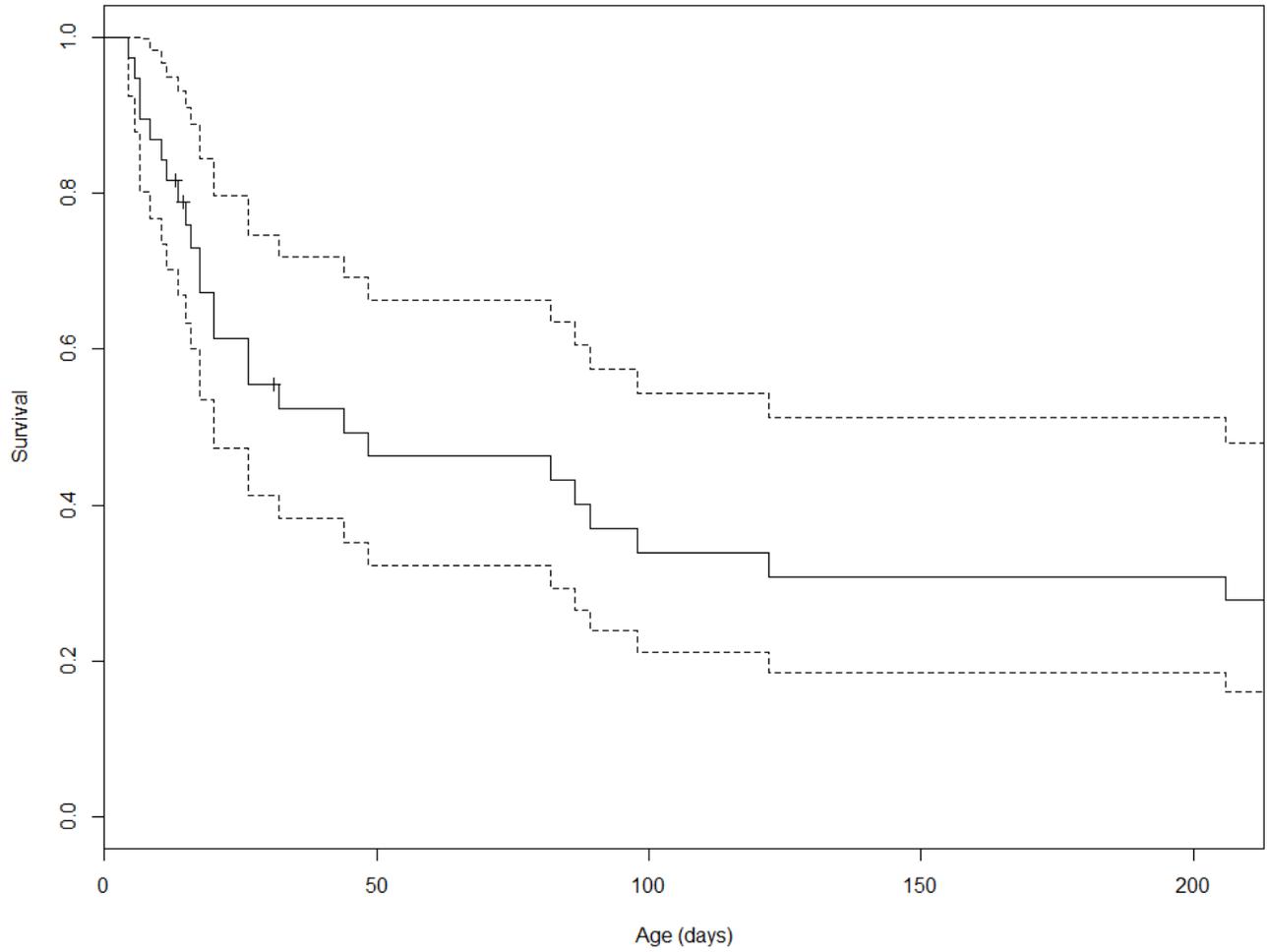


Figure 8. Kaplan-Meier survival curve ($\pm 95\%$ confidence intervals) plotting natural mortality for moose calves (0-230 days) in northeastern Minnesota, 2013 ($n = 34$). Plus symbols (+) indicate slipped (censored) collars.

David L. Garshelis and Karen V. Noyce

SUMMARY OF FINDINGS

During April 2013–March 2014, we monitored 24 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, poorest food), Chippewa National Forest (CNF; central), Camp Ripley (southern fringe), and a site at the northwestern (NW) edge of the range. Most of the focus of this study has been in the NW site in recent years. This area had the highest reproductive rate of our 4 study sites, due to an abundance of both agricultural crops and natural foods. Our longest span of reproductive data (1981–2014) is in the CNF, where we observed some evidence that litter size may have increased, with a concomitant increase in cub mortality; however, this inference is limited by small sample size in recent years. Hunting has been the primary source of mortality in all areas; but with a concerted effort to discourage hunters from shooting collared bears, and by clearly marking bears with large ear tags, no collared bears that we knew of were killed by hunters in fall 2012 or 2013 (although 2 collared bears could not be found). The oldest bear in this study, which was the oldest known wild bear of any species, died naturally at 39 years old.

INTRODUCTION

Telemetry-based research on black bears was initiated by the Minnesota Department of Natural Resources (MNDNR) in 1981, and has been ongoing since then. Objectives shifted over the years, and study areas were added to encompass the range of habitats and food productivity across the bear range. For the first 10 years, the bear study was limited to the Chippewa National Forest (CNF), near the geographic center of the Minnesota bear range (Figure 1). 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 Military Reserve, at the southern periphery of the bear range, was added as a second study site in 1991. The reserve is unhunted, but bears may be killed by hunters when they range outside, which they often do in the fall. Oaks (*Quercus* sp.) are plentiful within the reserve, and cornfields border the reserve. Voyageurs National Park (VNP), at the northern edge of the Minnesota range (but bordering bear range in Canada) was added as a third study site in 1997. Soils are shallow and rocky in this area, and foods are generally less plentiful than in the other sites. Being a national park, it is unhunted, but like Camp Ripley, bears may be hunted when they range outside.

In 2007 we initiated work in a fourth study site at the northwestern edge of the Minnesota bear range (henceforth NW; Figure 1). This area differs from the other 3 areas in a number of respects: (1) it is largely agricultural (including crop fields, like corn and sunflowers, that bears consume), (2) most of the land, including various small woodlots, is privately-owned, with some larger blocks of forest contained within MDNR Wildlife Management Areas (WMAs) and a National Wildlife Refuge (NWR); (3) the bear range in this area appears to be expanding and bear numbers have been increasing, whereas most other parts of the bear range are stable or declining in bear numbers; and (4) hunting pressure in this area is unregulated (it is within the no-quota zone, so there is no restriction on numbers of hunting licenses).

OBJECTIVES

1. Quantify temporal and spatial variation in cub production and survival;
2. Assess bear mortality in different parts of the bear range.

METHODS

We previously 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 used VHF collars in CNF, Camp Ripley, and VNP, and GPS collars in the NW study site. We used aerial telemetry to locate den sites. 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 dart gun. Bears were then removed from the den for processing. We measured and weighed them, assessed body condition, and took blood and hair samples. We changed or refit the collar, as necessary. All collared bears had brightly-colored, cattle-size ear tags (7x6 cm; Dalton Ltd., UK) that would be plainly visible to hunters.

Six bears were implanted in December dens with a new Insertable Cardiac Monitor developed for human heart patients (Reveal LINQ™, Medtronic Inc., Minneapolis, MN). The device is small enough (4.0 x 7.2 x 44.8 mm; 2.4 grams) to be injected subcutaneously in a left peristernal location. Surgical sutures were used to close the puncture site. The device provided wireless transmission of heart and activity data to an antenna buried under the nest material in the den, which was then relayed by cell phone to a base station. Besides providing physiological information, the heart rate and activity data also yielded birth dates of cubs, signaled by significantly increased heart rates and activity (just prior to birth), followed by a rapid decline to lower than pre-birth levels, which was maintained the rest of the denning period.

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

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

RESULTS AND DISCUSSION

Radiocollaring and Monitoring

As of April 2013, the start of the current year's work, we were monitoring 24 radiocollared bears: 4 in the CNF, 8 at Camp Ripley, 3 in VNP, and 9 in the NW (Table 1). We did not trap any new bears this year, but we collared 2 yearlings in dens, and 1 new bear was found and collared in a den at Camp Ripley. Since 1981 we have handled >800 individual bears and radiocollared >500, 361 of which were followed until they died (Table 2).

Reproduction

Six collared females produced cubs in 2014. All bears maintained a 2-year reproductive cycle. All 11 collared females that had cubs in 2013 had surviving yearlings in this winter's den. Based on heart rate data from 3 pregnant females with implanted wireless cardiac devices, the following birthdates were discerned: 2, 7, and 9 January 2014.

Since 1982, within the 4 study areas, we have checked 275 litters with 703 cubs ($\bar{x} = 2.6$ cubs/litter), of which 52% were male (Tables 3–6). Mortality of cubs during their first year of life averaged 21%, with mortality of male cubs (25%) exceeding that of females (16%; $\chi^2 = 5.73$, $P < 0.02$). The timing and causes of cub mortality are unknown.

Reproductive rates (cubs/female 4+ years old: combining litter size, litter frequency, and age of first reproduction into a single parameter) were highest in the NW study area, and lowest

in VNP (Figure 2). This corresponds with food conditions. The NW site contains not only agricultural crops consumed by bears, but also an abundance of natural foods, especially along the edges of woodlots. 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 just had their first litter, which tended to be smaller. Reproductive rates for 7+ year-old bears in the CNF and Camp Ripley were similar, although Camp Ripley bears tended to mature earlier (Figure 2). Reproductive rates for older bears at Camp Ripley appeared to decline this year because all of them had cubs last year and were with yearlings this year.

No strong trends in reproduction were evident through time. In the CNF, where we have been monitoring reproduction since 1982 (Table 3), we have observed a general increase in litter size (\bar{x} : 1982–1992=2.5; 1993–2003=2.7; 2004–2014=3.0 cubs/litter; Figure 3). Concomitantly, cub mortality has increased (\bar{x} :1982–1992=16%; 1993–2003=18%; 2004–2014=30%). Higher cub mortality may, in part, be related to a higher production of male cubs (Figure 3); however, female cub mortality in the CNF has increased dramatically, averaging only 11% during 1982–2003 but jumping to 28% since 2004 (nearly equal to males'). Sample sizes have declined in recent years (Figure 3), however, hampering interpretations of these data.

Notably all litters that we monitored in the last 10 years in the CNF have had at least 3 cubs (Table 3). We are cautious about inferring too much from these data, though, due to a small number of collared bears in the CNF: since 2009, we have only been monitoring 2 mature females (14 and 19 years old in 2014), neither of which has ever produced a 2-cub litter, and both of which recently produced 4-cub litters (one in 2010 and one in 2011). Only 1 other bear in this study (none from study areas other than CNF) have produced 5 or more consecutive litters of 3 or more cubs (while they were radio-collared). This goes to show that individual variation among females is important in reproductive output. The 3 most prolific bears were not unusually heavy (160–210 pounds in March dens at 7+ years old).

Mortality

Legal hunting has been the dominant cause of mortality among radiocollared bears from all study sites (Table 2). Recently, hunters seemed to have respected our request not to shoot collared bears (with large eartags): none were known to have been shot during either 2013 or 2014. However, we could not locate 2 collared bears when we searched for their den, leaving the possibility that they were killed and not reported. No bears were known to have died from any other human-related cause this year.

One particularly small adult female bear (143 pounds in March 2013 at 8 years old) in VNP lost its collar, which we found tangled in boughs of a fallen balsam fir tree (*Abies balsamea*) when we attempted to locate its den. It is unusual that a collar of an adult female could be pulled off, suggesting that the bear had been losing weight, and may have died. We could not search the site for remains because it was covered in deep snow.

The oldest bear on our study, a 39-year-old female in the CNF (as of January 2013) died this year (July 2013). This bear, identified as #56, was first trapped and radio-collared at 7 years old in 1981, the first summer of our research. From assessment of her cementum annuli, we concluded that the cubs with her at the time were from her second litter. We monitored her from then until her death, 31 years later.

Bear #56 was the only collared bear in our study that died of old age. She outlived the next longest survivor by 16 years. To our knowledge, she was older than any known-age bear of any species in the wild. Few wild bears reach even 30 years old; one Alaskan brown bear was reported to live to 34 years old (Schwartz et al. 2003). Among the >60,000 teeth that the MDNR has collected and aged from hunter-killed bears since 1975, only 3 (0.005%) lived past 30 years; the oldest appeared to be 33 when it was shot, however interpretation of cementum annuli in teeth of very old bears is imprecise.

The longevity of bear #56 is likely attributable to a combination of factors, including the

location of her home range in a forested area with relatively few people or major roads and a general wariness of people — she was never reported in a U.S. Forest Service campground within her home range, apparently did not visit hunters' baits, and was rarely caught during 10 years of our trapping efforts that included her home range. Additionally, her long life, and our chance to witness it, certainly entailed a bit of luck. Her last years of life were prolonged by having brightly-colored eartags and radiocollar, and our request to hunters not to shoot her.

The last time bear #56 was handled, in March 2010, her body condition was good, but her teeth showed excessive wear, with active gum disease (Figure 4), and both eyes were clouding. By that summer it was clear that her hearing was poor. Her eyesight and hearing continued to deteriorate in ensuing years. This enabled us to approach her and observe her occasionally (Figure 5), although she never became human-habituated (she would flee upon becoming aware of our presence). She was observed by local people with increasing frequency foraging along trails and traveling on forest roads, likely because travel was easier there. Despite her deteriorating mobility, in 2011, bear #56 travelled over 20 miles outside her home range to feed on acorns along the Prairie River for several weeks in late summer.

During her last few years, her notoriety helped keep her alive. Knowing of her advanced age and physical impairments, residents often reported her whereabouts and tolerated her odd behavior. One local resident reported her in their yard and inquired whether she was deaf, as she did not respond to their shouting and clapping. In at least one case a hunter passed up his chance to shoot a bear when he learned that it was bear #56 visiting his bait site.

Bear #56 was a key animal during the many years of our research in the CNF study area. She and her offspring provided an almost uninterrupted record of reproduction, survival, movements, and eventually, senescence within a single matriarchal lineage over a 32-year span. During 1981–1995, bear #56 produced 8 litters, totaling 22 cubs, and successfully reared a remarkable 21 of them to 1½ years, the age of family breakup. Signs of reproductive senescence started to appear when she uncharacteristically lost 2 of the 3 cubs born in 1997. In 1999, at age 25, she produced and raised only 1 cub. The last evidence of any reproductive activity occurred in 2001, when, despite being in excellent nutritional condition in her den, we found her lactating, but alone.

For the next 10 years, 2001 – 2010, bear #56 maintained body mass and condition, though by 2010 (weighing 192 pounds in March) she had declined somewhat from her peak weight (255 pounds in March). During her last 2 summers, we observed her on several occasions walking unsteadily, but she was nevertheless able to walk at a pace through the forest faster than we could keep up. Sometime in July, 2013, she moved several miles west of her normal summer home range and died, apparently non-violently of natural causes. When we found her remains, they were already well-decayed but mainly undisturbed (Figure 6).

We collected her skeleton as a reference specimen. It exhibited extensive bone loss in the skull and mandibles around the teeth and advanced osteoarthritis, particularly along the spine. The skull was scanned and a model recreated for posterity (Figure 7). One daughter, now 19 years old, remains collared in the CNF.

ACKNOWLEDGMENTS

We thank the collaborators in this study: Brian Dirks (MDNR) at Camp Ripley; Paul Iazzo (University of Minnesota) and Tim Laske (Medtronic, Inc.) in the NW and CNF study sites; and Steve Windels and Bryce Olson (National Park Service) in Voyageurs National Park. Agassiz NWR kindly provided use of their bunkhouse and assistance during the winter fieldwork.

LITERATURE CITED

Schwartz, C.C., Keating, K.A., Reynolds, H.V., Barnes, V.G., Sellers, R.A., Swenson, J.E., Miller, S.D., McLellan, B.N., Keay, J., McCann, R., Gibeau, M., Wakkinen, W.F., Mace, R.D., Kasworm, W., Smith, R., and S. Herrero. 2003. Reproductive maturation and senescence in the female brown bear. *Ursus*:109–119.

Table 1. Fates of radiocollared black bears in 4 study sites (Chippewa National Forest, Camp Ripley, Voyageurs National Park, and northwestern Minnesota), April 2013–March 2014.

	CNF	Camp Ripley	VNP	NW
Collared sample April 2013	4	8	3	9
Killed as nuisance				
Killed in vehicle collision				
Killed by Minnesota hunter				
Natural mortality	1			
Dropped collar		1	1	
Failed radiocollar				
Lost contact ^a			1	1
Collared in den	1	1		1
Collared sample April 2013	4	8	1	9

^a Due to radiocollar failure, unreported kill, or long-distance movement.

Table 2. Causes of mortality of radiocollared black bears ≥ 1 year old in 4 Minnesota study sites, 1981–2014. Bears did not necessarily die in the area where they usually lived (e.g., hunting was not permitted within Camp Ripley or VNP, but bears were killed by hunters when they traveled outside these areas).

	CNF	Camp Ripley	VNP	NW	All combined
Shot by hunter	223	11	15	12	261
Likely shot by hunter ^a	8	1	0	4	13
Shot as nuisance	22	2	1	3	28
Vehicle collision	12	8	1	3	24
Other human-caused death	9	1	0	0	10
Natural mortality	8 ^b	3	5	0	15
Died from unknown causes	4	2	0	3	9
Total deaths	286	28	22	25	361

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

^b Only 1 bear died of “old age”.

Table 3. Black bear cubs examined in dens of radiocollared mothers in or near the Chippewa National Forest during March, 1982–2014. High hunting mortality of radiocollared bears severely reduced the sample size in recent years.

Year	Litters checked	No. of cubs	Mean cubs/litter	% Male cubs	Mortality after 1 yr ^a
1982	4	12	3.0	67%	25%
1983	7	17	2.4	65%	15%
1984	6	16	2.7	80%	0%
1985	9	22	2.4	38%	31%
1986	11	27	2.5	48%	17%
1987	5	15	3.0	40%	8%
1988	15	37	2.5	65%	10%
1989	9	22	2.4	59%	0%
1990	10	23	2.3	52%	20%
1991	8	20	2.5	45%	25%
1992	10	25	2.5	48%	25%
1993	9	23	2.6	57%	19%
1994	7	17	2.4	41%	29%
1995	13	38	2.9	47%	14%
1996	5	12	2.4	25%	25%
1997	9	27	3.0	48%	23%
1998	2	6	3.0	67%	0%
1999	7	15	2.1	47%	9%
2000	2	6	3.0	50%	17%
2001	5	17	3.4	76%	15%
2002	0	0	—	—	—
2003	4	9	2.3	22%	0%
2004	5	13	2.6	46%	33%
2005	6	18	3.0	33%	28%
2006	2	6	3.0	83%	33%
2007	2	6	3.0	67%	17%
2008	1	3	3.0	100%	33%
2009	1	3	3.0	33%	33%
2010	1	4	4.0	100%	50%
2011	1	4	4.0	25%	50%
2012	1	3	3.0	67%	33%
2013	1	3	3.0	67%	0%
2014	1	3	3.0	67%	0%
Overall	179	472	2.6	53%	19%

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

Table 4. Black bear cubs examined in dens in northwestern Minnesota during March, 2007–2014.

Year	Litters checked	No. of cubs	Mean cubs/litter	% Male cubs	Mortality after 1 yr
2007	2	6	3.0	33%	100%
2008	5	15	3.0	67%	22%
2009	1	3	3.0	33%	33%
2010	6	17	2.8	41%	13%
2011	2	4	2.0	75%	25%
2012	4	10	2.5	60%	10%
2013	3	9	3.0	67%	18%
2014	3	8	2.7	0%	0%
Overall	26	72	2.8	48%	26%^a

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

Table 5. Black bear cubs examined in dens in or near Camp Ripley Military Reserve during March, 1992–2014.

Year	Litters checked	No. of cubs	Mean cubs/litter	% Male cubs	Mortality after 1 yr ^a
1992	1	3	3.0	67%	0%
1993	3	7	2.3	57%	43%
1994	1	1	1.0	100%	—
1995	1	2	2.0	50%	0%
1996	0	0	—	—	—
1997	1	3	3.0	100%	33%
1998	0	0	—	—	—
1999	2	5	2.5	60%	20%
2000	1	2	2.0	0%	0%
2001	1	3	3.0	0%	33%
2002	0	0	—	—	—
2003	3	8	2.7	63%	33%
2004	1	2	2.0	50%	—
2005	3	6	2.0	33%	33%
2006	2	5	2.5	60%	—
2007	3	7	2.3	43%	0%
2008	2	5	2.5	60%	0%
2009	3	7	2.3	29%	29%
2010	2	4	2.0	75%	25%
2011	3	8	2.7	50%	25%
2012	1	2	2.0	100%	0%
2013	6	14	2.3	50%	21%
2014	1 ^b	— ^b	—	—	—
Overall	40	94	2.4	52%	21%

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

^b Cubs heard, litter not handled. This litter not included in total.

Table 6. Black bear cubs examined in dens in Voyageurs National Park during March, 1999–2014. All adult collared females were killed by hunters in fall 2007, so no reproductive data were obtained during 2008–2009.

Year	Litters checked	No. of cubs	Mean cubs/litter	% Male cubs	Mortality after 1 yr ^a
1999	5	8	1.6	63%	20%
2000	2	5	2.5	60%	80%
2001	3	4	1.3	50%	75%
2002	0	—	—	—	—
2003	5	13	2.6	54%	8%
2004	0	—	—	—	—
2005	5	13	2.6	46%	20%
2006	1	2	2.0	50%	0%
2007	3	9	3.0	44%	—
2008	0	—	—	—	—
2009	0	—	—	—	—
2010	1	2	2.0	50%	0%
2011	1	2	2.0	0%	0%
2012	1	2	2.0	0%	50%
2013	1	2	2.0	50%	—
2014	1	3	3.0	33%	—
Overall	29	65	2.2	48%	27%

^a Blanks indicate no cub mortality data because no cubs were born to collared females, or collared mothers were lost from study (died or lost collar) before denning with yearlings.

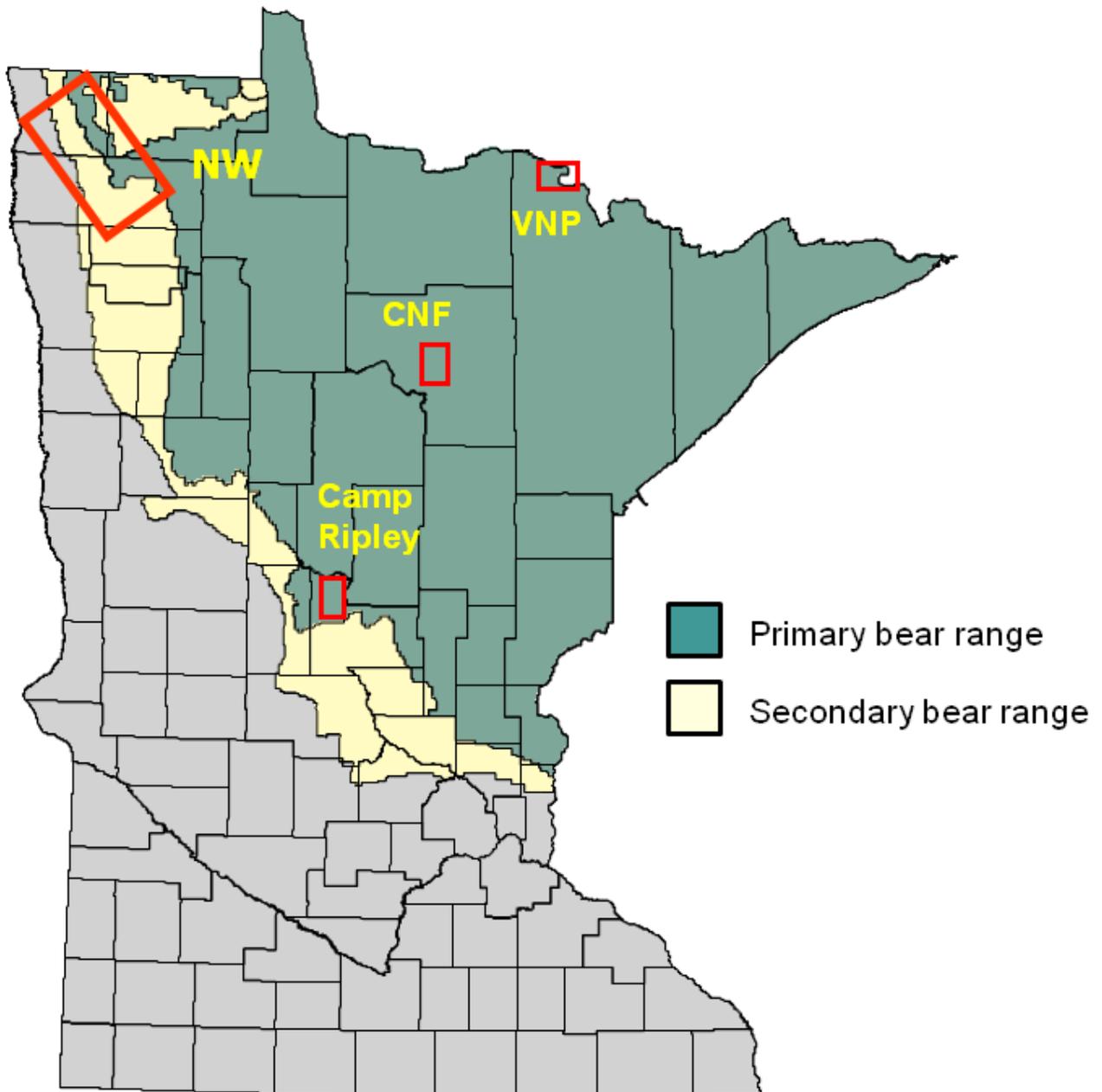


Figure 1. Location of 4 study sites within Minnesota's bear range: CNF (Chippewa National Forest, central bear range; 1981–2014); VNP (Voyageurs National Park, northern fringe of range; 1997–2014); Camp Ripley Military Reserve (near southern edge of range; 1991–2014); NW (northwestern fringe of range; 2007–2014).

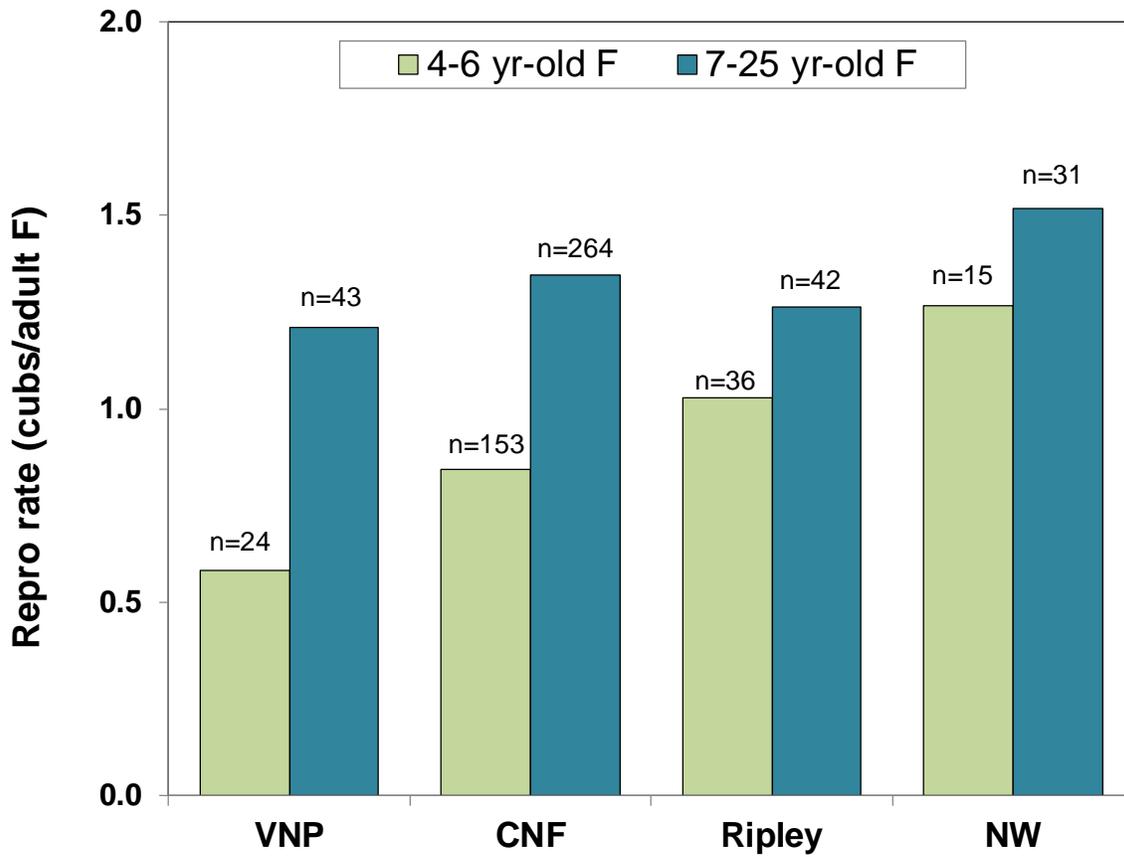


Figure 2. Reproductive rates of radiocollared bears within 4 study sites (see Figure 1) through March 2014. Sample sizes refer to the number of female bear-years of monitoring in each area for each age group. Data include only litters that survived 1 year (even if some cubs in the litter died). Some bears in CNF, Camp Ripley, and NW produced cubs at 3 years old, but are not included here.

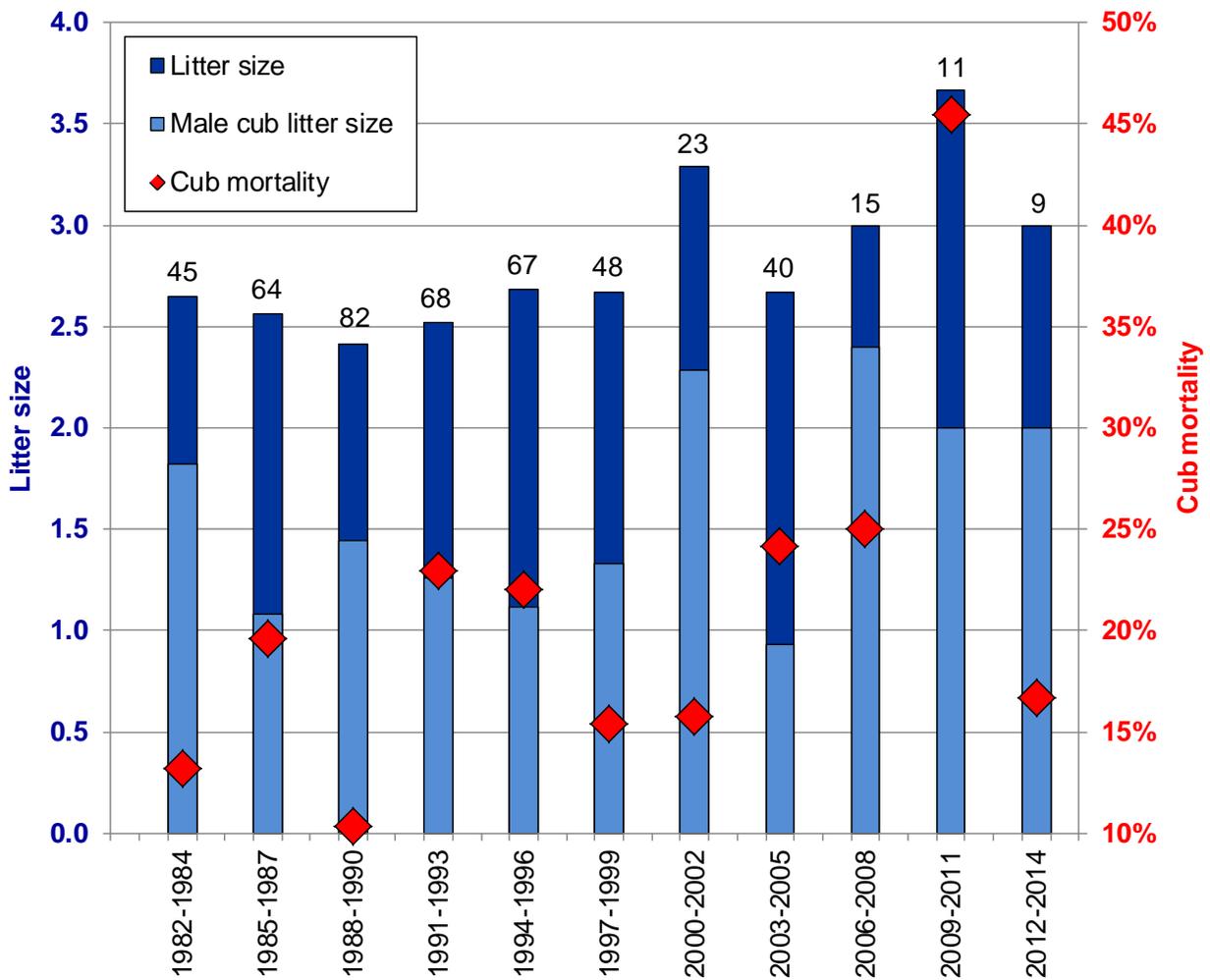


Figure 3. Cub litter sizes (examined in dens in March) in the CNF appeared to increase through time (1982–2014), with more males per litter. Correspondingly, cub mortality has also trended upward (assessed by examining yearlings in dens of radio-collared mothers that had cubs the year before). Each bar represents 3 years of combined data, with number of cubs indicated above bars. Note that secondary Y-axis begins at 10%.



Figure 4. Deteriorating teeth condition of bear #56 the last time she was handled (March 2010) at 36 years old. Note missing left lower canine and 2 missing lower incisors, as well as gum disease.



Figure 5. Despite poor teeth, bear #56 nevertheless foraged on natural foods throughout her life (left, spring 2011). However, her dulled senses made her more easily observable (right, within 1 month of death at age 39). When she did become aware of our presence, she quickly moved away (she was not human-habituated). The colored collar and eartag, made her more visible to hunters, who avoided shooting her at our request, enabling her to live to very old age.



Figure 6. Bear #56 was found dead in August, 2013, at 39 years old. The remains as found (shown unaltered in photo) indicate that she died naturally, and was not preyed upon by another bear or wolves.



Figure 7. The skull of bear #56, the oldest recorded wild bear of any species, was CT-scanned (top) and a model produced with a 3-D printer (bottom right) to preserve for posterity, as the original skull (bottom left) is quite fragile due to extreme wear and degeneration.

Karen V. Noyce and David L. Garshelis

SUMMARY OF FINDINGS

During the spring and summer of 2012, we conducted genetic capture–mark–recapture (CMR) of black bears in the Chippewa National Forest (CNF) using hair traps in order to ascertain changes in bear abundance since our last estimates of this same population during the 1980s and 1990s. Because previously we captured bears in physical traps or by camera traps, one objective here was to examine potential sources of bias specifically related to hair-trapping, which could hinder comparisons with previous estimates. We set 121 2-stranded barbed wire hair traps in the same study area as used in the 1980s and 1990s. We checked sites during 6 sampling sessions at 10-day intervals during late-May to mid-July. Visitation by bears was high (55% of site-session checks), yielding 2784 hair samples in 1642 clusters of 1–11 adjacent barbs. We assumed that clusters of barbs with hair represented places where a bear went over or under a wire and rubbed across several barbs. We submitted a sample from each of 1113 clusters for genetic analysis, and 1019 (92%) were successfully genotyped. We examined data for evidence of capture heterogeneity due to annual molt and barbed wire position. In nearly two-thirds of the clusters, hair occurred only on the lower strand of barbed wire, suggesting that most hair was snagged when bears crawled under the lower wire. Hair occurred on both upper and lower wires in 22% of clusters and only on upper wires in 16% of clusters. Samples from both males and females were more common on lower wires, but male hair was relatively more common than female hair on upper wires, suggesting that 2-strand designs may better capture population diversity than a single strand of barbed wire, as used in many studies. The total number of sites visited by bears per sampling session increased from the first to second session, and was relatively consistent thereafter. The number of barbs and clusters with hair at each site, however, declined through the study, as did the relative frequency of clusters involving both wires, suggesting that hair was harder to snag during later sessions because the long winter coat had been molted and the new summer coat was harder to pluck out. Awareness of these trends will help inform modeling and interpretation of CMR population estimates. Further work with camera traps at hair snares is needed to ascertain whether cubs are sampled by hair snaring, an important consideration in comparing this estimate to previous estimates that excluded cubs.

INTRODUCTION

During summer 2012 we conducted DNA-based capture–mark–recapture (CMR) of black bears to estimate bear population size in a 300-km² study area in northern Itasca County, Minnesota, mainly within the Chippewa National Forest. We specifically sought to examine population change since the 1980–1990s, when we last estimated abundance on the same study area using physical trapping and camera trapping. Multiple lines of evidence suggested that bear numbers had declined significantly since our last estimates.

DNA-based CMR, using hair collected from barbed-wire hair traps, has become the “gold standard” for estimating bear population size where logistics and budgets allow. Genetic CMR has many advantages over marking bears through physical captures and radiocollaring. Because bears are not handled, checking hair traps requires a lower level of skill and less equipment. Also, more traps can be set because they do not have to be checked daily, and bears likely have less aversion to the traps, so are more likely to be recaptured. Thus, sample sizes from hair trapping are apt to be large (improving precision) and less biased. However, hair trapping potentially introduces new sources of sampling heterogeneity that are unique to this method. Hence it was important to consider potential forms of bias that may have differed from our physical mark-recapture work.

Most hair trapping studies have employed a barbed wire enclosure constructed by

stretching a single strand of barbed wire around several trees at a height of about 50 cm above the ground. A scent lure, hung over the center of the enclosure but out of reach of bears, entices bears into the enclosure (Woods et al. 1999). This design was developed originally for grizzly bears (*Ursus arctos*) and was intended to snag hair from all sizes of bears – i.e., the barbed wire was high enough to sample large bears that climbed over the wire, but still low enough to sample smaller bears that crawled underneath, including cubs of the year (Woods et al. 1999, Kendall et al. 2008). Other protocols made use of hair left at trees (Kendall et al. 2008) or power poles (Karamanlidis 2007) where bears naturally rub, scratch, and mark. In this case, barbed wire is secured to the tree or pole to increase the amount of hair sampled when bears rub.

Each of these methods has some degree of bias with regard to size and/or sex of bears sampled (Kendall et al. 2008; Boulanger et al. 2006). Where both methods have been employed, as in Banff National Park, Alberta, Canada, hair traps caught relatively few male grizzly bears, whereas at rub trees, males were more likely to be captured than females (Sawaya et al. 2012).

This method has been adapted for use on black bears in locations around North America, with some modifications. Researchers found that often, bears entered hair traps but did not leave hair on the barbed wire. To increase capture efficiency, Tredick et al. (2007) added a lower wire to their hair traps (20 cm above ground) to force animals to squeeze through more restricted spaces and improve the detection of small bears that could otherwise crawl under wires without leaving hair. However, they concluded that samples from the lower wire were of poor quality and thus did not increase efficiency enough to offset the extra expense and time of construction. Lowe (2011), however, working in south central Louisiana, suspected that large males might be systematically under-represented using 1-strand hair traps; photos from remote cameras showed large males entering enclosures by stepping on or over wires and leaving no hair (Hooker 2010). Estimated capture rate for females was twice as high as for males. Subsequent work in the same study area, using hair traps with an added upper wire at 70 cm resulted in more equal estimates of capture probability for males and females (O'Connell 2013).

Other potential problems stem from the timing of genetic CMR studies, which often run through much of the spring and summer. In temperate regions, this coincides with a bear's annual molt. In May and June, when most underfur and guard hair are lost, shedding hair is easy to snag, whereas the hairs of new pelage are more tightly bound, so significant changes in capture probability may occur among sampling sessions. If the molt proceeds differently for different sex-age groups, this further complicates the task of categorizing capture heterogeneity and accounting for it in analysis of data.

In our CMR study, we wished to sample all age groups except cubs of the year, because our previous estimates (physical capture and camera capture) excluded cubs. A pilot project suggested that a 2-strand hair trap with wires placed at approximately 45 and 75 cm above the ground would be the most likely to catch bears of all sizes except cubs. Here we examine patterns in the hair we captured so as to discern types and magnitude of sampling bias that barbed wire hair traps introduced into our population estimation procedure.

OBJECTIVES

1. Determine if hair snaring introduces significant sampling bias and/or heterogeneity into CMR sampling, and if so, what type(s).
2. Determine if cubs of the year were sampled by hair snares.

METHODS

The study area was same CNF study site where previous CMR estimates were obtained (Figure 1). It contains good access via 2 main paved roads, smaller unimproved roads, and forest trails. Ownership is mainly national and state forest, with additional county and private lands.

We erected hair-snare traps using 2 strands of 4-pronged barbed wire wrapped around 3-5 trees, forming an enclosure. Barbed wire was placed at 45 cm and 75 cm off the ground (Figure 2). We erected 1 trap in each of 121 square-mile sections (121 mi²). Within each of these grid cells, we set a trap in what we perceived as good bear habitat to maximize visitation. We set traps at least 100m from main roads, but often along trails that bears might use.

We suspended a bag of bacon and a scent lure from a string (above the reach of a bear) across the middle of each trap, and put bait and scent lure on a pile of brush in the middle of the enclosure (Figure 2). Baits and lures were refreshed at each trap visit. We added different types of lures at each trapping session to maintain novelty for the bears. We checked all traps 6x at intervals of 10 days. We did not move traps between sessions. At each trap check, all bear hair was removed from the wire. Each clump of hairs on a barb was collected in a separate envelope, and labeled as to proximity to other barbs with hair, trap number, and date. We coded barbs of hair that were adjacent (next to, on either the same wire or the one above/below) as being from the same cluster. A cluster could include only a single barb.

We set camera traps at some of the hair traps that were visited by bears to gauge whether cubs of the year left hair on wires, and to assess the responses of different bears to the wires and the baits. Hair traps were erected the third week of May, 2012. We checked all 121 hair traps 5 times (605 site-sessions), then dismantled 36 traps that were never visited by a bear, leaving 85 to be checked in session 6 and removed the third week of July.

Hair samples were submitted to Wildlife Genetics International (Nelson, British Columbia, Canada) for genotyping. As our budget was not sufficient to analyze all collected hair samples, we subsampled the collection. In subsampling we made an attempt to maximize the number of different bears that visited the sites, so (1) we included at least 1 sample from each site-session with hair, and (2) we did not submit hairs from multiple barbs within the same cluster. We also submitted hair samples from 4 radiocollared bears and their current offspring living on the study area (collected during den visits) to determine whether they visited the hair traps. The lab also identified likely family groups.

RESULTS

Bears visited 101 of the 121 hair trap sites, resulting in 377 of 690 (55%) total site-sessions yielding hair (Table 1). We collected hair from 2784 barbs that occurred in 1642 separate clusters of 1–11 adjacent barbs. For genotyping, we initially chose (randomly) 1 barb from each of the 377 site-sessions with hair. We then chose additional random samples from among the remaining 1265 barb clusters.

Of 1113 samples that were analyzed, 14 appeared to be mixtures of >1 bear and 80 failed to amplify. Thus 1019 samples (92%) were successfully genotyped; these were from 96 different sites and 333 site-sessions. Genotyping identified 43 different individuals: 26 males and 17 females. Individual bears were detected up to 132 times each and up to 32 times in a single sampling session. Sex ratio of individuals visiting hair traps was heavy to males in all sampling sessions and did not vary through time ($\chi^2=0.96$, $df=5$, $P=0.97$).

Females that visited hair traps did not differ from males in either the number of sessions in which they were detected (Figure 3; $\chi^2=2.52$, $df=5$, $P=0.77$) or the number of sites they visited (Figure 4; $\chi^2=0.83$, $df=3$, $P=0.84$, for M vs.F visiting 1, 2-3, 4-7, or 8+ sites). About a third of both males (31%) and females (29%) were detected in only 1 sampling session. A similar percent of males (31%) and females (24%) were detected at only 1 site during the study. The number of sites and the number of sessions a bear was detected were positively related (Figure 5).

Camera trap photos revealed that many individuals visited the same hair trap multiple

times during a session. The same bear often entered and left a trap at different locations along its perimeter, variously crawling under, between, or over the wire (Figure 6). Multiple individuals sometimes used the same location for entering or exiting traps, conceivably producing mixed hair samples. Only 1% of analyzed samples were unusable for this reason, however, so this did not constitute a significant inefficiency in sampling.

The number of hair traps that yielded bear hair ranged from 30–79 per session (Table 1). The number of different individuals detected in a session ranged from 14–28 (Table 2). Visitation varied through time ($\chi^2=50.3$, $df=5$, $P<0.0001$), with fewer sites visited during the first session (late May) than in subsequent sessions (25% vs. 52 – 75% of hair traps visited; χ^2 multiple comparisons for proportions; $\alpha=0.05$). Visitation by bears was also higher during the last session than in sessions 2 or 3, at least in part because sites that had not been previously visited were removed for this session (Table 1).

We found that over the course of 6 sampling sessions, at sites visited by bears, the mean number of barbs/site that snagged hair, the number of clusters of barbs/site, and the number of adjacent barbs that comprised each cluster all showed significant declines (Figure 7; one-way AOV: $F=8.69$, 4.41, and 12.93, respectively; $df=5.0$, $P<0.001$). These changes, most noticeable in late June and July (sessions 4 – 6) coincided with changes observed in coat condition as bears molted (Figure 8), suggesting that hair became increasingly less likely to snag on barbs when bears entered or exited hair traps as the season progressed.

Bears left nearly twice as many hair samples on lower strands of barbed wire than on upper strands (Table 1). This was true for both males and females, although males were more likely to leave hair on upper wires only ($\chi^2=3.9$, $df=1$, $P=0.047$) or in clusters on both wires ($\chi^2=11.6$, $df=1$, $P<0.0001$) than were females (Figure 9). (Conversely, females were more likely to leave hair only on lower wire; $\chi^2=19.9$, $df=1$, $P<0.0001$). Clusters of barbs included only barbs on lower wires 62% of the time, only barbs on upper wires 16% of the time, and barbs on both wires 22% of the time. Clusters that included both wires decreased from May to July (Figure 10; $\chi^2=44.1$, $df=10$, $P<0.0001$), concurrent with an increase in the proportion left only on lower wires.

Genotyping identified several family groups that were sampled together at the same site and during the same session. Only 1 genotyped family group, a radio-collared mother and her yearlings, occurred where there was a camera. Two males also photographed at these sites were detected by hair. One camera site detected a mother with at least 1 cub present; this cub was not identified among the hair samples analyzed from the site. We had no means of ascertaining whether other detected family groups included cubs of the year.

DISCUSSION

Capture heterogeneity is a vexing problem in capture–recapture studies of most mammals because a fundamental requirement for deriving unbiased estimates of animal numbers and demographic parameters is that individuals are equally vulnerable to capture. This is rarely the case. Recent advances in analytic methods accommodate some degree of capture heterogeneity among groups and through time, however, these methods still require classifying data such that within identifiable groups likelihood of capture is rather uniform.

Most studies involving physical capture of bears in traps exhibit a trapping bias toward males, despite a living population with more females (at least in hunted populations). Previous trapping in the Chippewa National Forest produced a male-biased capture. Trap vulnerability was also influenced by age and, for females, reproductive status (with or without cubs of the year): subadult males (3–5 years old) and females without cubs were more vulnerable to capture than other bears and juvenile females (1–2 years) and adult males (>5 years old) were less vulnerable (Noyce et al. 2001). Some of these biases shifted through time (e.g. adult males and adult females with cubs were more likely to be caught toward the end of the breeding season in early July than during peak breeding in late May – mid-June). Further compromising any assumptions of equal catchability were differences among individuals that were not sex- or

age-based, likely attributable to individual differences in behavior and/or proximity to traps,

In designing this study we attempted to lessen some of the biases apparent in our earlier capture data by doubling trap density relative to that used in the 1980s and 1990s, making it more likely that small yearling home ranges would include at least 1 hair-trap, and by running the sampling season through mid-July, to sample adult females with cubs and adult males with about the same likelihood as other demographic groups (Noyce et al. 2001). We also specifically ended sampling by mid-July to minimize violation of the assumption of geographic closure, as 40% of bears in the study area, on average (and double that some years), left their summer home range after mid-July to migrate, mainly southward, in search of concentrated food where they could fatten for winter (Noyce and Garshelis 2011). Presumably other bears from elsewhere likewise moved into the study area

Despite these efforts, hair trapping in this study still produced a male-biased capture. We could not discern age-related effects because genetic samples do not provide information about age. The fact that we did not capture a rash of new bears in session 6, yet we caught more new bears than in sessions 4 and 5 (Table 2), suggests that we ended sampling just as seasonal movements began.

Nevertheless, our analysis also suggested that hair trapping may have introduced at least 2 more sources of capture heterogeneity that are unique to this method of capture, adding further complexity to the heterogeneity already present in bait- and trap-based capture projects. As in other studies (Lowe 2011, Sawaya 2012, O'Connell 2013), evidence suggested that the number and height of barbed wire strands potentially introduces size-based differences in capture probability (thus also tied to sex and age). Large bears (most likely males) can be under-represented in studies using a single-wire placed at ~50 cm because they can step over the wire without leaving a hair sample, particularly late in the molt when belly hair is sparse (Hooker 2010). We obtained a photograph of one large bear stepping over even a 75-cm-high wire (Figure 11), though clearly at this height contact with the bear's underside was likely.

The seasonal molt causes declines in capture vulnerability, likely in all individuals, though to varying degrees. Molt varies among individuals in timing (Figure 12), and it is unclear if the timing varies among identifiable demographic groups. We suspect that some bears that visited sites late in the trapping season did not leave hair, or left insufficient hair for genotyping. We did not collect hair from barbs containing only 1 or 2 hairs, and these became increasingly common in July. Nevertheless, the genetics lab used an average of 7.2 hairs per sample, and reported to us that this accounted for the genotyping success exceeding 90% (compared to ~70% in many similar studies).

One aspect of hair trapping in black bear studies that begs more documentation is how to optimize trap design to either include or exclude cubs of the year. Depending on rates of cub mortality in a population, this age class can be up to twice the size of the yearling class. In populations such as Minnesota, where juveniles experience heavy loss each year to hunters, cubs comprise 15–20% of the population. Clearly it is important to know whether or not this age class is included in hair sampling, particularly where an objective is to enumerate the hunted population for management purposes, which, in most places (including Minnesota), does not include cubs of the year. Specific to our case here, a population decline could be obfuscated by exclusion of cubs in the 1980–1990s trap and camera-based population estimates, but inclusion of this large cohort in the recent hair-snaring estimate. Although we attempted to exclude cubs, it is possible that as cubs grew, they became more susceptible to contacting the lower wire while visiting a site with their mother. This possibility requires further investigation with camera traps.

Whereas estimators have been developed that can handle a degree of capture heterogeneity, study design should still include measures to identify likely sources of sampling bias, minimize them by adjusting study design, and, where that is difficult, assess their potential effect on resulting estimates. We plan to further explore some of these questions with better camera documentation of bear behavior at hair traps.

ACKNOWLEDGMENTS

We sincerely thank the 2 volunteers who checked traps and meticulously collected hair: Chris Anderson and Chih-Chien (Jerry) Huang. We also thank the individuals who allowed us to set and check traps on their private land: Bradley Box, Mark Hawkinson, Dale Juntunen, Brad and Mary Nett, Jack Rajala, Scherer Brothers Lumber Company, and Thomas Schultz.

LITERATURE CITED

- Boulanger, J., M. Proctor, S. Himmer, G. Stenhouse, D. Paetkau, and J. Cranston. 2006. An empirical test of DNA mark-recapture sampling strategies for grizzly bears. *Ursus* 17:149–158.
- Hooker, M. J. 2010. Estimating population parameters of the Louisiana black bear in the Tensas River Basin, Louisiana, using robust design capture-mark-recapture. MSc Thesis, University of Tennessee, Knoxville, USA.
- Karamanlidis, A., Youlatos, D., Sgardelis, S., and Z. Scouras. 2007. Using sign at power poles to document presence of bears in Greece. *Ursus* 18:54 – 61.
- Kendall, K.C., Stetz, J., Roon, D.A., Waits, L.P., and J. Boulanger. 2008. Grizzly bear density in Glacier National Park. *Journal of Wildlife Management* 72:1693–1705.
- Lowe, C.L. 2011. Estimating population parameters of the Louisiana black bear in the Upper Atchafalaya River Basin. M.S. Thesis. University of Tennessee, Knoxville, USA.
- Noyce, K.V., D.L. Garshelis, and P. L. Coy. 2001. Differential vulnerability of black bears to trap and camera sampling and resulting biases in mark-recapture estimates. *Ursus* 12:211–226.
- Noyce, K.V., and D.L. Garshelis. 2011. Seasonal migrations of black bears (*Ursus americanus*): causes and consequences. *Behavioral Ecology and Sociobiology* 64: 823–835. DOI 10.1007/s00265-010-1086-x
- O'Connell, K.C., 2013. Population dynamics of the Louisiana black bear in the Upper Atchafalaya River Basin. MSc. Thesis. University of Tennessee. Knoxville, USA.
- Sawaya, M.A., Stetz, J, Clevenger, A.P., Gibeau, M.L., and S.T. Kalinowski. 2012. Estimating grizzly and black bear population abundance and trend in Banff National Park using noninvasive genetic sampling. 2012 PLoS ONE 7:e34777.
- Tredick, C.A., M.R. Vaughan, D.F. Stauffer, S.L. Simek, and T. Eason. 2007. Sub-sampling genetic data to estimate black bear population size: a case study. *Ursus* 18:179–188.
- Woods, J.G., D. Paetkau, D. Lewis, B.N. McLellan, M. Proctor, and C. Strobeck. 1999. Genetic tagging of free-ranging black and brown bears. *Wildlife Society Bulletin* 27:616–627.

Table 1. Bear hair collected at 121 barbed wire hair traps in the Chippewa National Forest during 6 sampling sessions in summer 2012.

Session	Trap sites with hair ^a	Total clusters ^b with hair	Total barbs with hair		
			Lower	Upper	Both
1	30 (25%)	149	206	92	298
2	63 (52%)	308	389	237	626
3	65 (54%)	279	318	152	470
4	79 (65%)	392	446	204	650
5	76 (63%)	303	321	127	448
6	64 (75%)	211	221	71	292
Total	377 (55%)	1642	1901	883	2784

^a Each hair-snare was checked in each of sessions 1 – 5. Snares that were never visited by bears during that period ($n = 36$) were dismantled prior to session 6.

^b Barbs with bear hair that were adjacent to each other, either on the same or different wires, were considered the same cluster, possibly representing a single bear entering or leaving a hair snare.

Table 2. Bears detected at hair traps in the Chippewa National Forest during 6 sampling sessions, summer 2012.

Session	Dates of hair collection	Different bears detected			New bears detected
		M	F	Total	
1	25 – 31 May	7	7	14	14
2	5 – 10 June	14	9	23	13
3	15 – 21 June	15	9	24	8
4	25 – 30 June	16	12	28	3
5	5 – 10 July	15	10	25	1
6	13 – 19 July	11	10	21	4

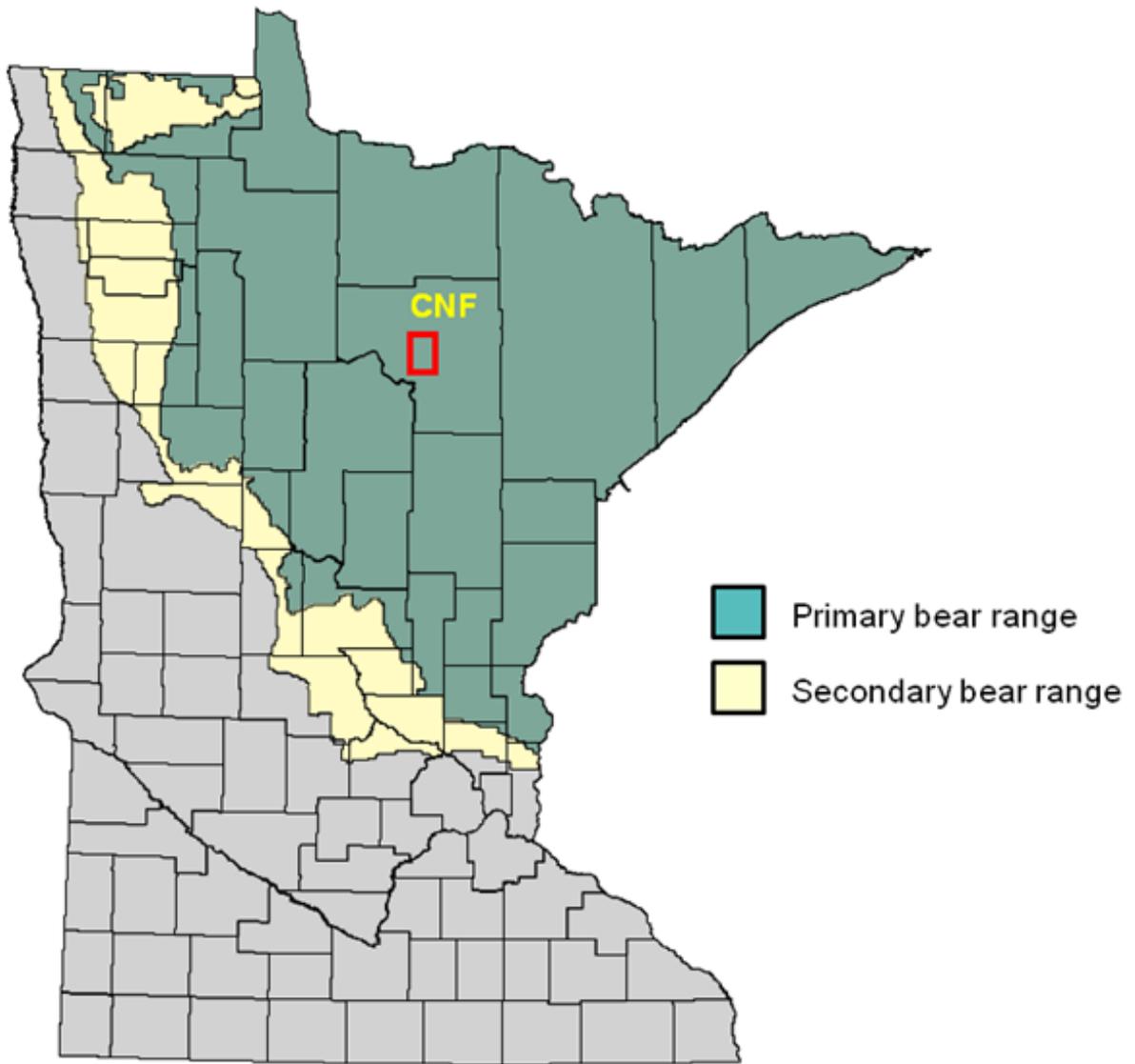


Figure 1. Location of hair-sampling study site in Chippewa National Forest, central bear range.



Figure 2. Set-up of barbed wire hair snare, showing 2 strands of barbed wire, central pile of bait and scent, and suspended bait and scent cup.

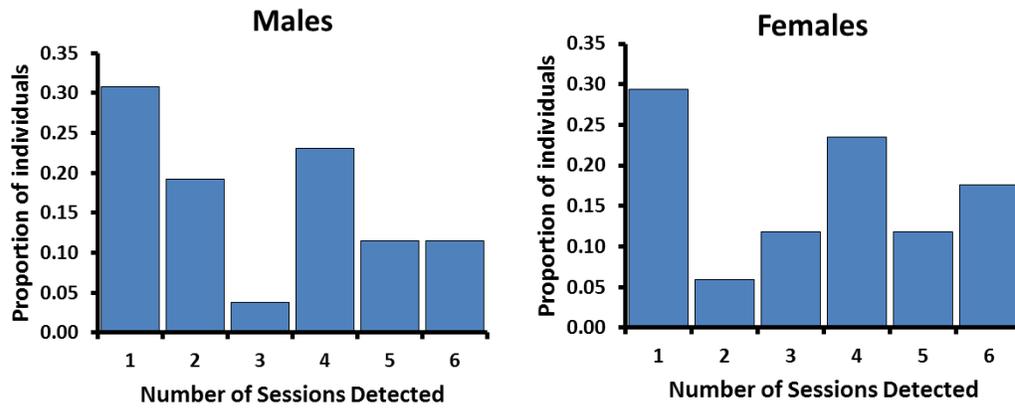


Figure 3. Number of sampling sessions in which black bears (26 M, 17 F) were detected at ≥ 1 hair trap during a DNA-based capture-mark-recapture study on the Chippewa National Forest, north-central Minnesota, 2012. Six 10-day sampling sessions ran from 25 May to 19 July.

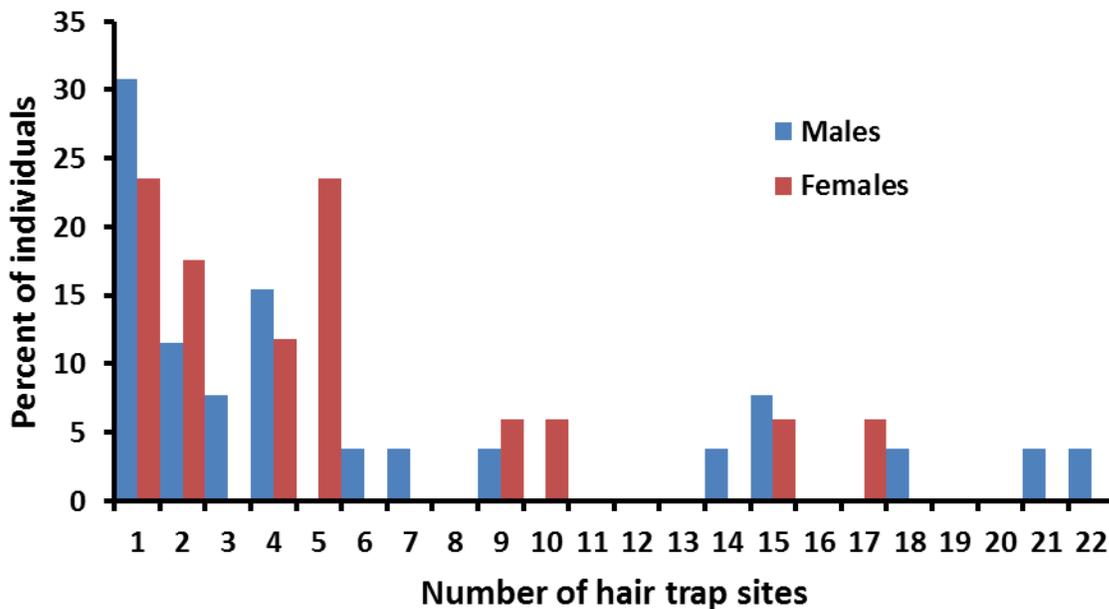


Figure 4. Number of hair trap sites at which individual male and female black bears (26 M, 17 F) were detected during 25 May – 19 July in the Chippewa National Forest, summer 2012. Hair traps were placed in a systematic grid at a density of $1/\text{mi}^2$.

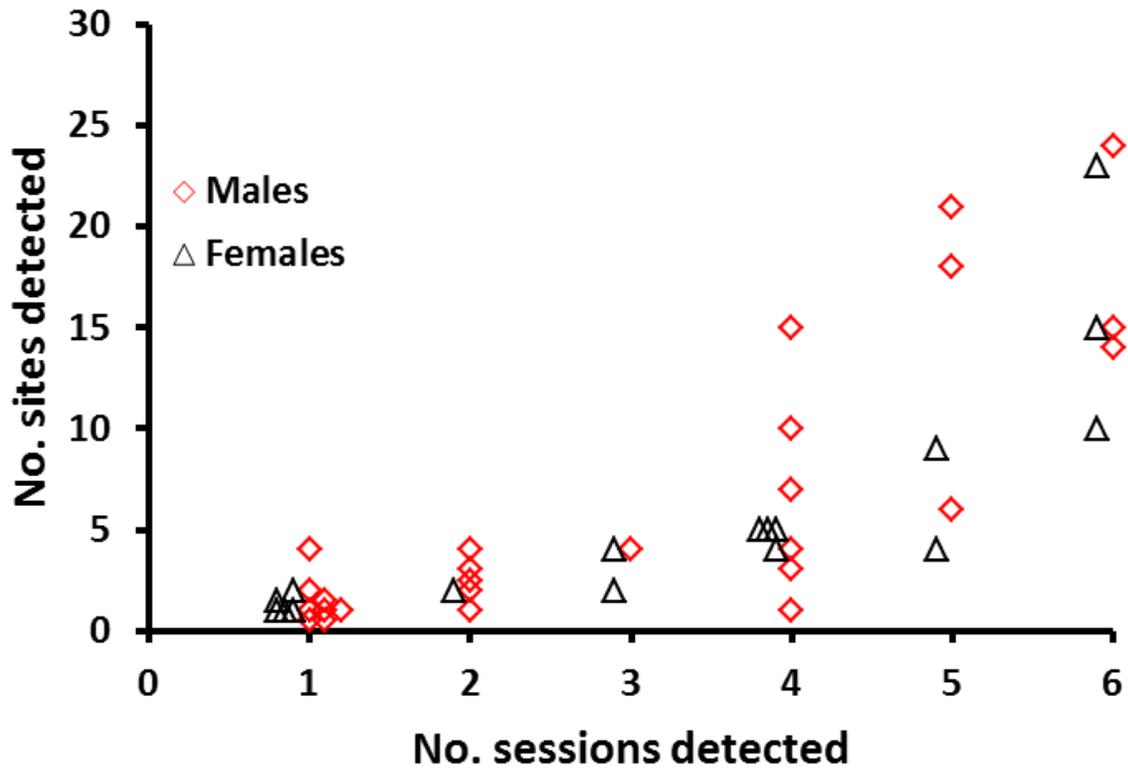


Figure 5. Relationship between the number of hair trap sites and the number of sessions in which individual male and female black bears (26 M, 17 F) were detected during 6 10-day sampling sessions in the Chippewa National Forest, north-central Minnesota, 2012.



Figure 6. Radiocollared and eartagged adult female bear entering a hair trap by going between wires (upper photo) and the same bear entering a hair trap by going below the lower wire (lower photo).

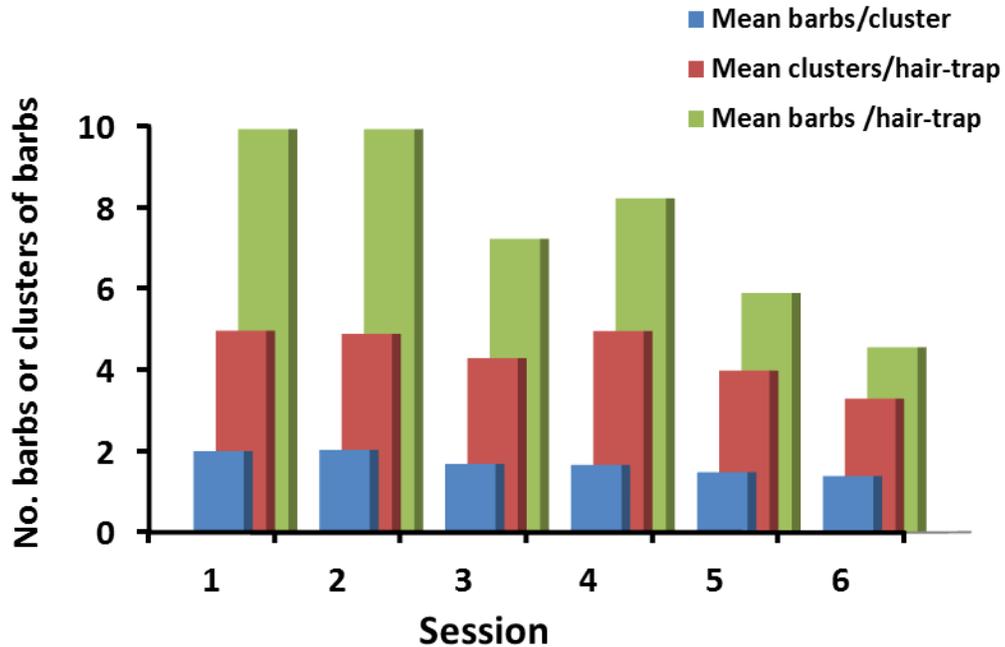


Figure 7. Changes through time in the mean number of barbs that snagged bear hair at barbed-wire hair traps (excluding sites with no hair), the number of clusters of adjacent barbs containing hair, and the number of barbs in each cluster during a DNA-based capture-mark-recapture study in the Chippewa National Forest, north-central Minnesota. Hair traps were checked 6 times, once every 10 days, from 25 May – 19 July, 2012.

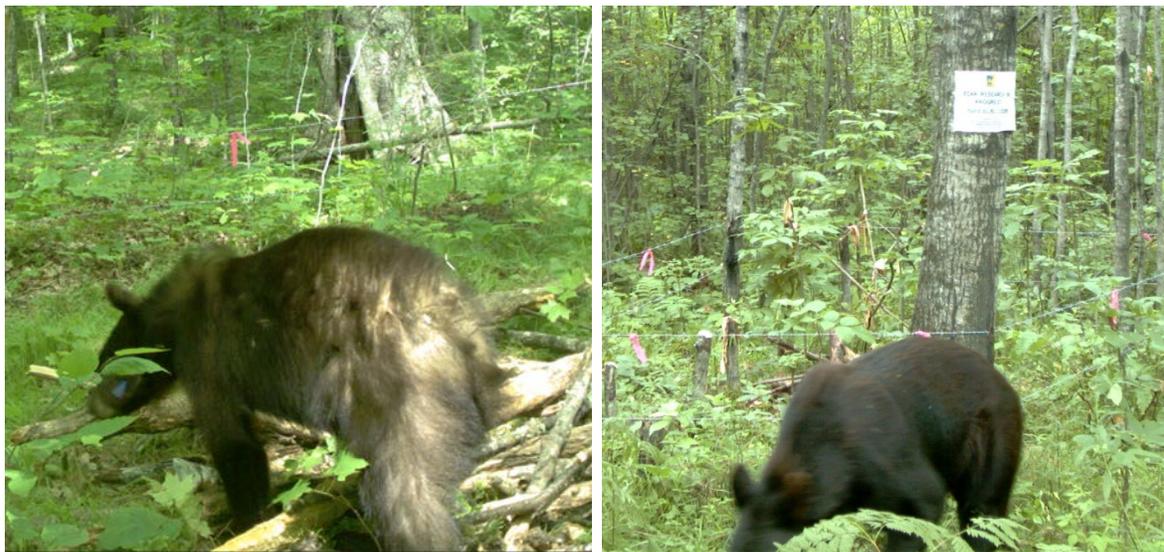


Figure 8. (Left) Bear in June still wearing its long winter coat: note the clumps of matted shed hair near the rump, ready to be snagged on the barbed wire. (Right) Bear in mid-July that has nearly finished molting its winter fur: notice areas where a thin layer of longer brown hair is still present over the new black shiny summer coat (which is harder to pluck).

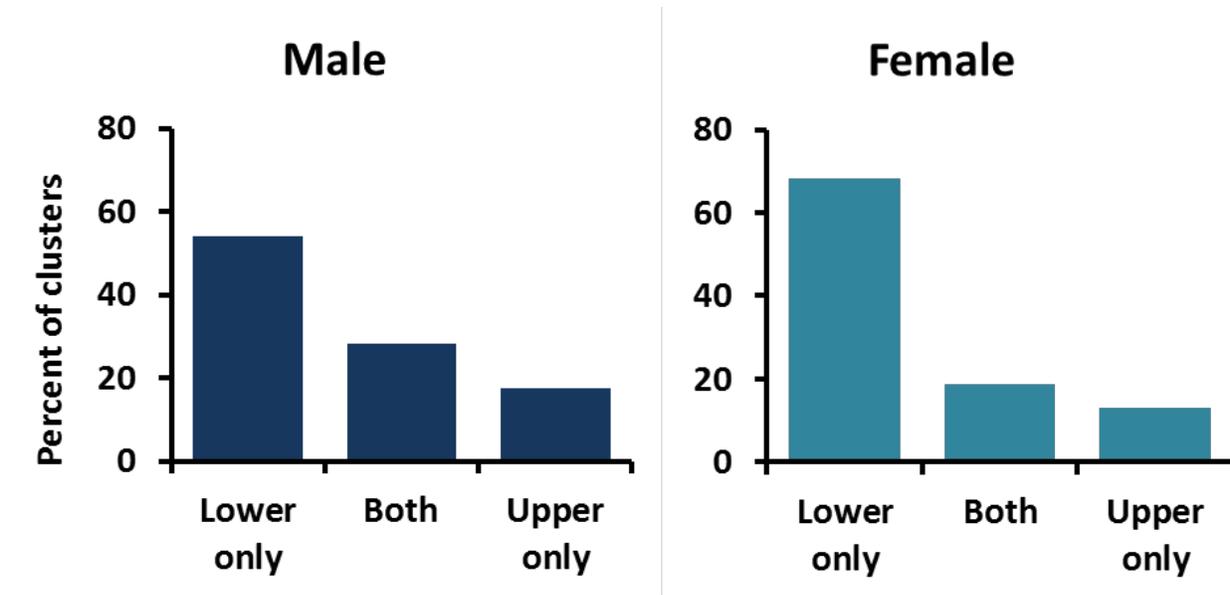


Figure 9. Percent of hair samples from barb clusters (adjacent barbs with hair) comprised of lower wires only, upper wires only, or both upper and lower wires. Only 1 barb from a cluster was genotyped, so clusters were categorized as male or female based on the 1 sampled barb (recognizing that some clusters could have included multiple bears). Males appeared more likely to leave hair on upper barbs only or both upper and lower wires than were females.

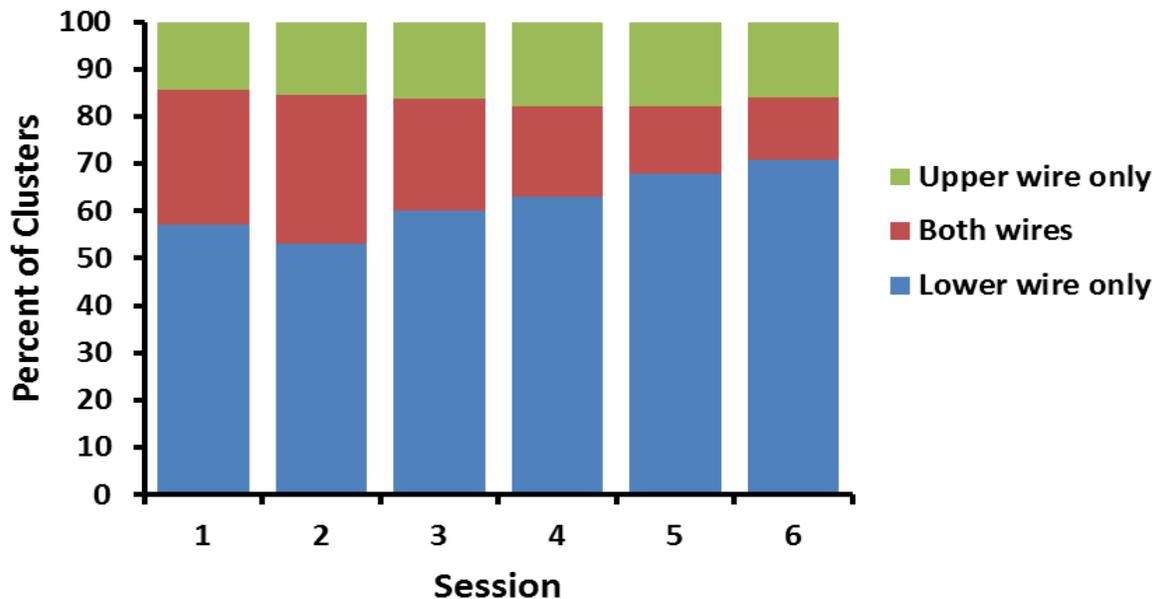


Figure 10. Changes through time in the relative proportion of bear hair samples that occurred in clusters comprised of adjacent barbs on only the lower wire, only the upper wire, or both wires of hair traps in the Chippewa National Forest, north-central Minnesota, 2012. Sampling sessions were 10 days long, spanning 25 May – 19 July.



Figure 11. Large bear stepping over the 75-cm high top wire to enter a hair trap at night. It is likely that its belly hair was caught on the wire.

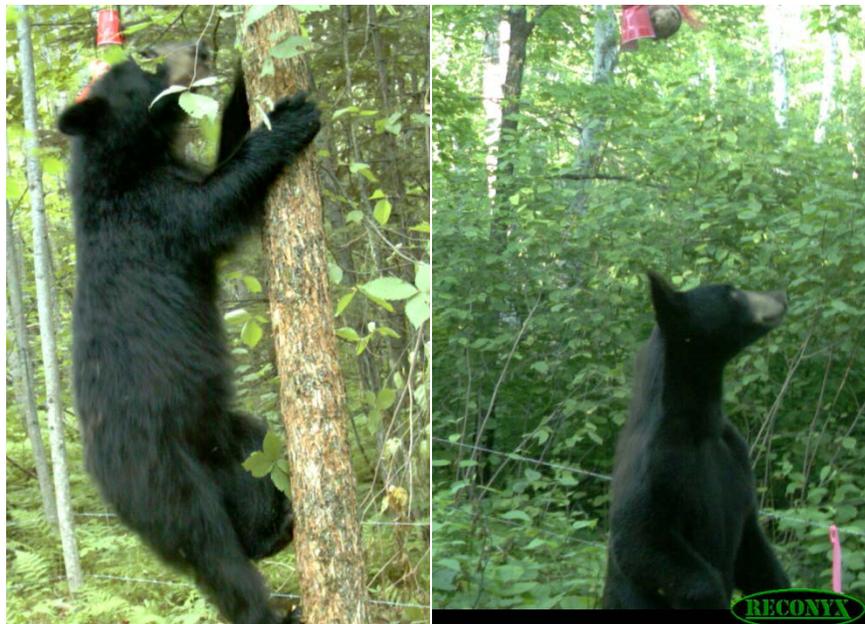


Figure 12. Two bears photographed only 1 week apart — 2 July (left) and 9 July (right) — exhibiting very different degrees of shedding.



DELINEATING THE ECOLOGICAL AND GEOGRAPHIC EDGE OF AN OPPORTUNIST: STRATEGIES OF THE AMERICAN BLACK BEAR (*URSUS AMERICANUS*) FOR THRIVING IN A FRAGMENTED, AGRICULTURAL LANDSCAPE

Mark A. Ditmer^{1,2}, David L. Garshelis, and Karen V. Noyce

FORAGING PATTERNS BY A FOREST-DWELLING SPECIES IN AN AGRICULTURAL LANDSCAPE: AMERICAN BLACK BEARS AT THE EDGE OF THEIR RANGE

Human-altered landscapes, including calorie-rich agricultural crops, may promote range expansion for omnivorous, opportunist species, such as the American black bear. Black bears live in the forest, but they are known to take advantage of human-related food sources. To better understand the causes and consequences of anthropogenic foraging, we studied the diet of an expanding population of black bears at the periphery of their range in an agriculturally-dominated region of northwestern Minnesota. We estimated the dietary contributions of various types of seasonally available foods; identified associations between consumption of agricultural foods and sex, age, body size, condition and reproductive status; and assessed relationships between use of agricultural foods and natural food abundance, and the juxtaposition of these food sources on the landscape. We used stable isotope analysis ($\delta^{15}\text{N}$ and $\delta^{13}\text{C}$) of 110 hair samples from 51 bears to estimate the relative dietary contributions of early season vegetation, ants, white-tailed deer, fruit and nuts, corn, and sunflowers (Figure 1). Males and females without cubs consumed appreciable fresh corn and sunflowers in fall (95% credible intervals: males = 19–46% of diet, females = 10–40%), and remnant corn in spring/summer (males = 9–23%, females = 3–16%); females with cubs and juvenile bears relied on crops much less (Table 1). Bear consumption of these crops as judged from isotopic analysis was positively correlated with the proportion of their GPS-collar locations in corn and sunflower fields. Weight and fat thickness were also positively correlated with use of crops for both males and females (Figure 2). Females, but not males, decreased crop consumption in years with high availability of natural fall foods and during years when they were rearing cubs (Figure 2). Male bears were not strongly influenced by landscape composition or proximity of cropfields, unlike females whose agricultural consumption was more closely tied to home range size, local crop availability and landscape composition (Figure 3). Male use of crops increased with increasing physical size, suggesting that a degree of dominance and boldness was associated with the use of this high-calorie resource (Figure 2). Overall, males and females exhibited very different levels of risk-aversion, with male bears more likely to seek out anthropogenic food sources associated with a high risk of human-caused mortality. We found that two crops, corn and sunflowers, had a profound effect on the behavior of bears, despite comprising only ~2% of the land area.

BEHAVIORAL AND PHYSIOLOGICAL RESPONSES OF BLACK BEARS TO NATURAL AND ANTHROPOGENIC FEATURES IN AN AGRICULTURALLY-DOMINATED LANDSCAPE

Human activities and variation in habitat quality and configuration have been shown to influence space use patterns in many species, but few studies have documented the physiological responses of free-roaming animals to these factors. We combined remote bilogger technology, capturing continuous heart rate values, with locational data from GPS collars to investigate the behavioral and physiological reactions of American black bears to a landscape dominated by agriculture (52.5% areal cover) in northwestern Minnesota. Our study occurred at the edge of the range of this species, with small, scattered patches of forest within a mosaic of crop fields and an extensive road network. However, only ~2–4% of the area

¹ Department of Fisheries, Wildlife, and Conservation Biology, University of Minnesota, St. Paul

² This report represents abstracts from three chapters of Ph.D. thesis

contained crops that bears consumed (corn, sunflowers, oats). We used GPS points to locate the habitat that bears occupied, and to estimate their rates of travel. Heart rates increased with movement rates, rising by over 30% from resting to the quickest travel speeds (Figure 4). We used a modeling approach to distinguish factors that influenced heart rates independent of movement rates. Bears commonly crossed agricultural areas that provided no food or cover and their heart rates, elevated beyond what was expected from their movement rate, were indicative of a stress response. Conversely, edible crops provided multiple metabolic benefits in the form of dense calories, reduced movement, and also reduced heart rates, suggesting that bears foraging there felt at ease. Unexpectedly, female bears had low heart rates and low levels of activity in the most fragmented patches of natural habitat, possibly a sign of human-avoidance behavior or a reaction related to crossing roads. Bears also had relatively low heart rates, given their high rate of travel, at the periphery of their home range, suggesting they were traveling but not foraging. During fall, as bears prepared for hibernation, their heart rates declined 2 weeks before their movements slackened (Figure 5, 6), evidence that their metabolism slowed to enhance fat accumulation; they also shifted to a nocturnal activity pattern (Figure 7), likely to reduce exertion during the heat of the day. The use of a physiological monitor provided new insights on bear biology and ecology that would not have otherwise been apparent.

DELINEATING THE ECOLOGICAL AND GEOGRAPHIC EDGE OF AN OPPORTUNIST: HOME RANGE AND CALORIC REQUIREMENTS OF AMERICAN BLACK BEARS IN A FRAGMENTED, AGRICULTURAL LANDSCAPE

Animals living at the edge of their geographic range often exhibit exceptional physical and behavioral attributes. American black bears living in northwestern Minnesota, the western edge of the eastern black bear's geographic distribution, are the largest and most fecund bears in Minnesota despite, or potentially because of, the agricultural-dominated and fragmented area they inhabit. We estimated annual home ranges; determined how age and gender, roads, habitat, weather, crop types and caloric availability affect weekly home range size. We then used these results to delineate the limits of bear range. Annual home ranges from 60 bear-years (33 female, 27 male) using 29 individual GPS-collared bears (14 female, 15 male) showed that male ranges were 5.7–9.7x larger than females' — the home ranges of these males were the largest ever recorded for the species (Table 2). We calculated weekly home ranges, utilizing 80,441 locations from 38 individual animals (17 female, 21 male) over 75 bear-years (37 female, 38 male, Figure 8). Sizes of weekly ranges were positively associated with high road density and traffic volume, and high-levels of agriculture; they were negatively related to caloric density on the landscape. Oak habitat and crop fields of corn, sunflowers, and oats were associated with smaller home ranges during summer and fall (Figure 9). This negative correlation between amount of crops and home range size was stronger for males. Future range expansion will be dictated by food and cover. Bears required less natural cover in summer and fall months than during spring (Figure 10). Males required less natural cover, so they could potentially range farther west than females (Figure 10). Nevertheless, significant expansion appears unlikely due to limited natural cover and the distance between the natural forage available in the forests and shrublands to the east and the western periphery. Suitable female habitat contracted with consistently low annual availability of food resources, but the presence of crops does not appear to be required for their persistence in the region (Figure 11). Males and females exhibited different strategies for living in this region: males high-graded the landscape by constantly moving among patches of natural habitat during spring through mid-summer but then slowed as they sought out and consumed calorically dense crops. In contrast, females were less risk tolerant and instead sought out home ranges that provided a minimal threshold of calories from natural foods. Overall, bears' annual home range size was not a good metric of habitat quality. By investigating drivers of short-term home range size and the minimal requirements in a landscape we were able to project range limits and understand how bears thrive in a region that was once considered marginal bear habitat.

Table 1. Population estimates of diet for spring/summer and fall food sources broken down by demographic groups found to have a strong influence on individual dietary estimates of crop consumption. Annual estimates of diet from crops were calculated by multiplying lower and upper values of corn and sunflowers by 2/3rds for the spring/summer estimates and 1/3rd for the fall.

Food Source	Adult Males	Juvenile males and females	Adult females without cubs	Adult females with cubs
Spring/Summer mixing model (2/3rds distal end)				
Ants	0-11%	2-12%	0-13%	0-7%
Corn	9-23%	0-7%	3-16%	0-11%
Deer	0-3%	0-2%	0-3%	0-2%
Spring vegetation and summer fruit	70-85%	84-94%	75-90%	84-95%
Fall mixing model (1/3rd root end)				
Sunflowers	0-7%	0-5%	0-10%	0-2%
Corn	19-39%	1-11%	10-30%	0-11%
Fall fruit and nuts	59-75%	87-96%	67-83%	88-99%
Annual Diet from Crops	12-31%	0-10%	5-24%	0-12%

Table 2. Average annual home range size (km²) (95% CI) of GPS-collared bears in northwestern Minnesota based on 33 female and 27 male bear-years, 2007–2013. Only fixes separated by 4 hours or more were included.

Sex	Method ^a	50% isopleth	95% isopleth	100% isopleth
Female	MCP	21.6 (8.3 - 34.9)	72.5 (45.5 - 99.5)	115.9 (80.5 - 151.2)
	Rule Based	18.1 (10.4 - 25.8)	77.8 (51.7 - 103.8)	---
	H-Ref	19.1 (11.5 - 26.7)	91.4 (54.5 - 128.3)	---
	k LoCoH	2.6 (1.6 - 3.6)	30.3 (18.5 - 42.1)	---
Male	MCP	172.8 (107.3 - 238.3)	616.9 (466.1 - 767.7)	861.4 (639.0 - 1083.8)
	Rule Based	168.6 (114.4 - 222.7)	754.0 (550.7 - 957.3)	---
	H-Ref	172.6 (118.7 - 226.6)	834.1 (596.3 - 1071.9)	---
	k LoCoH	7.6 (4.8 - 10.4)	172.9 (129.8 - 215.9)	---

^a MCP = minimum convex polygon, Ad-hoc = rule-based approach using an ad-hoc method for selecting a bandwidth, H-ref = kernel density estimator with reference bandwidth for H, k LoCoH = fixed k nearest neighbor convex hull.

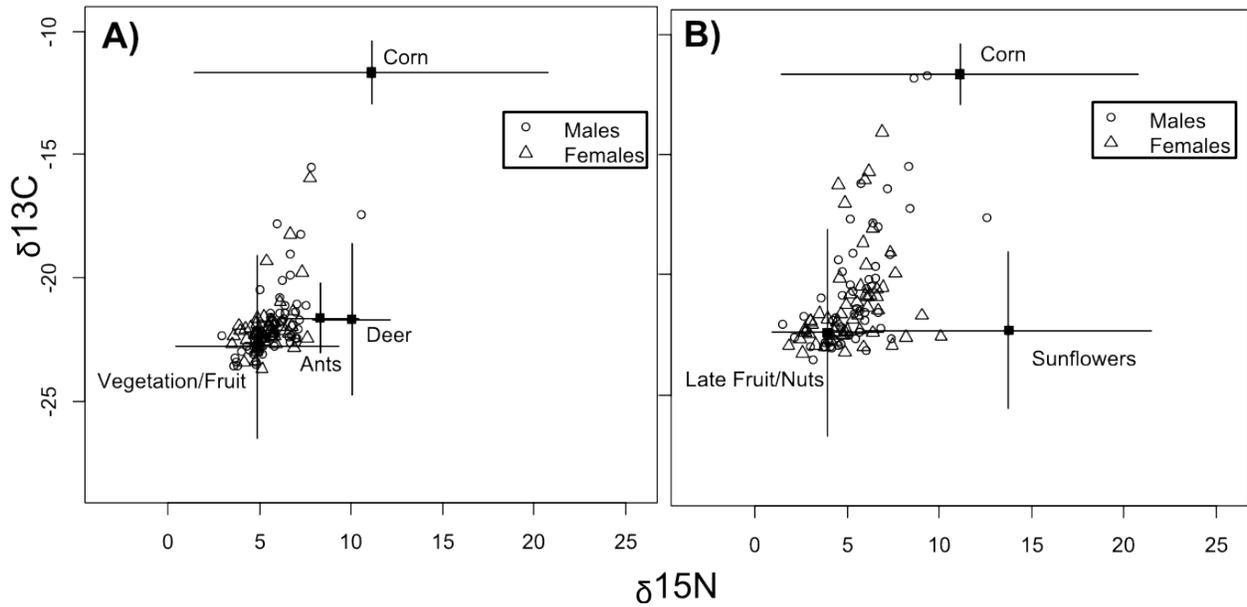


Figure 1. Isotopic signatures of bear-hair compared to bear foods. Bear hairs represent 56 female and 54 male bear-years. Food sources available for consumption are from (A) spring until ~mid-August and (B) ~mid-August until winter in northwestern Minnesota. Isotopic values of hair samples correspond to the distal 2/3rds end of the hair (A) and the 1/3rd root end of the hair (B).

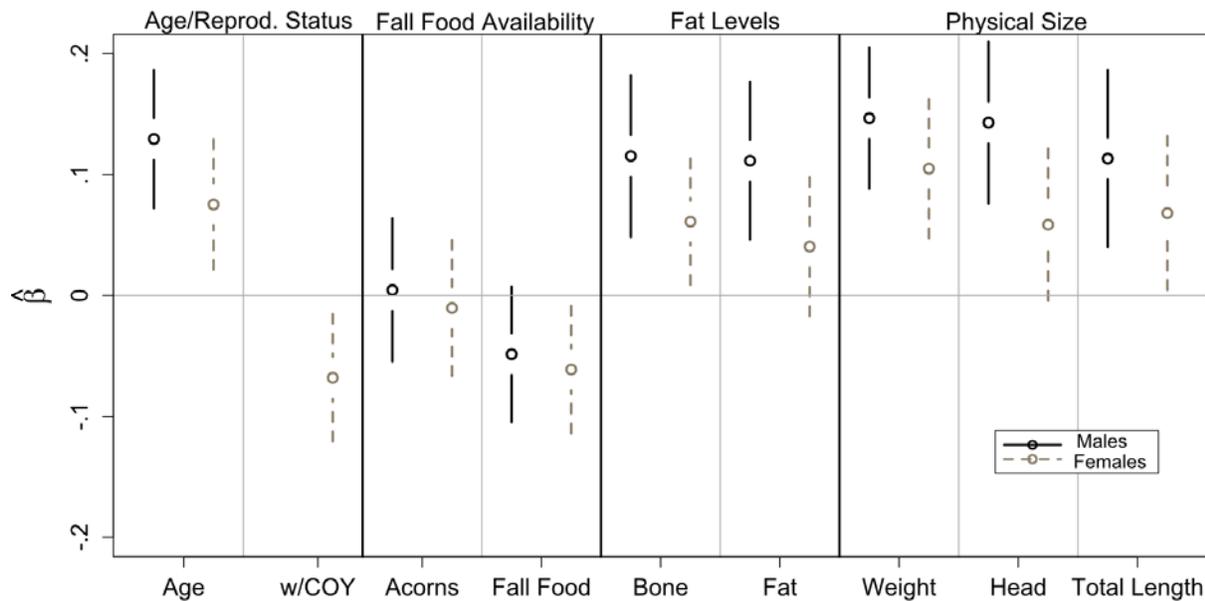


Figure 2. Parameter estimates ($\hat{\beta}$), with 95% confidence intervals, from linear mixed model regressions between estimates of individual bears' consumption of crops and covariates relating to food availability and physical status of animals. Covariates were scaled and centered for comparison purposes.

Covariates:

Age: Age of bear

w/COY: With cubs of the year

Acorns: Estimate of annual acorn mast (acorns/m²)

Fall Food: Indices of primary fall foods (hazelnut, dogwood, oak)

Bone: Bone prominence of bear (measure of amount of body fat)

Fat: Skin-fold thickness (measure of amount of body fat)

Weight: Weight of the bear at time of handling

Head: Head girth

Total Length: Total length (from nose-tip to tip of tail)

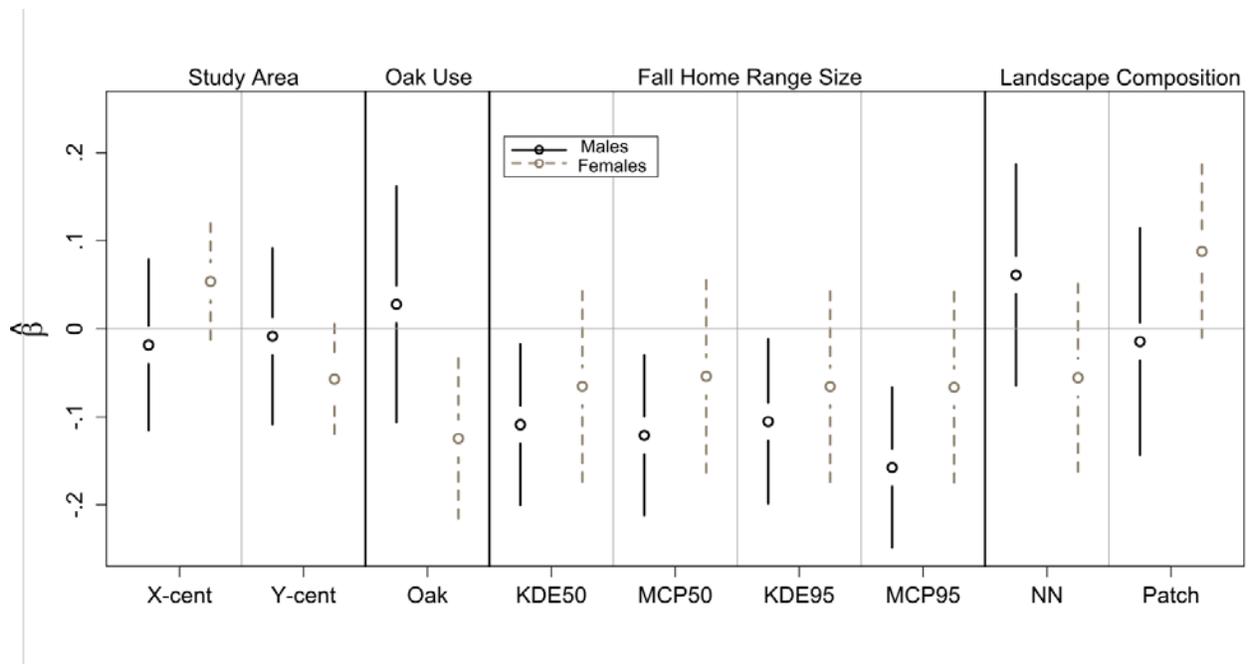


Figure 3. Parameter estimates ($\hat{\beta}$) with 95% confidence intervals, derived from linear mixed model regressions relating estimate of individual bears' consumption of crops with spatial covariates based on GPS data from radio-collars. Covariates were scaled and centered for comparison purposes.

Covariates:

X-cent: Center of activity (east - west) based on trap location, kill site, den or home range centroid

Y-cent: Center of activity (north-south) based on trap location, kill site, den or home range centroid

Oak: % of GPS locations in oak and upland shrub habitats

KDE50: 50% fixed kernel density estimate of fall home range area

MCP50: 50% minimum convex polygon estimate of fall home range area

KDE95: 95% fixed kernel density estimate of fall home range area

MCP95: 95% minimum convex polygon estimate of fall home range area

NN: Average shortest distance between each natural habitat patch with a home range (habitat connectivity estimate)

Patch: Mean size of natural habitat patches within a fall home range (habitat connectivity estimate)

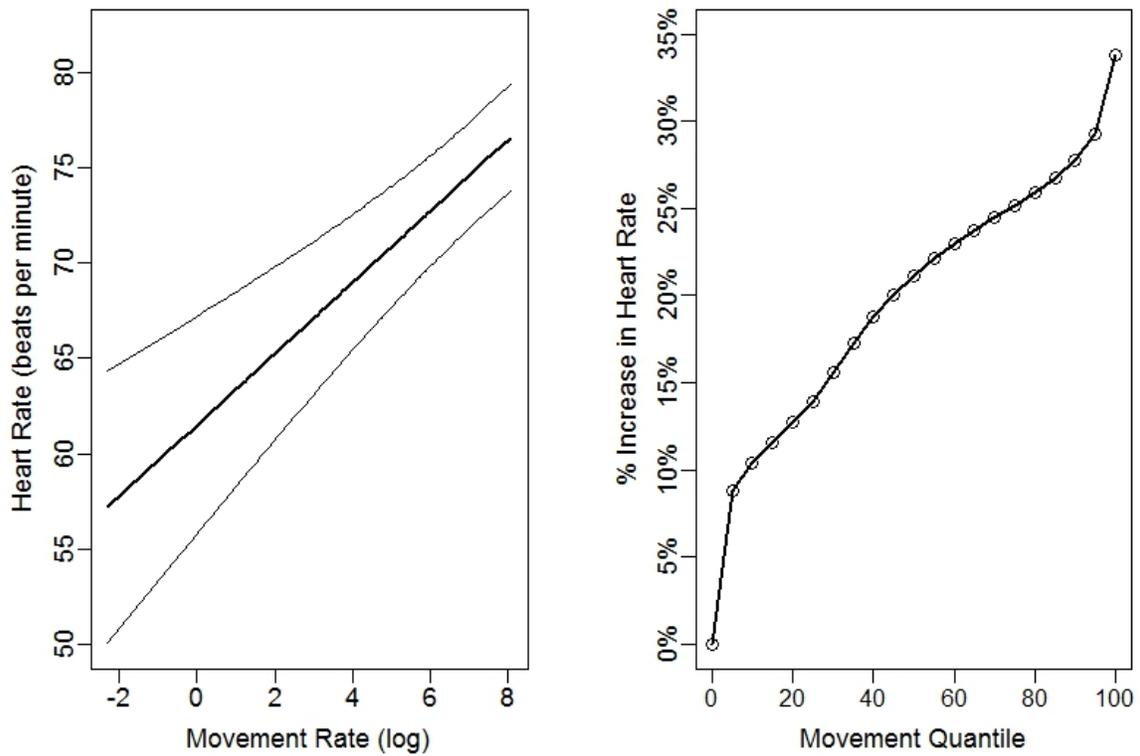


Figure 4. (A) Influence of movement rate ($\log(m/\text{hour})$) on heart rate (bpm, \pm 95% CI) of free-roaming black bears in northwestern Minnesota, based on the best-approximating population model with all other covariates held constant at their mean values. (B) Percent increase in mean estimated heart rate with quantile change in log movement rate.

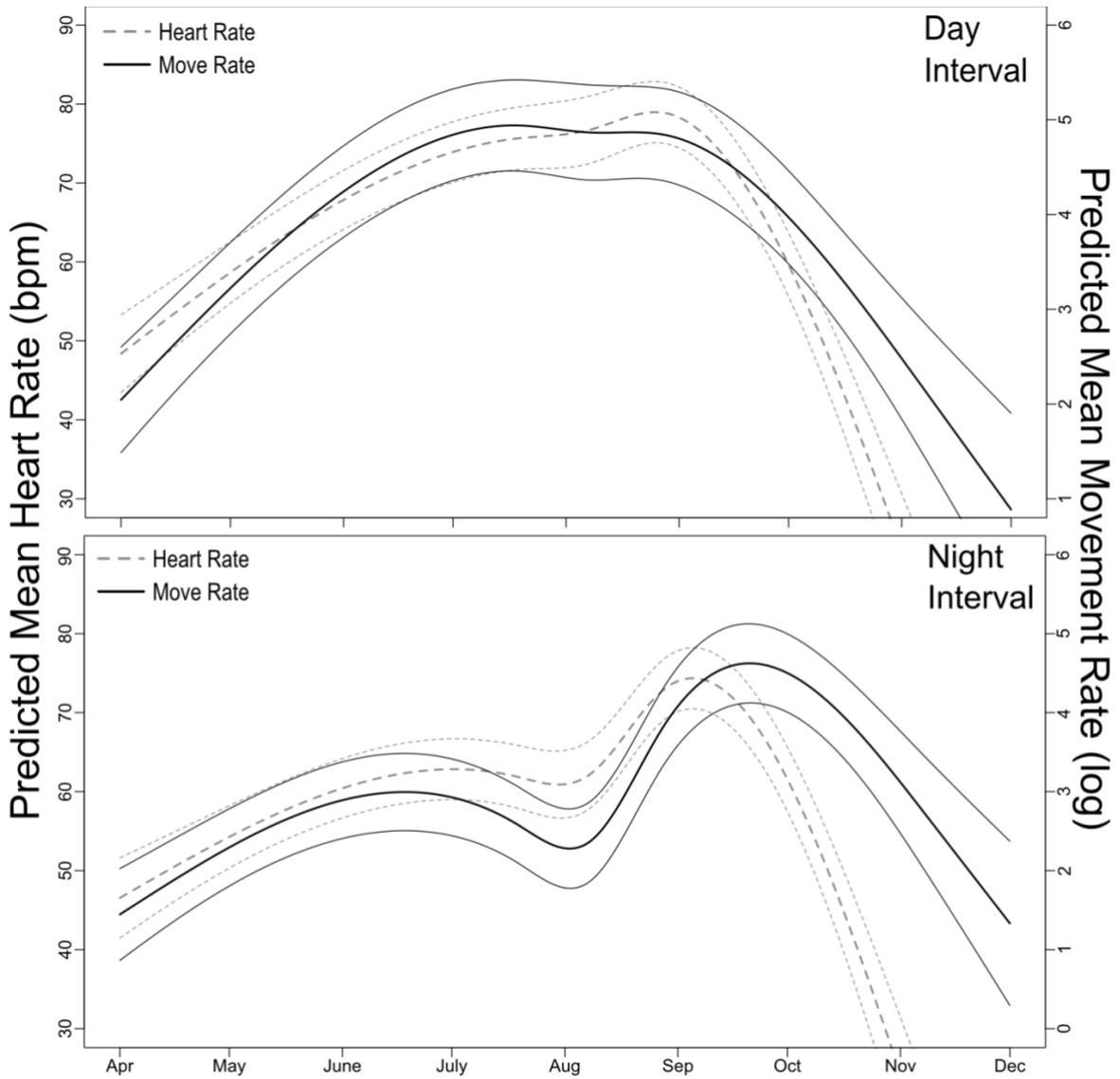


Figure 5. Estimates of mean (A) daytime and (B) nighttime heart rate (bpm, \pm 95% CI) versus mean log movement rate (m/hour; \pm 95% CI) for northwestern Minnesota black bears over the course of a year, 2009–2012.

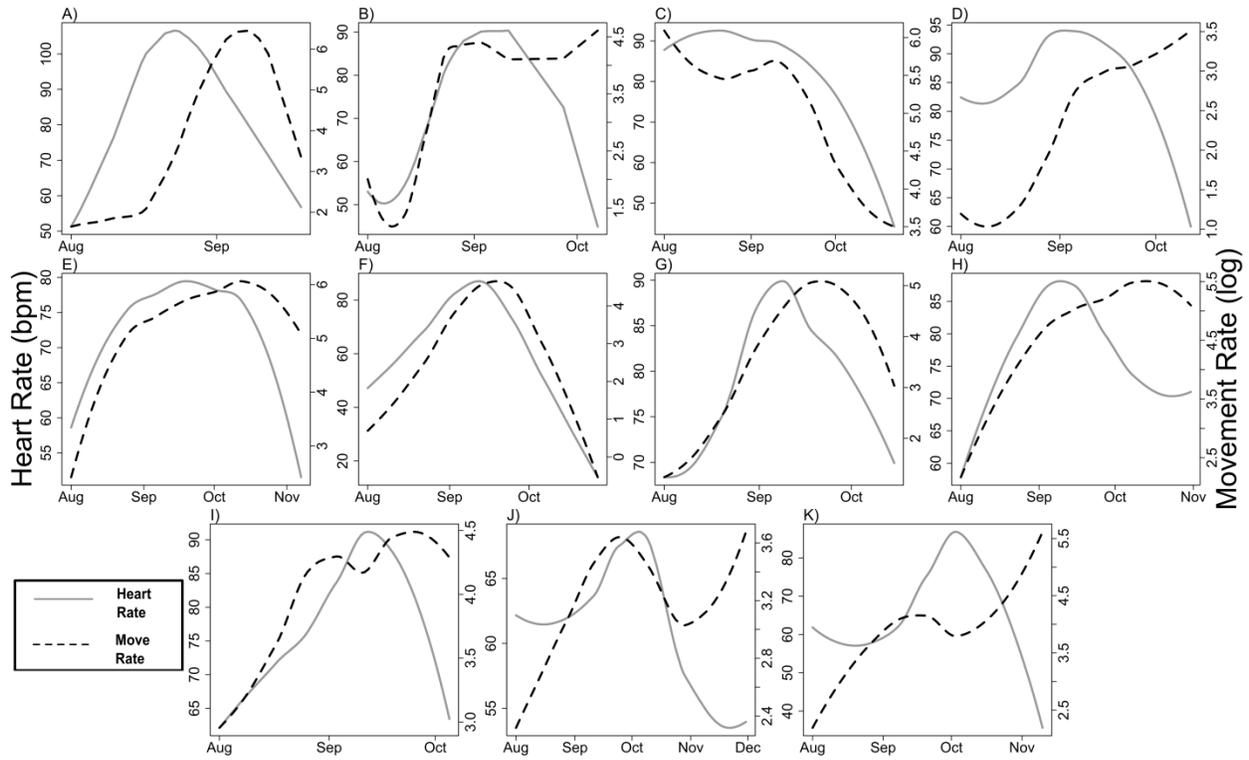


Figure 6. Loess smoothed data of nighttime heart rate (bpm; primary y-axis) and movement rate (log; secondary y-axis) for individual northwestern Minnesota black bears during autumn (August – den entry). All bear-years exhibited the pattern of a reduced average daily nighttime heart rate prior to reduced daily nighttime movement rates except for the largest bear in the study (C).

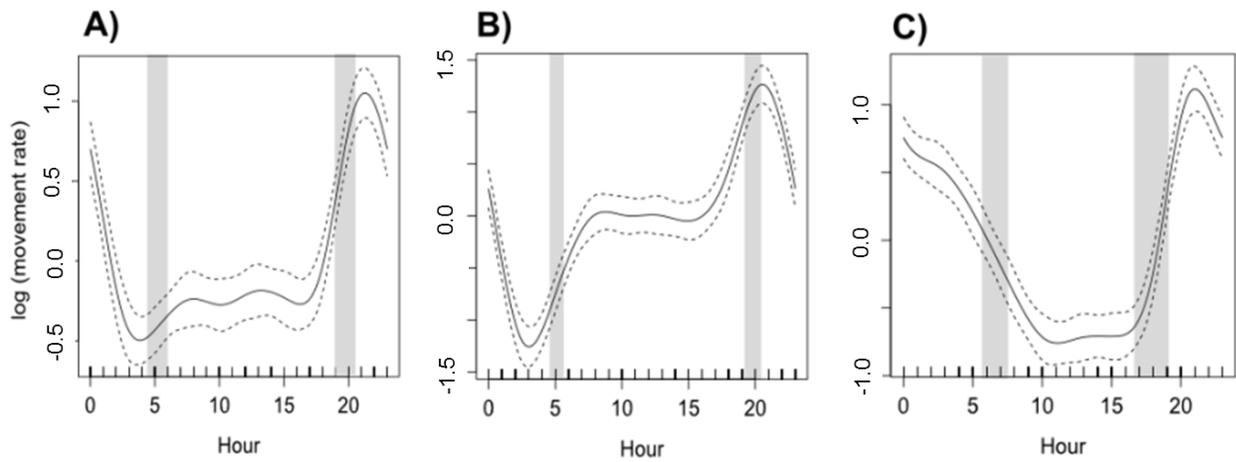


Figure 7. Influence of hour of day on log movement rate (\pm 95% CI) of northwestern Minnesota black bears during (A) spring, (B), summer, and (C) fall, 2009–2012. Gray shaded areas represent the approximate range of sunrise and sunset.

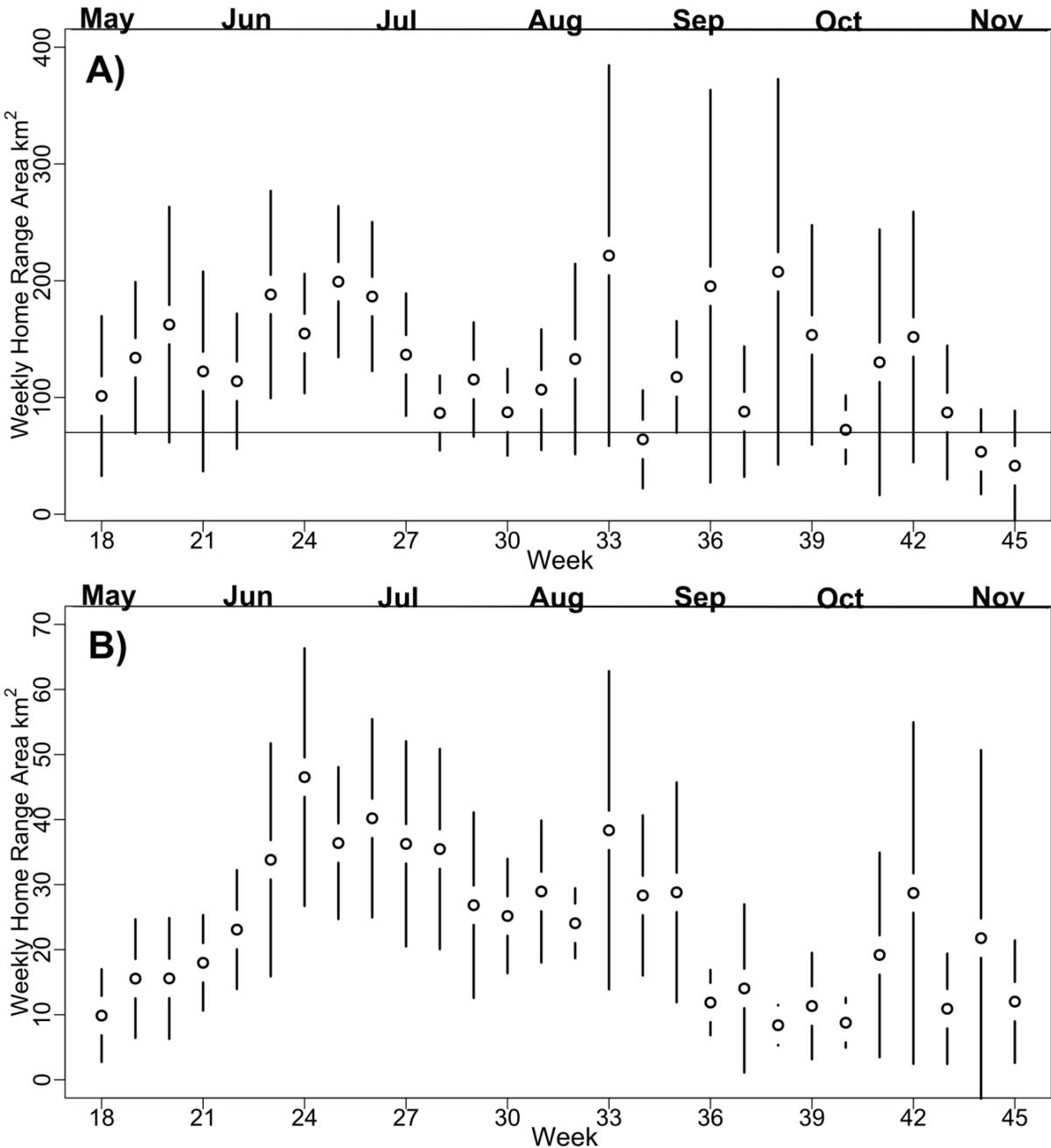


Figure 8. Average weekly home range (WHR) estimates and 95% confidence intervals (km^2) of GPS-collared (A) male and (B) female black bears in northwestern Minnesota, 2007-2013. Seventy-five bear-years (37 female, 38 male) were used to construct 1,453 weekly home range estimates using the 95% isopleth with the rule based method of kernel estimation. Averages and the corresponding confidence intervals were calculated using the point estimates of WHRs for each week. The gray horizontal line in the upper panel represents the maximum value of the females' y-axis (70 km^2) superimposed on the male data.

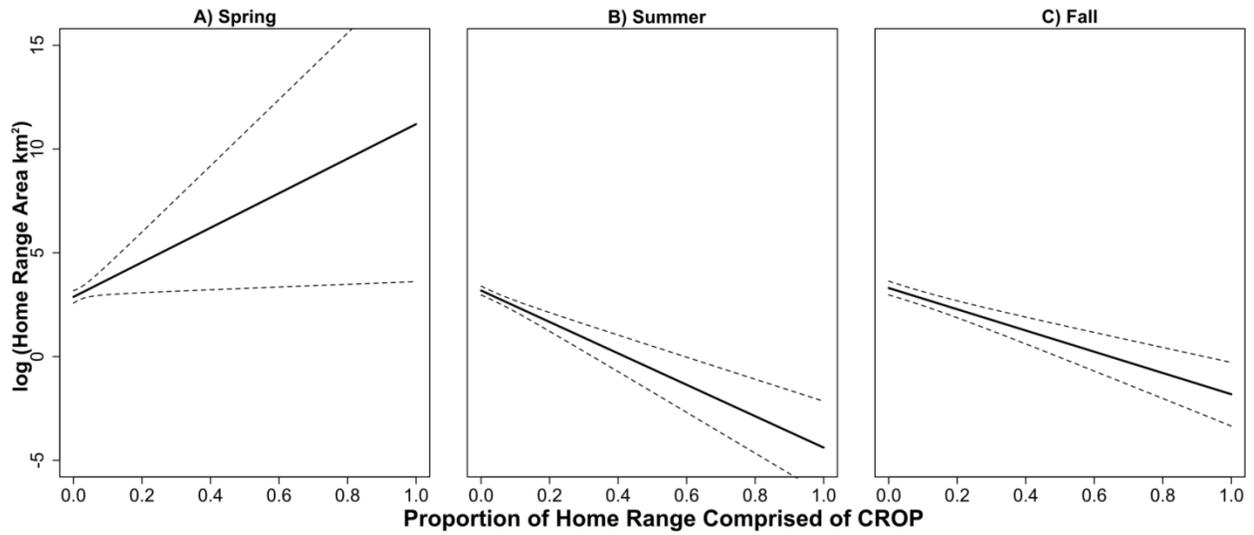


Figure 9. Predicted changes in weekly home range size ($\log \text{ km}^2$) for northwestern Minnesota black bears in response to changes in the proportion of edible crops within their home range during spring (left: ~May 1 – July 1), summer (middle: ~July 1 – Sept. 1) and fall (right: ~Sep 1 – Dec 1).

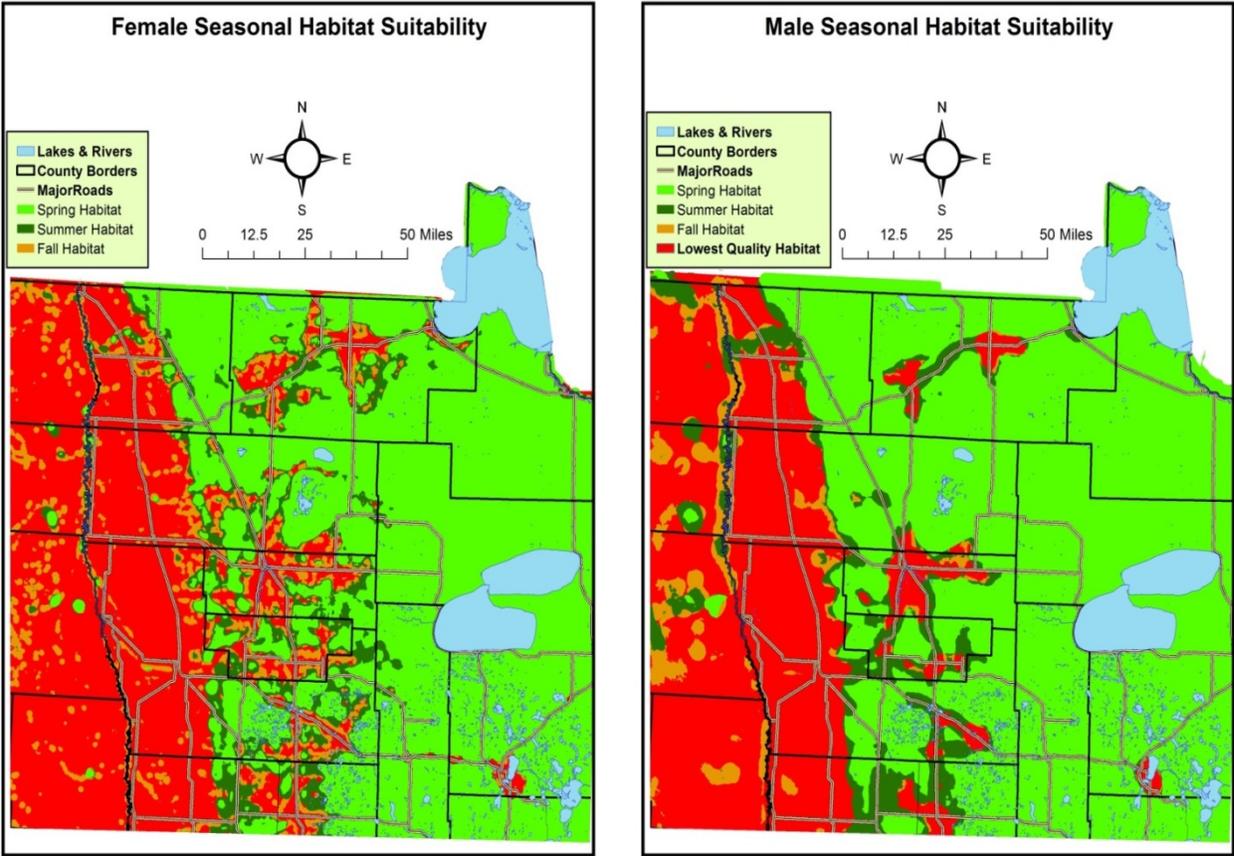


Figure 10. Projected maximum habitat suitability extent during spring (May 1 – July. 1), summer (July 1 – Sept. 1), and fall (Sep 1 – Dec 1) for female (left) and male (right) bears living in northwestern Minnesota and (potentially) eastern counties of North Dakota. The map illustrates the expansion of usable habitat throughout the year. Spring had the least overall coverage so is shown as the top layer with summer and fall maps below. Any areas showing fall habitat are only available during fall whereas all spring area is available during summer and fall.

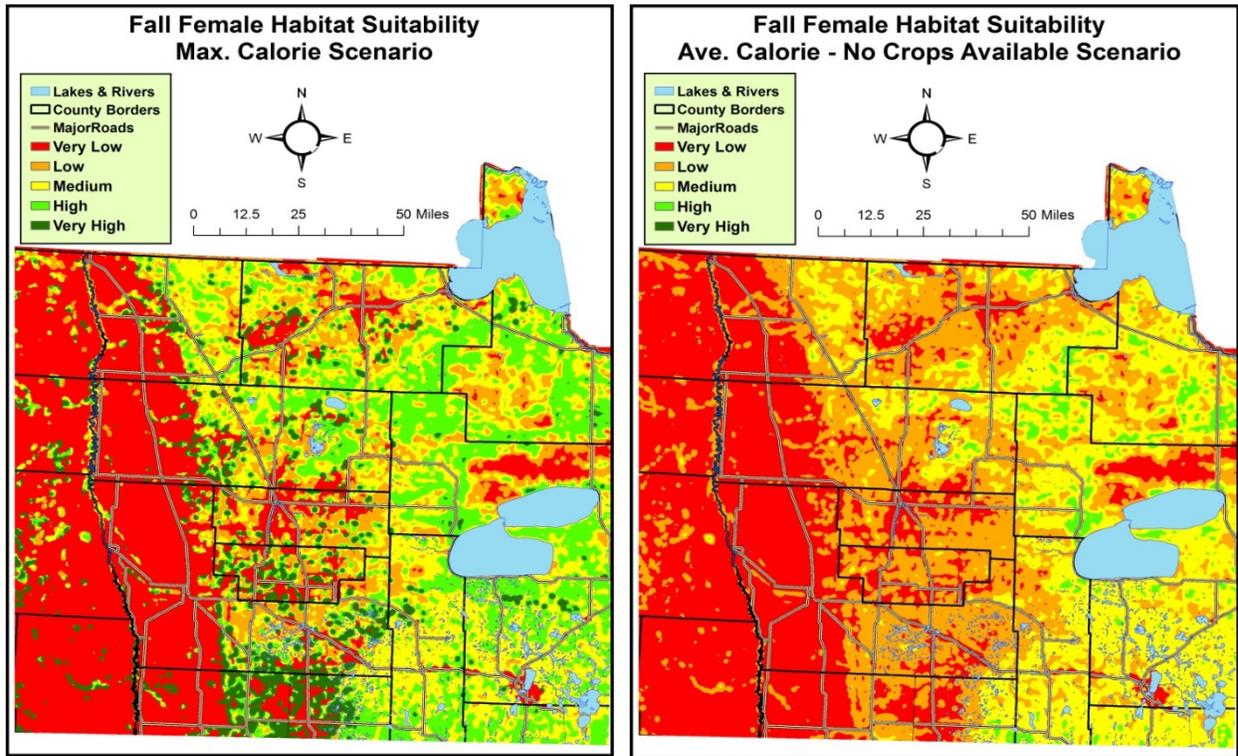


Figure 11. Projected habitat suitability during fall (Sep 1 – Dec 1) for female bears living in northwestern Minnesota and (potentially) eastern counties of North Dakota for years with high natural food conditions and crop availability(left panel) versus average natural food conditions with no edible crops available (right panel). Maps incorporate natural cover requirements, the influence of roads, and caloric values based on weekly home ranges of GPS-collared bears.



REPRODUCTIVE ECOLOGY OF FISHERS AND AMERICAN MARTENS 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 2008. Including the pilot year of the study, a total of 232 martens (111F, 121M) and 101 fishers (58F, 43M) have been radio-collared. To date, age and reproductive status have been confirmed on 40 adult (≥ 2 years old) female martens. Pooling years, pregnancy rate has been ~57% for 2-year-old martens, and ~85% for martens 3 years or older. Average size of 24 marten litters is 2.92 (range = 1-4), with minimal difference between litter size of 2- versus 3+ year-old females. Based on initial data, it appears marten kits are typically born in late-April through mid-May. A total of 45 marten natal or maternal dens have been located, of which 64% have been in tree (primarily aspen (*Populus tremuloides*) and cedar (*Thuja occidentalis*)) cavities and 36% in underground burrows. We have also confirmed both age and reproductive status for 51 female fishers. Pooling years, pregnancy rate has been ~57% for 2-year-old fishers, and ~93% for fishers 3 years or older. Average size of 37 fisher litters is 2.5 (range = 1-4). Data suggests that litter size for 2-year-old fishers is lower than for older females (2.2 versus 2.7). Based on data collected to date, it appears fisher kits are typically born in early to mid-March, or ~1.5 months earlier than marten kits. A total of 56 fisher natal or maternal dens have been confirmed, all but 2 being in elevated tree cavities. Cavities have been located in both live trees (74%) and snags (26%) with an overall average DBH of 20.5 in. Tree cavities used by female fishers have been located primarily in aspen (68.5%) (*Populus tremuloides*, *Populus grandidentata*) and oak (15%) (*Quercus* spp.) trees. Most female fishers appear to move kits from their natal den to at least 2 different maternal dens prior to June 1.

INTRODUCTION

American marten and fisher are native to Minnesota, but reliable documentation of their historic distribution is limited. Undoubtedly, northeastern Minnesota was a stronghold for the marten population, though notable numbers likely occurred in the northern border areas as far west as Roseau County. Limited information suggests they occurred as far south as Crow Wing County and as far southwest as Polk County. As a result of unregulated harvest, marten were considered rare in Minnesota by 1900, and extensive logging and burning around the turn of the century further contributed to the near extirpation of marten from Minnesota by the 1930s (Swanson et al. 1945). Fishers in Minnesota appear to have historically occupied a larger geographic area than martens, extending further south and west into the hardwood dominated transition zone, including southeast Minnesota (Swanson et al. 1945, Balsler 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 now 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). Over the past 10 years, fisher

abundance and harvest have been increasing along the southern and western edge of the 'forest zone' where forest historically transitioned to savanna and prairie and is now characterized by linear forest corridors (streams/rivers) or smaller forest patches interspersed with agriculture. Conversely, fisher abundance appears to have declined significantly over the same period in the core forested areas of north-central and northeast Minnesota. Peak statewide harvest levels have been near 4,000 and 3,500 for marten and fisher, respectively. However, due to apparent multi-year population declines for both species, harvest seasons from 2007 to the present have become progressively more conservative, with the 2013 harvest season lasting only 6 days with a combined fisher/marten limit of 2 per trapper.

While both species appear to have naturally re-colonized a significant portion of their historic range, Minnesota-specific information on reproductive ecology is limited to carcass (corpora lutea, placental scar) data collected from harvested animals primarily from 1985-90 (Kuehn 1989, Minnesota DNR unpublished data). Reproductive data is also available from other geographic areas, but questions remain on the accuracy of various methods to assess reproduction, and the amount of spatial and temporal variation in reproductive parameters. Minnesota-specific data on structures and sites used by fisher for natal and maternal dens is also lacking.

Martes pregnancy rate and litter size data are generally quantified from 1 of 4 methods: counts of corpora lutea (CL) in ovaries; counts of blastocysts (BC) in uteri; placental scar (PS) counts; or direct observation of litter size (Gilbert 1987, Mead 1994). Assuming both species are induced ovulators (but see Cherepak and Connor 1992, Frost et al. 1997), CL counts should accurately reflect copulation and ovulation rates, but all CL persist even if only 1 ovum is fertilized. Blastocyst counts reflect the number of fertilized ova, but not all BC may implant in the uterus and develop, and BC are often destroyed in poorly preserved carcasses. Hence, these 2 measures may not only overestimate litter size for parous females, but may also overestimate parturition rate (i.e., females may ovulate, 1 or more ova become fertilized, yet they fail to ultimately den and give birth). Placental scars, formed last in the reproductive process, would seem the most reliable carcass-based estimate of parturition rate and litter size. However, several authors (Gilbert 1987, Payne 1982, Strickland and Douglas 1987) have suggested that PS may not always persist long enough in mustelids to be detected during the harvest season when carcasses are easily collected. Furthermore, PS can persist in some species even if fetuses are resorbed (Conaway 1955) and detection and counts of PS 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 martens. This is within the range of average litter sizes reported from ovary or uterine analysis (~ 2.5 – 3.5; Strickland et al. 1982, Strickland and Douglas 1987, Flynn and Schumacher 1995, 2009, Aune and Schladweiler 1997, MN DNR unpublished data). For fishers, the same appears to be true, with an average litter size of 2.8 from 60 captive fisher litters (reviewed in Strickland and Douglas 1987) and 19 wild litters (York 1996), which compares favorably to estimates based on reproductive organs (2.7 – 3.9 (CL), 2.7 – 3.2 (BC), and 2.5 – 2.9 (PC); review in Powell 1993).

Of greater concern is the possibility that ovary, and to lesser degree uterine, analyses might consistently overestimate parturition rate, thereby also underestimating annual variability in parturition rates. Various indications of pregnancy may be detected, though not all of those females may den and produce kits in spring. For example, this might occur if BC fail to implant or fetuses are resorbed as a result of nutritional stress during the period of embryonic diapause (Arthur and Krohn 1991). Overall, CL counts have generally yielded ovulation rates for fisher of $\geq 95\%$ (Shea et al. 1985, Douglas and Strickland 1987, Paragi 1990, Crowley et al. 1990, MN

DNR unpublished data), while more 'direct' estimates of average parturition rate from radio-marked animals have been lower (46-75%; Crowley et al. 1990; Arthur and Krohn 1991; Paragi 1990; Paragi et al. 1994, York 1996, Truex et al. 1998, Higley and Mathews 2009), and are often highly variable. Conversely, in Minnesota, Kuehn (1989) did not detect changes in fisher pregnancy rate (from CL analysis) in spite of a 64% decline in a presumably important prey species (snowshoe hare; *Lepus americanus*).

For martens, several largely ovarian-based estimates of annual pregnancy rate have often been in the range of 80-90% (Archibald and Jessup 1984, Strickland and Douglas 1987, Aune and Schladweiler 1997, Flynn and Schumacher 1994, Fortin and Cantin 2004, MN DNR unpublished data). However, like for fishers, several marten studies have documented (also based largely on CL counts) lower or more variable pregnancy rates (Thompson and Colgan 1987, Aune and Schladweiler 1997, Strickland and Douglas 1987, Flynn and Schumacher 2009), perhaps a result of fluctuations in prey abundance (Hawley and Newby 1957, Weckwerth and Hawley 1962, Strickland 1981, Strickland and Douglas 1987, Thompson and Colgan 1987, Fryxell et al. 1999, Flynn and Schumacher 2009). We are aware of direct field-based estimates of parturition rate from radio-marked marten in only one state (Maine). Pooling samples across 4 years, the proportion of lactating adult females was 75, 81, and 92% for their 3 different study areas (Phillips 1994, Payer 1999), similar to many of the CL-based pregnancy studies.

Understanding reproductive ecology of these species also necessitates gathering information on natal and maternal den structures and selection of den sites. Natal dens are the structures where kits are born, whereas maternal dens are sites used subsequently by the female with her dependent young. Although data is absent for Minnesota, nearly all reported fisher natal dens have been in cavities of large-diameter trees or snags (Leonard 1986, Paragi et al. 1996, Powell et al. 1997, Truex et al. 1998). In northern studies, the majority of fisher natal dens have been in large diameter aspens (*Populus* spp), and females may use up to 3 or more different maternal dens (Powell et al. 2003, Higley and Mathews 2009). Marten natal and maternal dens are also frequently in tree cavities (Gilbert et al. 1997), but may occur in more varied features (e.g., under-ground burrows, exposed root masses of trees, rock piles, large downed logs; Ruggiero et al. 1998).

Though not further discussed here, the literature is also voluminous with documentation of the importance of tree cavities, large downed logs, and other forest 'structure' for fisher and marten resting sites (see Powell et al. 2003 for a review). Initial results from this study (Joyce 2013) appear consistent with other published findings on the importance of forest structure for marten den and rest sites. Given the continuing pressure to maximize fiber production from forests (i.e., short forest rotation, biomass harvesting, etc), the forest structural attributes critical to fishers and martens could become limiting in the future, if not already. Hence, acquiring Minnesota-specific information is critical to better inform forest management activities.

OBJECTIVES

As part of a larger project on *Martes* ecology (Erb et al. 2009), we began efforts to better describe the reproductive ecology of fisher and marten in Minnesota. Specific objectives are to:

- Document denning chronology;
- Determine structures used for natal and maternal dens;
- Quantify vegetative characteristics in the area surrounding natal and maternal dens;
- Develop a resource selection model specific to the denning season;
- Derive field-based estimates of pregnancy rate and litter size;
- Evaluate kit survival; and
- Assess the potential influence of age, diet, prey fluctuations, forest attributes, 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. We defer a more complete evaluation of results until additional data are collected or additional analysis is completed. Herein we present basic information on field methods and only report preliminary findings related to denning chronology, dens structures, and pregnancy rates and litter sizes. For initial analysis related to den and rest site selection for martens, we refer the reader to Joyce (2013).

STUDY AREA

Marten research is focused on 1 study area located in northeastern Minnesota (Figure 1; Area 1), though an occasional marten has been captured and radio-collared in Area 2 (Figure 1). Area 1 (~700 km²) 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 has taken place in 3 areas (Figure 1; Areas 1, 2, and 3). The work in Area 3 is a collaborative effort between Camp Ripley Military Reservation, Central Lakes Community College, and the Minnesota Department of Natural Resources. While we do include animals captured in that area in our basic summaries, we do not discuss other aspects of that project in this report. Area 2 (1075 km²), our primary fisher study area, is ~ 67% public ownership, including portions of the Chippewa National Forest and state and county lands. Extremely few martens occupy Area 2.

METHODS

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

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

During processing, we placed animals on heating pads connected to a power inverter and 12 volt battery. Portable shelters and propane heaters were also used to keep animals warm during processing. We monitored respiration, pulse, and rectal temperature during anesthesia. We weighed and sexed animals and typically removed a first pre-molar for aging. Morphological measurements taken included body length, tail length, hind foot length, and chest, neck, and head circumference. We removed guard hair samples for possible genotyping, and for evaluating the use of stable isotope analysis for deciphering food habits (Ben-David et al. 1997). To assist with determining which female fishers would likely produce kits, blood samples were drawn when possible to measure serum progesterone levels (Frost et al. 1997). Antibiotics were administered subcutaneously to all animals prior to release as a precaution against infection (Kreeger et al. 2002) from minor wounds that may have occurred while in the trap, and because of certain invasive procedures utilized during handling (ear-tagging, removal of tooth).

During the pilot year, we deployed several radio-collar designs on fishers, including an Advanced Telemetry Systems (ATS; Isanti, MN) M1585 zip-tie collar (~43 g), an ATS M1930 collar (~38 g), and a Lotek Wireless Inc. (Newmarket, ON, CA) SMRC-3 collar (~ 61 g; deployed on adult males only). Since the pilot year, we have primarily deployed ATS M1940 (~43 g) or Sirtrack (Havelock North, New Zealand) TVC-162 collars (~45 g) on fishers. The majority of martens have been fitted with Holohil Systems Ltd. (Carp, ON, CA) MI-2 collars (~31 g). We retrofitted each collar with a temperature data logger (I-button model DS1922L; Maxim Integrated, San Jose, CA) 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 ground-tracked collared females to locate possible den structures. When a suspected den structure was located, we then deployed remotely-activated cameras (Reconyx PC-85, RC-55, or HC600, Reconyx, Inc, Holmen, WI) to monitor female activity. We considered a female to have given birth only if kits were confirmed via sound or video/camera, or if other reliable evidence (e.g., obvious lactation, placental scars, or kit bite marks on collar) was obtained when an animal was subsequently handled as a mortality or recapture. Litter size was ascertained via visual confirmation in most cases, though we also utilized placental scar counts on any females that died during summer or fall, and for which other methods failed to produce a count. To confirm or count kits at dens located in tree cavities, we used an MVC2120-WP color video camera (Micro Video Products, Bobcaygeon, Ontario), attached to a telescoping pole if necessary, and connected to a laptop computer. Dens were only examined when the radio-marked female was not present. If video inspection equipment did not work at a particular den structure, we deployed remote cameras in an effort to obtain pictures of kits when they emerged or were moved by the female (Jones et al. 1997).

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

We are also collecting more detailed information on vegetative characteristics of the site surrounding each den structure, with a goal of developing a biologically meaningful den site selection model using methods and metrics that should be available from existing and periodically collected forest sampling data (e.g., see Zielinski et al. 2006). Following the United States Forest Service's Forest Inventory and Analysis (FIA) protocol, we 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 is 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 are collecting quantitative data on 1) mean DBH and basal area of live trees, overall and by species; 2) % overhead (angular) canopy; 3) sapling density; 4) understory cover density; 5) density and volume of snags and stumps; 6) volume of coarse woody debris; 7) distance to improved road; and 8) distance to water. Canopy structure will also be categorized based on number and distribution of canopy layers. Lower-resolution (1 ppm) LIDAR data will also be analyzed in all study areas, along with higher-resolution (8 ppm) data for a portion of the marten study area.

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 on the north-facing side of a tree in each of 6 cover types. Each sensor records temperature every 30 minutes from December 1 – June 1. At approximately 10-day intervals from 1 December – 1 April, we also recorded snow depth and 2 measures of snow compaction at 3 locations along transects situated in each of 6 cover types. 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-loading measures (females) reported by Krohn et al. (2004). Depth of snow compaction was recorded by dropping each load tool from 1 in. above snow level and measuring compaction depth.

Prey sampling transects have also been established in both study areas. Prey sampling is being conducted primarily to document between-area differences in prey abundance, annual within-area fluctuations in prey, and ultimately to assess whether fisher or marten habitat use, diet, survival, or reproductive success is correlated with prey dynamics. Prey-sampling transects ($n \approx 125$ in each study area) consist of 10 sampling locations (2 parallel lines of 5 stations) spaced 20m apart, with transects distributed in 6 cover types throughout each study area. Transects are generally oriented perpendicular to roads or trails, with the first plot 30m off the trail. In spring, we count snowshoe hare (*Lepus americanus*) pellets in a 1-m^2 plot at each sampling station (McCann et al. 2008). During fall, small mammal snap-trapping occurs for 2 consecutive days at the same sampling stations, similar to protocols used on an existing small mammal survey in Minnesota (Aarhus-Ward 2009). During both spring (hare pellet sampling) and fall (small mammal trapping), we also count the number of red squirrels (*Tamiasciurus hudsonicus*) observed or heard along each transect. Rather than using 10-min point counts (e.g., Mattson and Reinhart 1996, Bayne and Hobson 2000) with our small mammal/hare pellet stations as the sampling points, we record the number of unique squirrels detected per transect (summarized per unit time) while checking pellet plots and small mammal traps. Information on white-tailed deer and ruffed grouse (*Bonasa umbellus*) populations may be available from existing surveys or population models.

RESULTS AND DISCUSSION

Including the pilot year of the study, a total of 232 martens (111F, 121M) and 101 fishers (58F, 43M) have been radio-collared. Because tooth aging has not yet been completed for all animals, some of which may be only 1 year of age (i.e., not capable of producing kits), we present results only for animals known to be ≥ 2 years of age during spring den visits, or those of unknown age but for which we have confirmed parturition at the time of this writing (i.e., until age is known, we do not include animals that we have confirmed to be nulliparous). As of this writing, spring 2014 reproductive status assessment is largely complete for radio-collared fishers, but still ongoing for a few martens.

Treating females that were alive during multiple parturition periods (years) as independent units, and excluding females known to be < 2 years of age, we have confirmed age and reproductive status for 40 female martens (Table 1). Pooling years, pregnancy rate has been 57% for 2-year-old martens ($n=14$), and 85% for martens 3 years or older ($n=26$) (Table 1). We have been able to confidently assess litter size for 24 marten litters, for which average litter size is 2.92 (range = 1-4); litter size averaged 2.83 for 2-year-olds ($n=6$) and 3.0 for 3+ year-old ($n=16$) martens (Table 1). Data suggests marten kits are typically born from late-April through mid-May.

A total of 45 natal or maternal dens have been confirmed for martens (Table 2). For temporal reference, 37 (82%) of the marten dens were documented to be used from mid-April through June 1st, with the low number ($n=8$) of maternal dens located after this a result of time constraints and increasing difficulty in finding dens in summer. Of the 45 dens, 64% were in tree cavities whereas 36% were in underground burrows (Table 2). Of the 29 dens in tree

cavities, 69% have been in live trees whereas 31% have been in snags. Pooling live trees and snags, most tree cavity dens used by martens have been in aspen (n=11) and cedar (n=10), with from 1 – 3 dens located each in tamarack (*Larix laricina*), red maple (*Acer rubrum*), black ash (*Fraxinus nigra*), and white pine (*Pinus strobus*) (Table 2). We have completed DBH measurements on 16 of the 29 den trees, yielding an average DBH of 18.6" (range = 13.3 – 30.0") (Table 2). Of the 16 underground dens, 8 were characterized as being in soils with an abundance of medium to large rocks, 5 were under the base of larger trees or associated with shallow roots or sphagnum 'soils' adjacent to the base of the tree, while 3 were under 'tip-ups' or stumps (Table 2). Most female martens appear to move their kits from their natal den to 1 or more maternal dens in the first 8 weeks following birth.

Similar to martens, we treat female fishers that were alive during multiple parturition periods (years) as independent units. Excluding individuals known to be 1 year of age during the parturition period, we have confirmed both age and reproductive status for 51 female fishers (Table 1). Pooling years, pregnancy rate for female fishers has been 57% for 2-year-olds (n=21), and 93% for fishers 3 years or older (n=30) (Table 1). We have been able to confidently assess litter size for 37 fisher litters from which average litter size is 2.5 (range = 1–4); litter size averaged 2.25 for 2-year olds (n=12) and 2.7 for 3+ year-old (n=23) fishers (Table 1). Based on data collected to date, it appears fisher kits are typically born in early to mid-March, or ~1.5 months earlier than marten kits.

A total of 56 fisher natal or maternal dens have been located to date (Table 3). For temporal reference, 52 (93%) of the fisher dens confirmed were documented to be used in March and April, with the low number (n=4) of maternal dens located after June 1 resulting from time constraints and increasing difficulty in finding dens in summer. Of the 56 dens confirmed, all but 2 were in elevated tree cavities; the remaining 2 maternal dens were in large hollow logs either on or suspended above the ground (Table 3). Of the dens in tree cavities, 74% have been in live trees whereas 26% have been in snags. Pooling live trees and snags, most tree cavity dens used by fishers have been in aspen (n=37) and oak (n=8), with from 1 – 3 dens located each in sugar maple (*Acer saccharum*), red maple, white cedar, and white pine (Table 3). We have completed DBH measurements on 53 of the 55 fisher den trees, yielding an average DBH of 20.5" (range = 13.9 – 29.1") (Table 3). Similar to martens, most female fishers appear to move their kits from their natal den to 1 or more maternal dens in the first 8 weeks following birth.

ACKNOWLEDGEMENTS

We thank volunteer Richard Nelles for his dedicated assistance with fisher trapping and den monitoring efforts. We also acknowledge Michael Joyce, Carolin Humpal, and Ron Moen for ongoing assistance with various aspects of the project. Special thanks to pilots Al Buchert, Tom Pfingsten, John Heineman, Jason Jensen, Don Murray, Tom Buker, and Bob Geving for aerial telemetry efforts during this project. This project was funded in part by the Wildlife Restoration Program (Pittman-Robertson).

LITERATURE CITED

- Aarhus-Ward, A. J. 2009. 2007-2008 Small mammal survey report. 1854 Treaty Authority Technical Report 09-05.
- Archibald, W. R., and R. H. Jessup. 1984. Population dynamics of pine marten in the Yukon Territory. Pages 81-97 in R. Olson, R. Hastings, F. Geddes, eds. Northern ecology and resource management. University of Alberta Press, Edmonton, Canada.
- Arthur, S. M., and W. B. Krohn. 1991. Activity patterns, movements, and reproductive ecology of fishers in southcentral Maine. *Journal of Mammalogy* 72: 379-385.

- Aune, K. E., and P. Schladweiler. 1997. Age, sex structure, and fecundity of the American marten in Montana. Pages 61-77 in G. Proulx, H. N. Bryant, P. M. Woodard, eds. *Martes: Taxonomy, ecology, techniques, and management*. Provincial Museum of Alberta, Edmonton, Canada.
- Balsler, D. S., and W. H. Longley. 1966. Increase of the fisher in Minnesota. *Journal of Mammalogy* 47: 547-550.
- Bayne, E., and K. Hobson. 2000. Relative use of continuous and fragmented boreal forest by red squirrels (*Tamiasciurus hudsonicus*). *Canadian Journal of Zoology* 78: 359-365.
- Ben-David, M., R. W. Flynn, and D. M. Schell. 1997. Annual and seasonal changes in diet of martens: evidence from stable isotope analysis. *Oecologia* 111:280-291.
- Cherepak, R.B., and M. L. Connor. 1992. Constantly pregnant - well almost. Reproductive hormone levels of the fisher (*Martes pennanti*), a delayed implanter. *Norwegian Journal of Agricultural Sciences (Supplement)* 9:150-154.
- Coulter, M. W. 1966. Ecology and management of fishers in Maine. Ph.D. Dissertation, Syracuse University, Syracuse, NY.
- Crowley, S. K., W. B. Krohn, and T. F. Paragi. 1990. A comparison of fisher reproductive estimates. *Transactions of the Northeast Section of the Wildlife Society* 47: 36-42.
- Douglas, C. W., and M. A. Strickland. 1987. Fisher. Pages 511-530 in M. Novak, J. A. Baker, M. E. Obbard, and B. Malloch, eds. *Wild furbearer management and conservation in North America*. Ontario Ministry of Natural Resources, Toronto.
- Erb, J., B. Sampson, and P. Coy. 2009. Fisher and marten demography and habitat use in Minnesota. *Summaries of Wildlife Research Findings, 2008*. Minnesota Department of Natural Resources, St. Paul.
- Flynn, R. W., and T. V. Schumacher. 1997. Ecology of martens in southeast Alaska. Alaska Department of Fish and Game Federal Aid Progress Report.
- Flynn, R. W., and T. V. Schumacher. 2009. Temporal changes in population dynamics of American marten. *Journal of Wildlife Management* 73:1269-1281.
- Fortin, C., and M. Cantin. 2004. Harvest status, reproduction, and mortality in a population of American martens in Quebec, Canada. Pages 221-234 in D. J. Harrison, A. K. Fuller, and G. Proulx, eds. *Martens and fishers in human-altered environments: An international perspective*. Springer Science, New York.
- Frost, H. C., W. B. Krohn, and C. R. Wallace. 1997. Age-specific reproductive characteristics in fishers. *Journal of Mammalogy* 78: 598-612.
- Frost, H. C., E. C. York, W. B. Krohn, K. D. Elowe, T. A. Decker, S. M. Powell, and T. K. Fuller. 1999. An evaluation of parturition indices in fishers. *Wildlife Society Bulletin* 27: 221-230.
- Fryxell, J. M., J. B. Falls, E. A. Falls, R. J. Brooks, L. Dix, and M. J. Strickland. 1999. Density dependence, prey dependence, and population dynamics of martens in Ontario. *Ecology* 80:1311-1321.
- Gilbert, F. F. 1987. Methods for assessing reproductive characteristics of furbearers. Pages 180-189 in M. Novak, J. A. Baker, M. E. Obbard, and B. Malloch, eds. *Wild furbearer management and conservation in North America*. Ontario Ministry of Natural Resources, Toronto.
- Gilbert, J. H., J. L. Wright, D. J. Lauten, and J. R. Probst. 1997. Den and rest-site characteristics of American marten and fisher in northern Wisconsin. Pages 135-145 in G. Proulx, H. N. Bryant, P. M. Woodard, eds. *Martes: Taxonomy, ecology, techniques, and management*. Provincial Museum of Alberta, Edmonton, Canada.
- Hawley, V. D., and F. E. Newby. 1957. Marten home ranges and population fluctuations. *Journal of Mammalogy* 38: 174-184.
- Higley, J. M., and S. Matthews. 2009. Fisher habitat use and population monitoring on the Hoopa Valley Reservation, California. Final Report USFWS TWG U-12-NA-1. Hoopa Valley Tribe. Hoopa, CA.

- Johnson, W. N., T. F. Paragi, and D. D. Katnik. 1995. The relationship of wildland fire to lynx and marten populations and habitat in interior Alaska. Final Report. U.S. Fish and Wildlife Service, Galena, Alaska.
- Jones, L. L. C., M. G. Raphael, J. T. Forbes, and L. A. Clark. 1997. Using remotely activated cameras to monitor maternal dens of martens. Pages 329-349 in G. Proulx, H. N. Bryant, P. M. Woodard, eds. *Martes: Taxonomy, ecology, techniques, and management*. Provincial Museum of Alberta, Edmonton, Canada.
- Joyce, M. J. 2013. Space use and multi-scale habitat selection of American marten (*Martes americana*) in northeastern Minnesota. Master's Thesis, University of Minnesota-Duluth.
- Kreeger, T. J., J. M. Arnemo, and J. P. Raath. 2002. Handbook of wildlife chemical immobilization: International edition. Wildlife Pharmaceuticals Inc., Fort Collins, Colorado.
- Krohn, W. B., C. Hoving, D. Harrison, D. Phillips, and H. Frost. 2004. *Martes* foot-loading and snowfall patterns in eastern North America: implications to broad-scale distributions and interactions of mesocarnivores. Pages 115-131 in D. J. Harrison, A. K. Fuller, and G. Proulx, eds. *Martens and fishers in human-altered environments: An international perspective*. Springer Science, New York.
- Kuehn, D. W. 1989. Winter foods of fishers during a snowshoe hare decline. *Journal of Wildlife Management* 53: 688-692.
- Leonard, R. D. 1986. Aspects of reproduction of the fisher in Manitoba. *Canadian Field-Naturalist* 100: 32-43.
- Mattson, D. J., and D. P. Reinhart. 1996. Indicators of red squirrel (*Tamiasciurus hudsonicus*) abundance in the whitebark pine zone. *Great Basin Naturalist* 56: 272-275.
- McCann, N. P., R. A. Moen, and G. J. Niemi. 2008. Using pellet counts to estimate snowshoe hare numbers in Minnesota. *Journal of Wildlife Management* 72:955-958.
- Mead, R. A. 1994. Reproduction in the *Martes*. Pages 404-422 in S. W. Buskirk, A. S. Harestad, M. G. Raphael, and R. A. Powell, eds. *Martens, sables, and fishers: Biology and conservation*. Cornell University Press, Ithaca, N.Y.
- Paragi, T. F. 1990. Reproductive biology of female fisher in southcentral Maine. M. S. Thesis, University of Maine, Orono.
- Paragi, T. E., W. B. Krohn, and S. M. Arthur. 1994. Using estimates of fisher recruitment and survival to evaluate population trend. *Northeast Wildlife* 51: 1-11.
- Paragi, T. E., S. M. Arthur, and W. B. Krohn. 1996. Importance of tree cavities as natal dens for fishers. *Northeastern Journal of Applied Forestry* 13: 79-83.
- Payer, D. C. 1999. Influences of timber harvesting and trapping on habitat selection and demographic characteristics of American marten. Ph. D. Dissertation, University of Maine, Orono.
- Payne, N. F. 1982. Assessing productivity of furbearers. Pages 39-50 in G. C. Sanderson, ed. *Midwest furbearer management*. Proceedings of the 43rd Midwest Fish and Wildlife Conference, Wichita, KS.
- Phillips, D. M. 1994. Social and spatial characteristics, and dispersal of marten in a forest preserve and industrial forest. M. S. Thesis, University of Maine, Orono.
- Powell, R. A. 1993. *The fisher: Life history, ecology, and behavior*. University of Minnesota Press, Minneapolis.
- Powell, R. A., S. W. Buskirk, and W. J. Zielinski. 2003. Fisher and marten. Pages 635-649 in G. A. Feldhamer, B. C. Thompson, and J. A. Chapman, eds. *Wild mammals of North America: Biology, management, and conservation*, The Johns Hopkins University Press, Baltimore, MD.
- Powell, S. M., E. C. York, J. J. Scanlon, and T. K. Fuller. 1997. Fisher maternal den sites in central New England. Pages 265-278 in G. Proulx, H. N. Bryant, P. M. Woodard, eds. *Martes: Taxonomy, ecology, techniques, and management*. Provincial Museum of Alberta, Edmonton, Canada.

- Ruggiero, L. F., D. E. Pearson, and S. E. Henry. 1998. Characteristics of American marten den sites in Wyoming. *Journal of Wildlife Management* 62: 663-673.
- Shea, M. E., N. L. Rollins, R. T. Bowyer, and A. G. Clark. 1985. Corpora lutea number as related to fisher age and distribution in Maine. *Journal of Wildlife Management* 49: 37-40.
- Strickland, M. A. 1981. Fisher and marten study 1979-80 and 1980-81. Ontario Ministry of Natural Resources Algonquin Region Progress Report 7.
- Strickland, M. A., and C. W. Douglas. 1987. Marten. Pages 531-546 *in* M. Novak, J. A. Baker, M. E. Obbard, and B. Malloch, eds. *Wild furbearer management and conservation in North America*. Ontario Ministry of Natural Resources, Toronto.
- Strickland, M. A., C. W. Douglas, M. Novak, and N. P. Hunziger. 1982. Pages 599-612 *in* G. A. Feldhamer and J. A. Chapman, eds. *Wild mammals of North America: Biology, management, and economics*. The Johns Hopkins University Press, Baltimore, MD.
- Swanson, G., T. Surber, and T. S. Roberts. 1945. The mammals of Minnesota. Minnesota Department of Conservation Technical Bulletin No. 2.
- Thompson, I. D., and P. W. Colgan. 1987. Numerical responses of martens to a food shortage in northcentral Ontario. *Journal of Wildlife Management* 51: 824-835.
- Truex, R. L., W. J. Zielinski, R. T. Gollightly, R. H. Barrett, and S. M. Wisely. 1998. A meta-analysis of regional variation in fisher morphology, demography, and habitat ecology in California (draft report). California Department of Fish and game, Wildlife Management Division, Nongame Bird and Mammal Section.
- Weckwerth, R. P., and V. D. Hawley. 1962. Marten food habits and population fluctuations in Montana. *Journal of Wildlife Management* 26: 55-74.
- York, E. C. 1996. Fisher population dynamics in north-central Massachusetts. M. S. thesis, University of Massachusetts, Amherst.
- Zielinski, W. J., R. L. Truex, J. R. Dunk, and T. Gaman. 2006a. Using forest inventory data to assess fisher resting habitat suitability in California. *Ecological Applications* 16: 1010-1025.

Table 1. Parturition rate and litter size for radio-collared¹ female fishers and martens in Minnesota from 2008-2014.

Species*Age	Parturition Rate		Litter Size		
	# females	% with litters	# litters	Average	Range
Martens					
2-year-olds	14	57	6	2.83	1 - 4
3+ year-olds	26	85	16	3.0	1 - 4
All	42 ²	74	24 ³	2.92	1 - 4
Fishers					
2-year-olds	21	57	12	2.25	1 - 4
3+ year-olds	30	93	23	2.7	1 - 4
All	51	78	37 ³	2.5	1 - 4

¹ Excludes unknown-aged nulliparous females and all 1-year-olds. Multiple years for same female treated as independent.

² Includes 2 females with age ≥ 2 , but otherwise unknown age

³ Includes known litters from unknown-aged females

Table 2. Natal and maternal den structures (n=45) used by radio-collared female martens in Minnesota from 2008 - 2014.

Den Structure	# dens	% of total	Average DBH (in.)	DBH Range (in.)
Above-Ground, All Tree Cavities	29	64.4	18.6	13.3 – 30.0
Cavity, live tree	20	42.2	19.6	15.9 – 30.0
Cavity, snag	9	20.0	15.3	13.3 – 20.4
All Aspen cavities	11	24.4	18.0	13.3 – 23.8
All Cedar cavities	10	22.2	19.2	16.1 – 21.5
All Tamarack cavities	3	6.7	16.8	Only 1 measured to date
All Red Maple cavities	2	4.4	16.3	15.9 – 16.7
All Black Ash cavities	2	4.4	16.1	14.3 – 17.8
All White Pine cavities	1	2.2	30.0	
Below-Ground Dens	16	35.6		
Burrow, under base of tree	5	11.1	14.5	9.0 – 18.6
Burrow, rocky soils/outcrop	8	17.8		
Burrow, under stump/tip-up	3	6.7		

Table 3. Natal and maternal den structures (n=56) used by radio-collared female fishers in Minnesota from 2008 - 2014.

Den Structure	# dens	% of total	Average DBH (in.)	DBH Range (in.)
Above-Ground, Tree Cavities	54	96.4	20.5	13.6 – 29.1
Cavity, live tree	40	71.4	20.9	13.9 – 29.1
Cavity, snag	14	25.0	19.6	13.6 – 26.1
Aspen cavities	37	66.1	20.4	15.1 – 29.1
Oak cavities	8	14.3	20.5	15.1 – 28.0
White pine cavities	3	5.3	23.7	21.9 – 25.6
Sugar Maple cavities	2	3.6	20.6	19.1 – 22.1
Red Maple cavities	2	3.6	20.8	18.0 – 23.6
Cedar cavities	2	3.6	17.1	13.9 – 20.3
Hollow Log	2	3.6	16.9	15.4 – 18.3

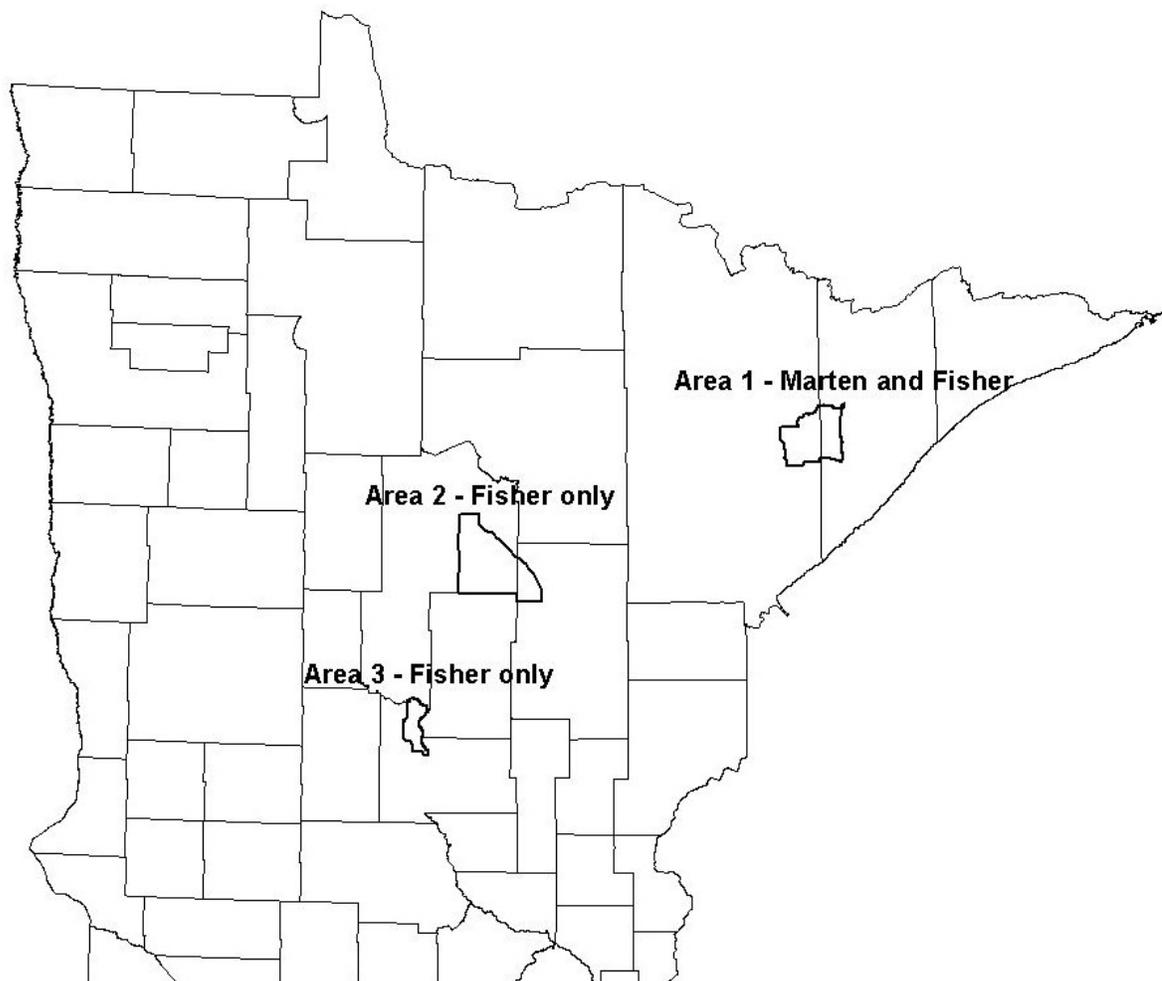


Figure 1. Fisher and American marten study areas in Minnesota, 2008-2014.



SURVIVAL AND CAUSES OF MORTALITY FOR FISHERS AND MARTENS 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 survival of radio-collared fishers (*Martes pennanti*) and martens (*Martes americana*) during winter 2007-08. Including the pilot year of the study, a total of 232 martens (111F, 121M) and 101 fishers (58F, 43M) have been radio-collared. An additional 6 animals (3 martens, 3 fishers) were ear-tagged only. Of the 232 martens radio-collared, 33 are still actively monitored (15F, 18M), radio-contact has been lost on 61, 7 whose collars are inaccessible have either slipped their collars or died, and 131 deaths have been confirmed (of which 12 were censored due to death within 2 weeks of capture). Of the 119 non-censored marten deaths (53F, 66M), most have been from legal fur trapping (n=47; 34M, 13F) and predation (n=55; 33F, 22M). Approximately 91% of the marten predation deaths have been attributed to mammalian carnivores and 9% to raptors. Although natural mortality of martens has occurred in most seasons, it is highest in spring and lowest in fall. No significant sex bias has been observed in overall mortality - female martens make up 46% of our sample and 44% of the known deaths. However, marten harvest mortality has been male-biased (70% male) while natural mortality has been female-biased (60% female).

Of the 101 fishers radio-collared, 10 are still being monitored (6F, 4M), radio contact was lost on 33, 7 whose collars are inaccessible have either slipped their collars or died, and 51 deaths (32F, 19M) have occurred. Of the 51 fisher deaths, most have been from predation (n=25; 20F, 5M) and fur trapping (8 in-season, 8 accidentally out-of-season). Two fishers have been car-killed and 5 others died from unknown but apparent natural mortality. Similar to martens, natural mortality for fishers is highest in spring and lowest in fall. Of 20 female fishers predated, 19 were killed by other mammalian carnivores. Conversely, 4 of the 5 male fisher predation mortalities were attributed to raptors (all bald eagles). Of particular note, 18 of the 20 female fishers killed by predators were adults, and 17 of them were killed while they still had dependent young in natal dens, indirectly resulting in the death of all their offspring. The deaths of these 18 nursing females and their litters represent ~33% of the reproductive 'opportunities' for adult female fishers monitored during the kit-rearing season since the study began. We hypothesize that the observed timing and magnitude of female mortality is a result of increased activity and increased vulnerability during the kit-rearing season. Because the magnitude of the mortality would not likely have been sustainable for an extended period, we suggest that survival patterns have probably changed in the last 10 years. We continue to explore several hypotheses, but suspect a partial explanation to be that cumulative changes in the environment may have had both direct (e.g., reduction in denning habitat quality) and indirect (e.g., weather and habitat more favorable to competing bobcats) effects on survival of female fishers in the core of Minnesota's fisher range.

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, Balsler 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 now 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). Over the past 10 years, fisher abundance and harvest have been increasing along the southern and western edge of the 'forest zone' where forest historically transitioned to savanna and prairie and is now characterized by linear forest corridors (streams/rivers) or smaller forest patches interspersed with agriculture. Conversely, fisher abundance appears to have declined significantly over the same period in the core forested areas of north-central and northeast Minnesota. Peak statewide harvest levels have been near 4,000 and 3,500 for marten and fisher, respectively. However, due to apparent multi-year population declines for both species, harvest seasons from 2007 to the present have become progressively more conservative, with the 2013 harvest season lasting only 6 days with a combined fisher/marten limit of 2 per trapper.

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 focusing management activities on issues of concern.

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

Natural mortality, particularly via predation, appears more common with martens. Marten survival data is available from Wisconsin (McCann et al. 2010), Maine (Hodgman et al. 1994, 1997), Ontario (Thompson 1994), Oregon (Bull and Heater 2001), British Columbia (Poole et al. 2004), Alaska (Flynn and Schumacher 1995, 2009), Quebec (Potvin and Breton

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

OBJECTIVES

As part of a larger project on *Martes* ecology in Minnesota (Erb et al. 2009), we began monitoring survival and causes of mortality for fishers and martens. 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 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). Our primary objectives are to:

- Determine causes of mortality;
- Estimate cause- and sex-specific mortality rates;
- Document seasonal patterns of mortality; and
- Examine potential effects of winter weather, prey fluctuations, competitor density, activity patterns, and habitat on survival probability.

Herein we present basic descriptive information regarding number of captures and number and causes of deaths. We defer a more comprehensive statistical analysis until a later time.

STUDY AREA

Marten research is focused on 1 study area located in northeastern Minnesota (Figure 1; Area 1), though an occasional marten has been captured and radio-collared in Area 2 (Figure 1). Area 1 (~700 km²) 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 has taken place in 3 areas (Figure 1; Areas 1, 2, and 3). The work in Area 3 is a collaborative effort between Camp Ripley Military Reservation, Central Lakes Community College, and the Minnesota Department of Natural Resources. While we do include animals captured in that area in our basic summaries, we do not discuss other aspects of that project in this report. Area 2 (1075 km²), our primary fisher study area, is ~ 67% public ownership, including portions of the Chippewa National Forest and state and county lands. Extremely few martens occupy Area 2.

METHODS

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

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

During processing, we placed animals on heating pads connected to a power inverter and 12 volt battery. Portable shelters and propane heaters were also used to keep animals warm during processing. We monitored respiration, pulse, and rectal temperature during anesthesia. We weighed and sexed animals and typically removed a first pre-molar for aging. Morphological measurements taken included body length, tail length, hind foot length, and chest, neck, and head circumference. We removed guard hair samples for possible genotyping, and for evaluating the use of stable isotope analysis for deciphering food habits (Ben-David et al. 1997). To assist with determining which female fishers would likely produce kits, blood samples were drawn when possible to measure serum progesterone levels (Frost et al. 1997). Antibiotics were administered subcutaneously to all animals prior to release as a precaution against infection (Kreeger et al. 2002) from minor wounds that may have occurred while in the trap, and because of certain invasive procedures utilized during handling (ear-tagging, removal of tooth).

During the pilot year, we deployed several radio-collar designs on fishers, including an Advanced Telemetry Systems (ATS; Isanti, MN) M1585 zip-tie collar (~43 g), an ATS M1930 collar (~38 g), and a Lotek Wireless Inc. (Newmarket, ON, CA) SMRC-3 collar (~ 61 g; deployed on adult males only). Since the pilot year, we have primarily deployed ATS M1940 (~43 g) or Sirtrack (Havelock North, New Zealand) TVC-162 collars (~45 g) on fishers. The majority of martens have been fitted with Holohil Systems Ltd. (Carp, ON, CA) MI-2 collars (~31 g). We retrofitted each collar with a temperature data logger (I-button model DS1922L; Maxim Integrated, San Jose, CA) 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.

All radio-locations, except for some taken during the den-monitoring period, were 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 232 martens (111F, 121M) and 101 fishers (58F, 43M) have been radio-collared. An additional 6 animals (3 martens, 3 fishers) were ear-tagged only. Tooth aging has not yet been completed for all animals; however we note that because capture operations took place during winter, all animals were a minimum of ~ 8 months of age at initial capture. We have yet to derive formal estimates of survival rate. Instead, we provide a simple overview of the fate of collared animals in this summary.

Excluding 12 martens that were censored due to death within 14 days post-release, 33 (15%) of the 220 collared martens are actively being monitored, radio contact has now been lost on 53 (24%), 8 (4%) slipped their collars, the status of 7 (3%) is uncertain due to unrecoverable collars, and 119 (54%) have died (Table 1). Of the 119 non-censored deaths, most have been from legal fur trapping ($n=47$) and predation ($n=55$) (Table 2). Nine 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 6 from unknown but assumed natural causes (Table 2).

Of the 53 non-censored marten deaths that we believe could be attributed to either avian or mammalian predation, evidence suggests 48 (91%) were killed by mammalian predators and 5 (9%) by raptors. Although predation deaths have occurred in nearly all months, predation and overall natural mortality is highest in the spring and lowest in the fall (Figure 2). Forensic (DNA) analysis of samples collected from predated marten (mammalian predation only) is still incomplete. To date, field evidence and DNA analysis suggests bobcats (*Lynx rufus*) as a common mammalian predator, with red fox (*Vulpes vulpes*) and fisher also confirmed in several cases.

Excluding martens censored within 14 days of capture, our sample of radio-collared marten has been comprised of 46% females. In comparison, female martens have accounted for 44% of the total marten deaths, 30% of the total deaths due to harvest, and 60% of the predation deaths. Although there is no apparent sex-bias to overall mortality, marten harvest has been notably male-biased while natural mortality has been female-biased.

Of the 101 fishers radio-collared, 10 (10%) are still being monitored, radio contact has been lost with 18 (18%), 15 (15%) collars have come unattached, the fate of 7 (7%) (unrecovered collars) is uncertain, and 51 (50%) have died (Table 1). General cause of death (human versus natural) could be assigned to 48 of the 51 fisher deaths. Of these, 18 (37.5%) were attributable to humans (8 trapped during legal season, 8 accidental trapping, and 2 car-killed) whereas 30 (62.5%) were attributable to natural causes (25 predated, 5 unknown natural cause) (Table 2). The seasonal pattern of natural mortality for fishers has been similar to that of martens, being greatest in spring and lowest in fall (Figure 3).

Of the 25 predated fishers, 20 were females (Table 2). Of the 20 females predated, only 1 was attributed to an avian predator (Great-horned owl (*Bubo virginianus*) suspected). Conversely, 4 of the 5 male fisher predation deaths were attributed to bald eagles (*Haliaeetus leucocephalus*), though we can't rule out scavenging in 1 case where only the radio-collar was retrieved directly underneath an active eagle nest. The fifth male fisher predated appears to have been killed by a wolf. We are awaiting forensic DNA analysis on many fishers killed by mammalian predators. However, similar to martens, field evidence and forensic DNA analysis suggests bobcats as the most common predator, with canids (wolf or coyote) suspected in 2 fisher deaths.

Of particular note, 18 of the 20 female fishers killed by other predators were adults (≥ 2 years old), and 17 of those 18 were killed while they still had dependent kits in natal or maternal dens, indirectly resulting in the death of all their dependent offspring. The deaths of these kit-rearing females represent a complete loss of ~33% of the fisher litters monitored during spring and early summer since the study began.

We continue to suspect that 2 broad factors may explain the high mortality of kit-rearing female fishers during late-winter and spring: increased activity, and increased vulnerability (independent of activity level). Given the potential for negative energy balance during parts of winter, compounded in early spring by the added energy demands of gestation and lactation, female fishers may need to increase activity in spring to meet energy demands. Combined with the need to locate suitable (and multiple) natal or maternal dens, this activity, much of which may be in localized areas near den trees and hence more predictable/detectable to other carnivores, may increase predation risk. Preliminary data from temperature data loggers attached to radio-collars suggests that fishers do spend increasing amounts of time (compared to winter) outside of den and rest sites during late-winter and spring. Secondly, independent of their activity level, fishers may be more vulnerable in spring because concealment cover is diminished (i.e., before 'green-up'), other predators may also increase activity in spring, and potential prey for carnivores is at the low point in the annual cycle. Collectively, this may yield a period of high energetic need that overlaps a high risk competitive environment for female fishers. Progressing into summer and fall, concealment cover is maximal, prey abundance (for all carnivores) is maximal, and energetic demands (and hence movements) of female fishers decrease as kits are weaned.

Regardless of the explanation, it seems unlikely that the level of predation we have observed on nursing female fishers during the study would be sustainable for long periods, which may partially explain the decline in fisher abundance in core areas over the previous decade. However, many of the correlates to the timing of predation mortality that we have mentioned are not new challenges for adult female fisher, and since 1977 the core fisher population appears to have only been in decline over the last ~ 9 years suggesting that other more recent changes may be altering dynamics. Likely explanations for the observed and presumably new mortality pattern for female fishers continue to be assessed, including potential declines in fisher habitat quality in core fisher range and changes in habitat and weather that may have contributed to an increase in competing bobcats. Overharvest of fisher, particularly in the 4 years preceding the start of this study, may also have contributed to the decline in fisher abundance, but would not explain the high natural mortality of female fishers, and to some extent female martens, that we have observed during the study.

ACKNOWLEDGEMENTS

We thank volunteer Richard Nelles for his dedicated assistance with trapping and other project activities. We also acknowledge Carolin Humpal, Michael Joyce, and Ron Moen for their ongoing assistance with numerous aspects of the project. Special thanks to pilots Al Buchert, Tom Pfingsten, John Heineman, Jason Jensen, Don Murray, Tom Buker, and Bob Geving for aerial telemetry efforts during this project. We also thank Greta Wengert for assistance with forensic analyses. This project was funded in part by the Wildlife Restoration Program (Pittman-Robertson).

LITERATURE CITED

- Balser, D. S., and W. H. Longley. 1966. Increase of the fisher in Minnesota. *Journal of Mammalogy* 47: 547-550.
- Bull, E. L., and T. W. Heater. 2000. Resting and denning sites of American martens in northeastern Oregon. *Northwest Science* 74: 179-185.
- Erb, J., B. Sampson, and P. Coy. 2009. Fisher and marten demography and habitat use in Minnesota. *Summaries of Wildlife Research Findings, 2008*. Minnesota Department of Natural Resources, St. Paul.

- Flynn, R. W., and T. V. Schumacher. 1997. Ecology of martens in southeast Alaska. Alaska Department of Fish and Game Federal Aid Progress Report.
- Flynn, R. W., and T. V. Schumacher. 2009. Temporal changes in population dynamics of American marten. *Journal of Wildlife Management* 73:1269-1281.
- Fredrickson, R. J. 1990. The effects of disease, prey fluctuation, and clear-cutting on American marten in Newfoundland, Canada. M.S. thesis, Utah State University, Logan, UT.
- Frost, H. C., W. B. Krohn, and C. R. Wallace. 1997. Age-specific reproductive characteristics of fisher. *Journal of Mammalogy* 78:598-612.
- Hodgman, T. P., D. J. Harrison, D. D. Katnik, and K. D. Elowe. 1994. Survival in an intensively trapped marten population in Maine. *Journal of Wildlife Management* 58: 593-600.
- Hodgman, T. P., D. J. Harrison, D. M. Phillips, and K. D. Elowe. 1997. Survival of American marten in an untrapped forest preserve in Maine. Pages 86-99 in G. Proulx, H. N. Bryant, P. M. Woodard, eds. *Martes: Taxonomy, ecology, techniques, and management*. Provincial Museum of Alberta, Edmonton, Canada.
- Kelly, G. M. 1977. Fisher (*Martes pennanti*) biology in the White Mountain national Forest and adjacent areas. Ph.D. Dissertation, University of Massachusetts, Amherst.
- Koen, E. L., J. Bowman, and C. S. Findlay. 2007. Fisher survival in eastern Ontario. *Journal of Wildlife Management* 71:1214-1219.
- Krohn, W. B., S. M. Arthur, and T. F. Paragi. 1994. Mortality and vulnerability of a heavily trapped fisher population. Pages 137-146 in S. W. Buskirk, A. S. Harestad, M. G. Raphael, and R. A. Powell, eds. *Martens, sables, and fishers: Biology and conservation*. Cornell University Press, Ithaca, N.Y.
- McCann, N. P., P. A. Zollner, and J. H. Gilbert. 2010. Survival of adult martens in northern Wisconsin. *Journal of Wildlife Management* 74:1502-1507.
- Mech, L. D., and L. L. Rogers. 1977. Status, distribution, and movements of martens in northeastern Minnesota. USDA Forest Service Research Paper NC-143. North Central Forest Experiment Station, St. Paul, Minnesota.
- Paragi, T. E., W. B. Krohn, and S. M. Arthur. 1994. Using estimates of fisher recruitment and survival to evaluate population trend. *Northeast Wildlife* 51: 1-11.
- Poole, K. G., A. D. Porter, A. de Vries, C. Maundrell, S. D. Grindal, and C. C. St. Clair. 2004. Suitability of a young deciduous-dominated forest for American marten and the effects of forest removal. *Canadian Journal of Zoology* 82: 423-435.
- Potvin, F., and L. Breton. 1997. Short-term effects of clearcutting on martens and their prey in the boreal forest of western Quebec. Pages 452-474 in G. Proulx, H. N. Bryant, P. M. Woodard, eds. *Martes: Taxonomy, ecology, techniques, and management*. Provincial Museum of Alberta, Edmonton, Canada.
- Strickland, M. A., C. W. Douglas, M. Novak, and N. P. Hunziger. 1982. Pages 599-612 in G. A. Feldhamer and J. A. Chapman, eds. *Wild mammals of North America: Biology, management, and economics*. The Johns Hopkins University Press, Baltimore, MD.
- Strickland, M. A., and C. W. Douglas. 1987. Marten. Pages 531-546 in M. Novak, J. A. Baker, M. E. Obbard, and B. Malloch, eds. *Wild furbearer management and conservation in North America*. Ontario Ministry of Natural Resources, Toronto.
- Strickland, M. A., and C. W. Douglas. 1981. The status of the fisher in North America and its' management in southern Ontario. Pages 1443-1458 in J. A. Chapman and D. Pursley, eds. *Proceedings of the worldwide furbearer conference*, Frostburg, MD.
- Swanson, G., T. Surber, and T. S. Roberts. 1945. The mammals of Minnesota. Minnesota Department of Conservation Technical Bulletin No. 2.
- Thompson, I. D. 1994. Marten populations in uncut and logged boreal forests in Ontario. *Journal of Wildlife Management* 58: 272-280.
- York, E. C. 1996. Fisher population dynamics in north-central Massachusetts. M. S. thesis, University of Massachusetts, Amherst.

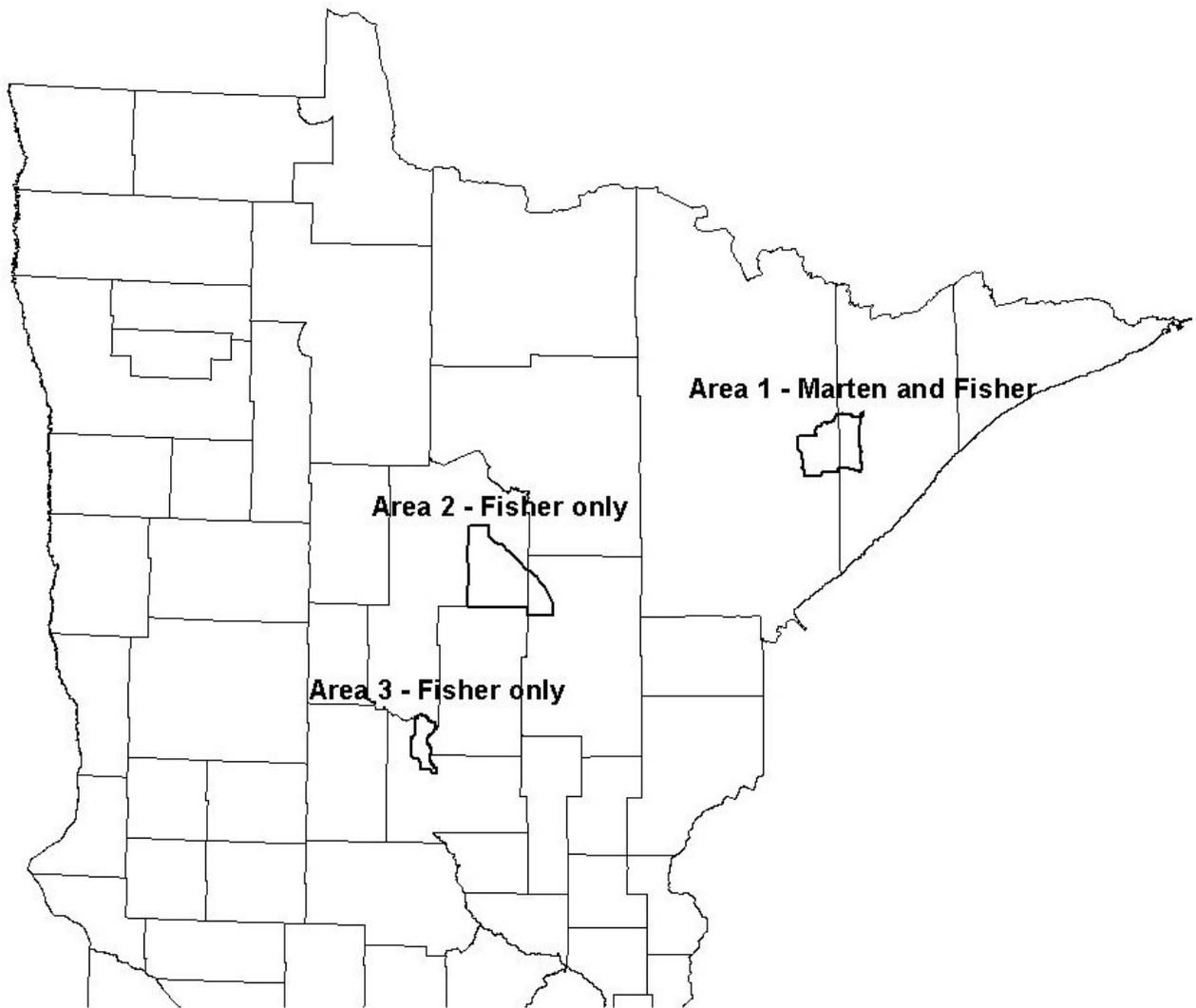


Figure 1. Fisher and marten study areas in Minnesota 2008-2014.

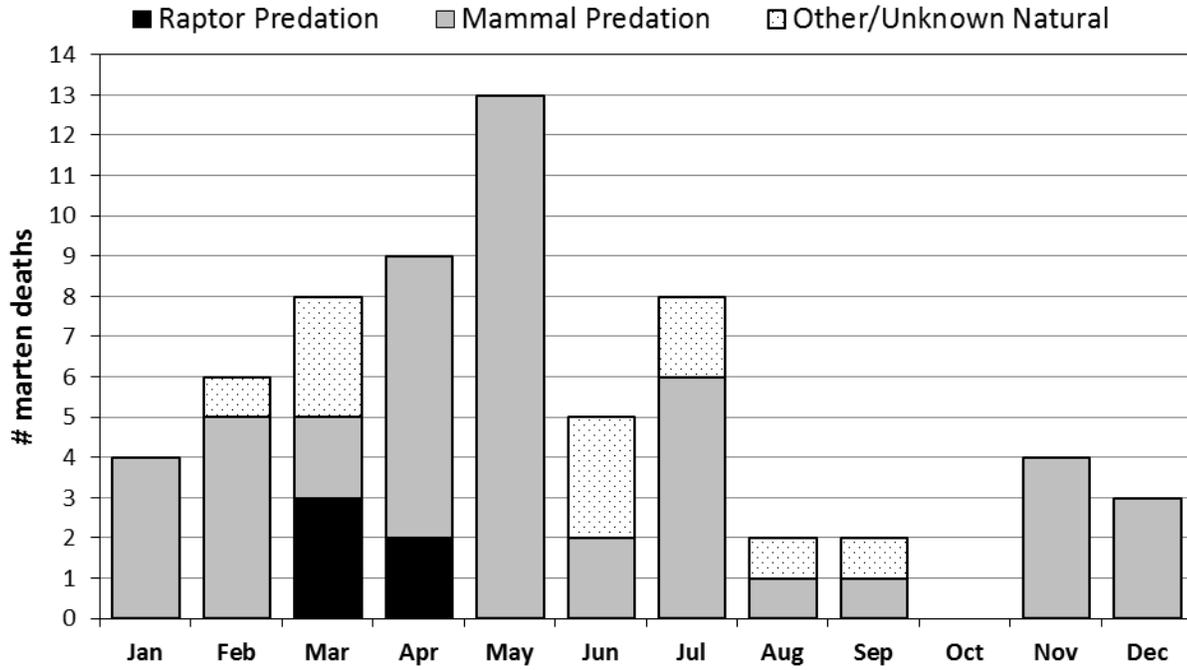


Figure 1. Seasonal timing of natural mortality for martens in northeast Minnesota, 2007 - 2014.

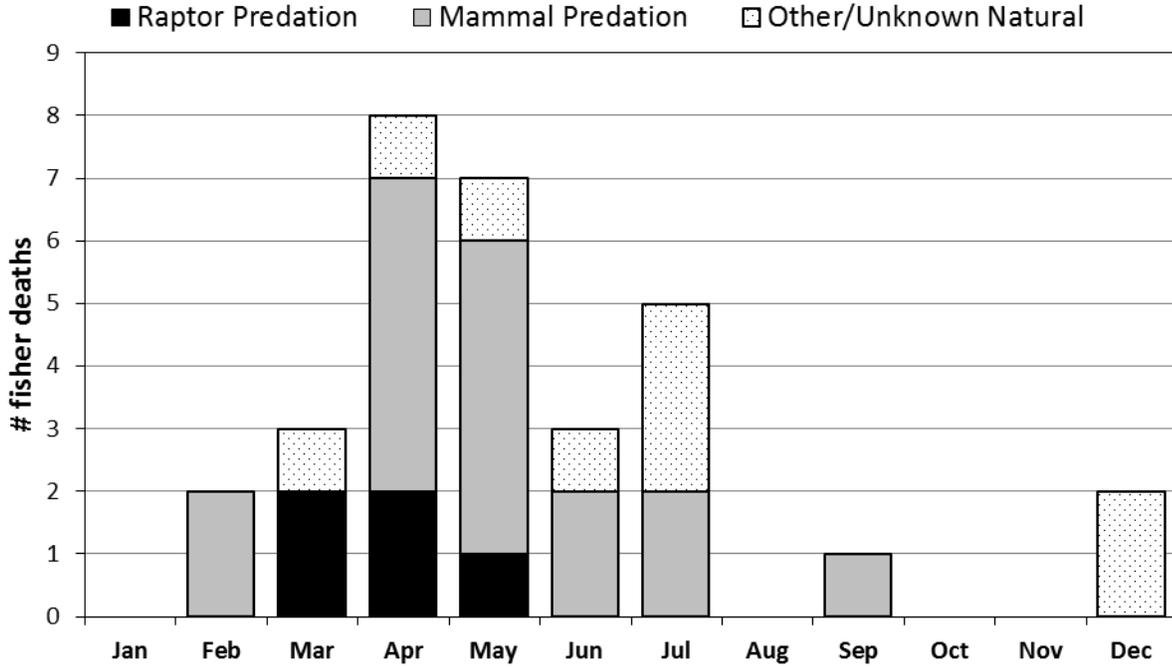


Figure 2. Seasonal timing of natural mortality for fishers in northeast Minnesota, 2007 - 2014.

Table 1. Number and status of fishers and martens radio-collared¹ in Minnesota from 2007 - 2014.

Sex*Species	# Active	# Deaths	# Missing	# Slipped Collars	Unknown ²	Total
Male Martens	18	66	23	4	1	112
Female Martens	15	53	30	4	6	108
Male Fishers	4	19	10	6	4	43
Female Fishers	6	32	8	9	3	58

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

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

Table 2. Cause of death for fishers and martens radio-collared¹ in Minnesota from 2007 - 2014.

Sex*Species	Predation	Natural Accident	Disease/ Illness	Unknown Natural	Car-Killed	Trapped In Season	Trapped Out of Season	Collar Complication	Unknown ²	Total
Male Martens	22	2	1	4	0	34	2	1	0	66
Fem. Martens	33	0	0	2	0	13	2	2	1	53
Male Fishers	5	0	0	3	1	4	5	0	1	19
Fem. Fishers	20	0	0	2	1	4	3	0	2	32

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

² Unknown represents animals where evidence was insufficient to assign to natural versus human-related cause.



IDENTIFYING BARRIERS TO MOVEMENT AND THE EFFECTIVENESS OF CORRIDORS FOR CONNECTING CORE AREAS: LANDSCAPE GENETICS OF PRAIRIE GROUSE IN FRAGMENTED LANDSCAPES

Charlotte Roy, Eric Nelson¹, and Andrew Gregory²

SUMMARY OF FINDINGS

Cooperators and staff began collecting feathers at sharp-tailed grouse (*Tympanuchus phasianellus*) and greater prairie-chicken (*Tympanuchus cupido*) leks during the spring of 2014. Samples will be sent for genetic analysis at the close of the season. Feather collection will continue in 2015.

INTRODUCTION

The grassland habitats that prairie grouse require have become increasingly fragmented as a result of competing pressures on the land (Berg 1997). Core habitat areas are isolated from each other by unsuitable areas that may prevent successful movement and the colonization of newly created habitat. The Minnesota Prairie Conservation Plan recognizes the importance of providing dispersal corridors to connect isolated core areas and identifies the greater prairie-chicken as an indicator species for upland prairie and grassland habitat (Minnesota Prairie Plan Working Group 2011). Similarly, sharp-tailed grouse must be able to move among isolated grassland, brushland, savanna, and peatland habitat patches (Berg 1997), through areas that may pose difficulty for successful movement. If the resistances of various landscapes to movement are understood, then more effective corridors can be identified, and management efforts can be prioritized using this information (Epps et al. 2007, Braunisch et al. 2010, Spear et al. 2010).

Landscape genetics is an emerging field that provides methods to examine connectivity on the landscape by combining a GIS with information about genetic variation in a population (Braunisch et al. 2010, Lowe and Allendorf 2010, Sork and Waits 2010, Haig et al. 2011). This tool can be used to examine effective dispersal (gene flow) on the landscape, without having to rely on telemetry techniques, which can be expensive and may require large numbers of marked animals if successful dispersal events are infrequent (Coulon et al. 2004, Spear et al. 2010). Landscape genetic methods have been used in recent years to identify barriers to dispersal, including human development, non-habitat land cover types, and distance in species like capercaillie (*Tetrao urogallus*, Braunisch et al. 2010), northern bobwhite (*Colinus virginianus*, Berkman et al. 2013a,b), and prairie-chickens (Gregory 2011). Thus, landscape genetics can be used to examine the movements of birds in a spatially explicit manner.

OBJECTIVES

- 1- To identify barriers to movement for sharp-tailed grouse and greater prairie-chickens in Minnesota (e.g., distance, urban development, treed areas) as measured by genetic connectivity
- 2- To identify landscape features and types that enable movements of prairie grouse among areas of suitable habitat in Minnesota (e.g., agriculture) as measured by genetic connectivity

¹ Wildlife Damage Program Coordinator, Minnesota Department of Natural Resources

² Assistant Professor of Spatial Ecology, Bowling Green State University, Bowling Green, OH

- 3- To improve corridor planning and provide guidance to keep connected populations connected

METHODS

Wildlife managers and cooperators surveyed prairie-chickens and sharp-tailed grouse at leks throughout Minnesota in the spring of 2014. Feathers were lost at leks during male contests, copulations, and as a result of other activities. Following completion of 1 survey at each lek (Figs.1 and 2), managers and cooperators collected feathers from dispersed locations on the lek. This maximized the probability of sampling different individuals. Each cluster of feathers, or single feather if necessary, was placed in an envelope and labeled with the lek location, date, and a unique number for each individual (e.g., UTM coordinates, date, species). Information from each envelope was recorded in a database, and the samples will be shipped to a commercial lab (Wildlife Genetics International, British Columbia) for extraction of genetic material when all samples have been received for the season.

If necessary, feather samples can be supplemented with hunter-harvested birds. Harvested samples can be restricted to adult birds which will be identified based on plumage characteristics (Bihrlé 1993), so that analyses are not confounded by the unknown dispersal status of juveniles. The sex of birds can be determined molecularly (Fridolfsson and Ellegren 1999), if not possible from available plumage (Bihrlé 1993).

At the commercial genetic lab, genetic material will be extracted and amplified at numerous microsatellite loci. Microsatellites are highly variable, neutral (non-coding) genetic loci. Recent studies of prairie-chickens and sharp-tailed grouse have identified microsatellite loci that are polymorphic in these species (and populations, see citations in Gregory 2011 and Malone 2012). This genetic information can be linked to spatial information to examine the connectivity of the landscape. Areas that share greater connectivity will be similar genetically, whereas areas with restricted connectivity will be more dissimilar genetically. Analytical methods will be revisited for the most recent advances prior to initiating data analysis.

DISCUSSION

Results of this study will provide information about landscape features that isolate habitat fragments and those that serve to promote connectivity. Following the methods of Gregory (2011) or Barton et al. (2010) we can also use the described landscape genetic analyses to understand the relative influence of different landscape elements to promote or inhibit dispersal. This information will be useful to target management efforts in ways that can more effectively accomplish the goal of connecting core areas, enhancing local habitat conditions, and providing new habitat sufficiently close to existing leks to promote colonization.

LITERATURE CITED

- Barton, H. D., A. J. Gregory, R. Davis, C. A. Hanlon, and S. M. Wisely. 2010. Contrasting landscape epidemiology of two sympatric rabies virus strains. *Molecular Ecology* 19:2725-2738.
- Berg, W. E. 1997. The sharp-tailed grouse in Minnesota. Minnesota Wildlife Report 10, Department of Natural Resources, St. Paul, Minnesota.
- Berkman, L. K., C. K. Nielsen, C. L. Roy, and E. J. Heist. 2013a. Population genetic structure among bobwhite in an agriculturally modified landscape. *Journal of Wildlife Management* 77:1472-1481.
- Berkman, L. K., C. K. Nielsen, C. L. Roy, and E. J. Heist. 2013b. Resistance is futile: Effects of landscape genetic features on gene flow of the northern bobwhite. *Conservation Genetics* 14:323-332.
- Bihrlé, C. 1993. Upland game identification- A basic guide for aging and sexing the bird in your hand. *North Dakota Outdoors* 56(3):9-23. Jamestown, ND: Northern Prairie Wildlife Research Center Online. <http://www.mpwrc.usgs.gov/resource/tools/upland/index.htm> Accessed 25 September 2013.
- Braunisch, V., G. Segelbacher, and A. H. Hirzel. 2010. Modeling functional landscape connectivity from genetic population structure: a new spatially explicit approach. *Molecular Ecology* 19:3664-3678.
- Coulon, A., J. F. Cosson, J. M. Angibault, B. Cargnelutti, M. Galan, N. Morellet, E. Petit, S. Aulagnier, and A. J. M. Hewison. 2004. Landscape connectivity influences gene flow in a roe deer population inhabiting a fragmented landscape: an individual-based approach. *Molecular Ecology* 13:2841-2850.
- Epps, C.W., J. D. Wehausen, V. C. Bleich, S. G. Torres, and J. S. Brashares. 2007. Optimizing dispersal and corridor models using landscape genetics. *Journal of Applied Ecology* 44:714-724.
- Fridolfsson, A-K, and H. Ellegren. 1999. A simple and universal method for molecular sexing of non-ratite birds. *Journal of Avian Biology* 30:116-121.
- Gregory, A. 2011. Landscape genetics and behavioral ecology of greater prairie-chickens (*Tympanuchus cupido*). Dissertation. Kansas State University, Division of Biology. Manhattan, Kansas.
- Haig, S. M., W. M. Bronaugh, R. S. Crowhurst, J. D'Elia, C. A. Eagles-Smith, C. W. Epps, B. Knaus, M. P. Miller, M. L. Moses, S. Oyler-McCance, W. D. Robinson, and B. Sidlauskas. 2011. Genetic applications in avian conservation. *The Auk* 128:205-229.
- Lowe, W. H., and F.W. Allendorf. 2010. What can genetics tell us about population connectivity. *Molecular Ecology* 19:3038-3051.
- Malone, K. 2012. Assessing the need for and impact of translocation as a means of genetic rescue in a declining population of sharp-tailed grouse (*Tympanuchus phasianellus*). M.S. Thesis, Central Michigan University, Mount Pleasant, MI.
- Minnesota Prairie Plan Working Group. 2011. Minnesota Prairie Conservation Plan. Minnesota Prairie Plan Working Group, Minneapolis, MN. 55p.
- Sork, V. L. and L. Waits. 2010. Contributions of landscape genetics- approaches, insights, and future potential. *Molecular Ecology* 19:3489-3495.
- Spear, S. F., N. Balkenhol, M. Fortin, B. H. McRae, and K. Scribner. 2010. Use of resistance surfaces for landscape genetic studies: considerations for parameterization and analysis. *Molecular Ecology* 19:3576-3591.

Figure 1. Prairie-chicken leks ($n = 188$) surveyed in northwestern Minnesota during 2013. County boundaries and Minnesota GAP cover classes are provided for context. These leks will be the focus of feather collection efforts in 2014 and 2015.

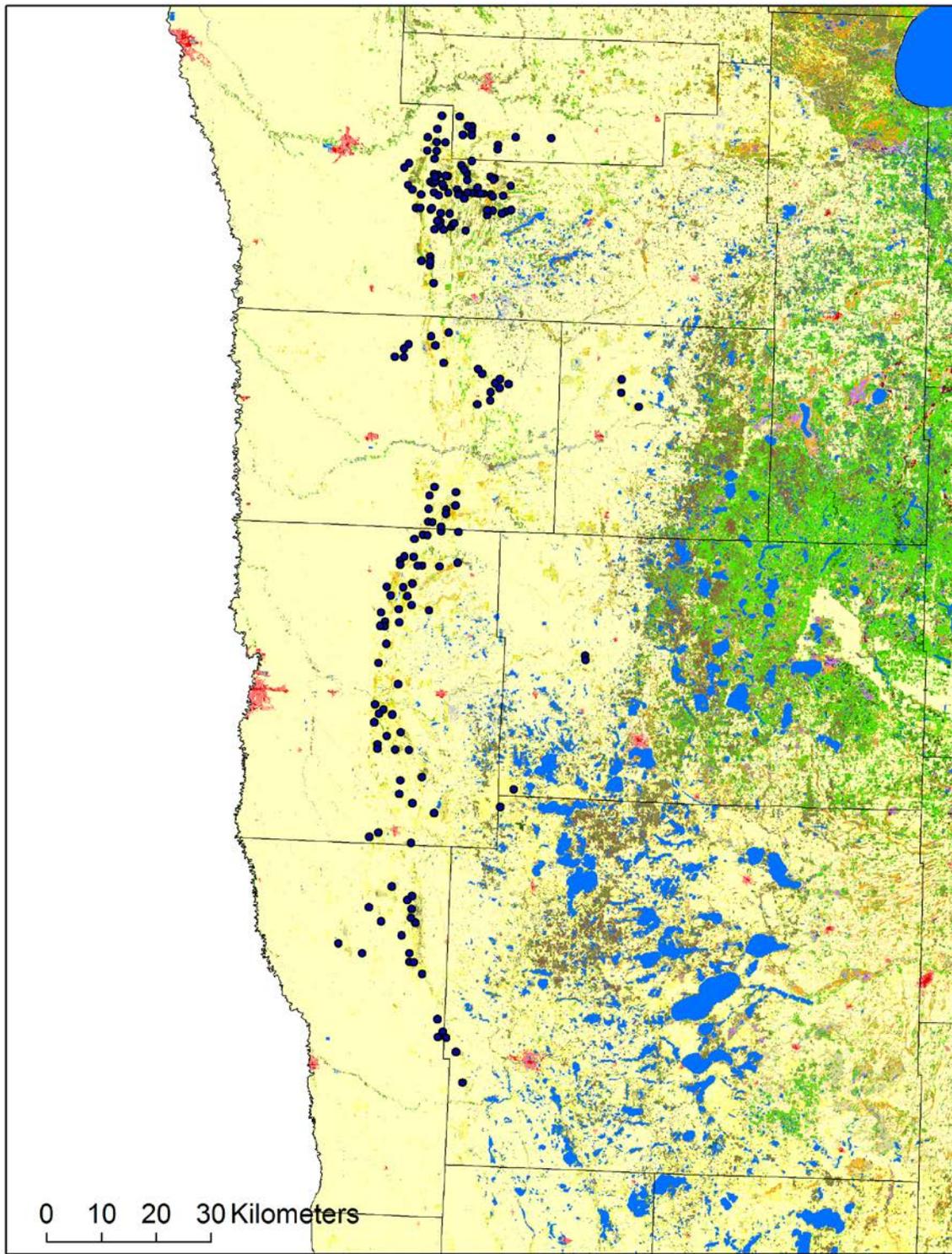
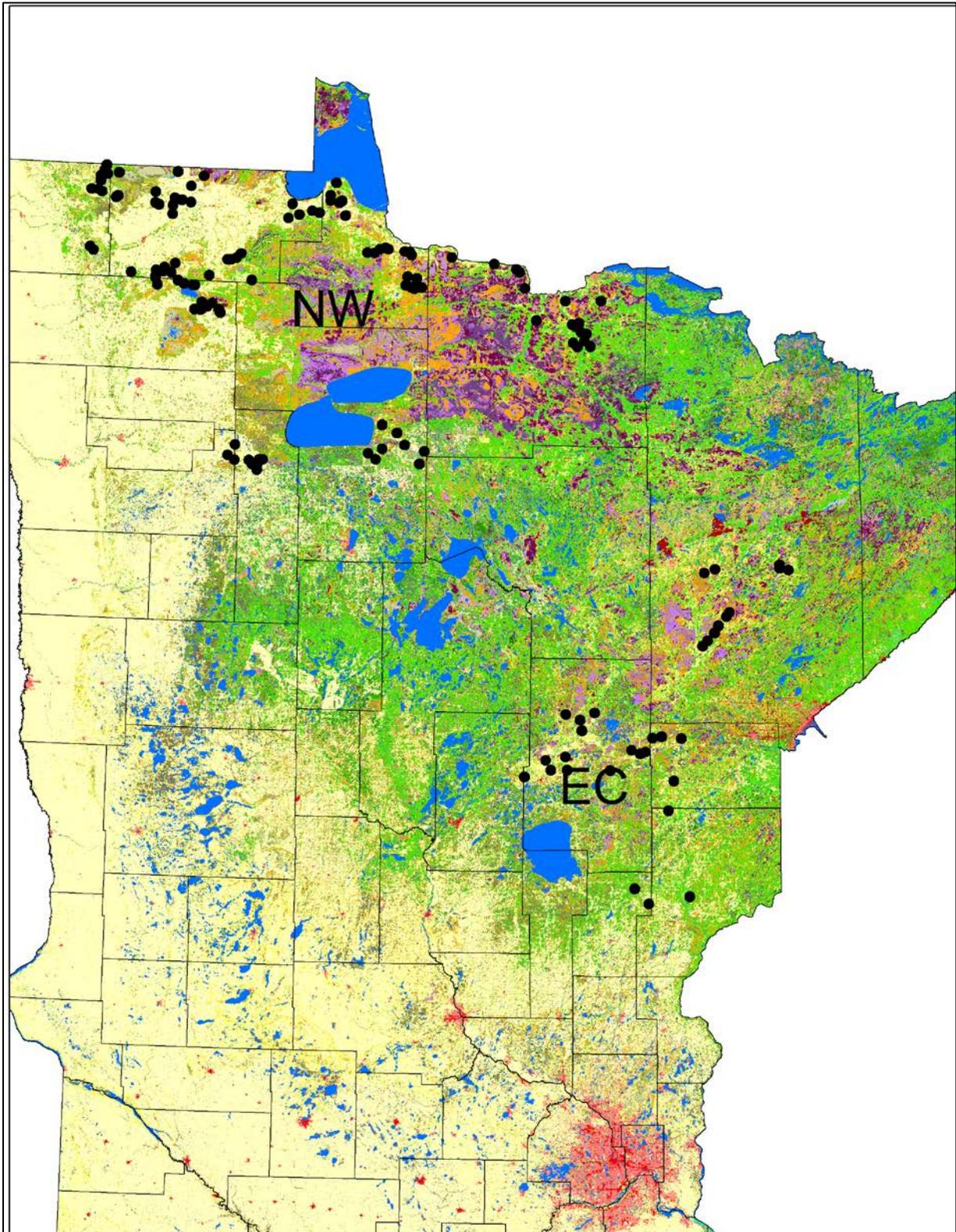


Figure 2. Sharp-tailed grouse leks ($n = 139$) surveyed in northwestern (NW) and east-central (EC) Minnesota during 2013. County boundaries and Minnesota GAP cover classes are provided for context. These leks will be the focus of feather collection efforts in 2014 and 2015.





DEVELOPING SURVEY AND STUDY METHODOLOGY FOR SPRUCE GROUSE: A PILOT STUDY

Charlotte Roy, Mike Larson, and John Giudice

SUMMARY OF FINDINGS

Data collection began in April 2014 for a 2-year pilot study. Results will be forthcoming in future reports.

INTRODUCTION

The spruce grouse, *Falci pennis canadensis*, is a boreal forest inhabitant existing at the southern edge of its range in the Lake States of northern Minnesota, Wisconsin, and Michigan. Spruce grouse occupy forest stands with a variety of coniferous tree species including spruce, fir, pine, hemlock, and tamarack, and habitat selection varies with the forest community in various portions of spruce grouse range (Boag and Schroeder 1992). In the Lake States, spruce grouse prefer areas of black spruce (*Picea mariana*) with some jack-pine (*Pinus banksiana*) interspersed (Robinson 1969, Pietz and Tester 1982, Anich et al 2013), but have also been reported using balsam fir (*Abies balsamea*), tamarack (*Larix laricina*), eastern hemlock (*Tsuga canadensis*) and northern white cedar (*Thuja occidentalis*). In nearby Wisconsin, closed canopy coniferous uplands near lowland conifer swamps were important, especially mature black spruce-tamarack swamps and jack pine 16-29 years old (Anich et al. 2013). Similarly in Minnesota, but with a more modest sample, spruce grouse used lowland conifers with black spruce and mixed bogs during the breeding season, and used jack pine during winter (Pietz and Tester 1982). Boag and Schroeder (1992) indicated that early successional stands were preferred but Anich et al. (2013) found mature forest to be important.

Spruce-fir forests are predicted to be vulnerable to climate change at the southern edge of their range (Scheller and Mladenoff 2005, Prasad et al. 2007, Iverson et al. 2008). Thus, spruce grouse are expected to experience a range contraction due to climate change-induced habitat loss, and will likely have a more limited distribution in Minnesota in the future. A northward shift in spruce grouse distribution is already underway in the state, as they once occurred in Nemadji State Forest (Pine and Carlton Counties), Itasca State Park (Clearwater County), and Hubbard County (W. Berg, pers. comm., Janssen 1987).

Minnesota is unique among the Lake States in having a sizeable spruce grouse population that still permits spruce grouse hunting. The spruce grouse is considered a Species of Special Concern in Michigan (Michigan DNR 2005) and was listed as threatened in Wisconsin in 1997 (Wisconsin DNR 2004). Yet, the only data we collect for spruce grouse is part of the Small Game Harvest Mail Survey (Dexter 2013), which provides an estimate of harvest reported by hunters between 9,000 and 27,000 birds per year over the last 10 years (Dexter 2013). However, spruce grouse harvest is more reflective of ruffed grouse hunter numbers than spruce grouse numbers, and thus these data cannot be used as a population index (Gregg et al. 2004). Thus, we have sparse data on spruce grouse distribution, and no data on population size or trends, despite a responsibility to manage spruce grouse during a period of expected habitat loss.

Minnesota currently has long term survey efforts for all grouse species except spruce grouse, including annual lek surveys of prairie grouse and annual roadside drumming counts of ruffed grouse (*Bonasa umbellus*). Other states like New York (Fritz 1979) and Wisconsin (Worland et al. 2009) have executed surveys of spruce grouse. However, these surveys were conducted during a few years, were labor intensive, and were not designed to be long term monitoring projects. Any long term, large scale monitoring effort of spruce grouse in Minnesota

would need to be easy to execute, repeatable, and representative of spruce grouse populations. Logistical constraints of doing large scale surveys often constrain survey design. In this case, spruce grouse occupy habitats that are very difficult to access away from roads. A roadside survey would possess the logistical ease desirable for a statewide effort, but several potential biases would need to be understood and addressed.

Surveys for spruce grouse are usually conducted using playback of female cantus calls (Fritz 1979, Boag and McKinnon 1982, Whitcomb et al. 1996, Lycke et al. 2011, among others). Schroeder and Boag (1989) compared counts obtained with playback of female calls to counts from a wing-clapping index and concluded that 'the female call may be a useful compromise between the accuracy of a complete count and the time efficiency of the wing-clapping index' and that both were useful for estimating the size of male spruce grouse (Franklin's subspecies) populations in Alberta. Because spruce grouse of the subspecies in Minnesota do not execute the wing-clap display, we will use playbacks of cantus calls to survey spruce grouse.

OBJECTIVES

1. Assess the feasibility of a roadside survey for spruce grouse to determine distribution and population trends of spruce grouse
2. Estimate capture success and identify constraints to radio-tracking for future study

STUDY AREA

The pilot survey will be conducted on Red Lake Wildlife Management Area (WMA) and Beltrami Island State Forest. This study area is on the southwestern edge of the assumed current range of spruce grouse, where changes would be expected to occur before changes more centrally within their Minnesota range. The long term goal is to expand the survey throughout spruce grouse range in northern Minnesota, if the survey is deemed feasible, and for it to be repeated at a regular interval (to be determined).

METHODS

We developed a pilot spruce grouse survey using playback of female cantus calls (Fritz 1979, Boag and McKinnon 1982, Schroeder and Boag 1989, Whitcomb et al. 1996, Lycke et al. 2011). In spring 2014, we attempted to survey as many survey routes/points as possible to inform point survey duration (1-15 min), the duration of time it takes to complete a route, and the duration of responsiveness of spruce grouse to cantus calls (both time of day and duration of season). The duration of responsiveness to playback varies among years depending on spring phenology (i.e., shorter period of responsiveness in early springs, Anich, pers. comm.), which is consistent with findings in our study area by Larson (unpublished data) in 2011 and 2012. The duration of responsiveness may also vary throughout the day, although Lycke et al. (2011) determined that birds responded to cantus calls between sunrise and noon.

Survey routes (10 points per route) were placed along low maintenance road segments that bisected or were adjacent to forest stand types known to be used by spruce grouse in the Lake States in the spring (i.e., mature black spruce and tamarack swamps, and young jack pine; Robinson 1969, Pietz and Tester 1982, Anich et al 2013; Fig. 1). Points were spaced ≥ 300 m apart along routes to ensure independence among points, based on estimates that playback can be heard 100-150 m from the speaker (Schroeder and Boag 1989, Lycke et al. 2011, Anich, unpubl. data). Surveys were conducted during April and May beginning at sunrise. We conducted surveys when winds were < 10 mph and when precipitation was absent or light. Each point count lasted 15 min (Lycke et al. 2011, Anich et al. unpubl. data) with broadcast of the

cantus call once each minute. Intervals were subdivided into five 3-min intervals, and we recorded both new detections and redetections of each bird during each interval for time-to-detection analysis (Alldredge et al. 2007). The distance to each bird detected was recorded in <50, 50-100, 101-150, >150 m intervals. Each observer recorded date, arrival time at each stop, approximate wind speed, temperature, type of forest (jack pine, black spruce, tamarack, white cedar, red pine, other), type of detection (flutter flight, approach, etc.) and background noise (e.g., wind) on a relative scale (e.g., none, low, medium, high).

After the completion of playback surveys, we surveyed the same points with trained dogs and their handlers and attempted to locate all birds <150 m from the survey point. Other researchers have had success using dogs to locate spruce grouse (Robinson 1969, Keppie 1987, Ratti et al. 1984). We attempted to quantify the variability in skill among dogs by surveying the same points with multiple dogs. We also incorporated a grouse pellet survey along a circular transect 75 or 100 m from each point, with transect length depending on the distance that dogs roamed from their handlers, so that both surveys could be completed at the same time. We counted all grouse pellet piles within 1 m of the traversed path and calculated the length of the path traversed. We distinguished ruffed grouse pellets from spruce grouse pellets on the basis of size, shape, and color.

Data Analysis

We plan to use time-to-detection methods to estimate available birds (Alldredge et al. 2007, Johnson 2008). Time-of-detection methods treat each time interval as a separate trapping occasion in a mark-recapture framework that can be analyzed in Program MARK (White and Burnham 1999). Auditory surveys will be compared to data from dogs and pellet surveys using double-sampling techniques (Cochran 1977, Thompson 2002). We will use results of this pilot study to determine whether a statewide survey is feasible by examining the number of routes and stop locations necessary to estimate population trends and distribution statewide.

DISCUSSION

The goal is to develop a statewide survey that would provide an index of population size that could be used to estimate the trend over time and monitor changes in distribution. Current predictions of climate change suggest that the impending impact on spruce grouse will be large. Activities related to the second objective will begin the second year of the pilot study.

LITERATURE CITED

- Aldredge, M. W., K. H. Pollock, T. R. Simons, J. A. Collazo, and S. A. Shriner. 2007. Time-of-detection method for estimating abundance from point-count surveys. *Auk* 124:653-664.
- Anich, N. M., M. Worland, and K. J. Martin. 2013. Habitat use by spruce grouse in northern Wisconsin. *Wildlife Society Bulletin* DOI: 10.1002/wsb.318
- Boag, D. A., and D. T. McKinnon. 1982. Spruce grouse. Pages 61-62 in D.E. Davis, ed. *Handbook of census methods for terrestrial vertebrates*. CRC Press, Inc., Boca Raton, Fla.
- Boag, D. A., and M. A. Schroeder. 1992. Spruce grouse (*Falci pennis canadensis*) in A. Poole, ed. *The birds of North America online*. Cornell Lab of Ornithology, Ithaca, New York, USA. <http://bna.birds.cornell.edu/bna/species/005> last accessed 5 November 2013
- Cochran, W. G. 1977. *Sampling techniques*. Third edition. Wiley, New York, New York, USA.
- Dexter, M. 2013. 2012 Small game hunter mail survey. http://files.dnr.state.mn.us/publications/wildlife/2012_small_game_harvest_survey.pdf last accessed 31 October 2013.
- Fritz, R. S. 1979. Consequences of insular population structure: distribution and extinction of spruce grouse populations. *Oecologia* 42:57-65.
- Gregg, L., B. Heeringa, and D. Eklund. 2004. Conservation assessment for spruce grouse (*Falci pennis canadensis*). USDA Forest Service, Eastern Region. 33 pp.
- Iverson, L., A. Prasad, and S. Mathews. 2008. Modeling potential climate change impacts on trees of the northeastern United States. *Mitigation and Adaptation Strategies for Global Change* 13:517-540.
- Janssen, R. B. 1987. *Birds in Minnesota*. University of Minnesota Press, Minneapolis.
- Johnson, D. H. 2008. In defense of indices: the case of bird surveys. *Journal of Wildlife Management* 72:857-868.
- Keppie, D. M. 1987. Impact of demographic parameters upon a population of spruce grouse in New Brunswick. *Journal of Wildlife Management* 51:771-777.
- Keppie, D. M., and P. W. Herzog. 1978. Nest site characteristics and nest success of spruce grouse. *Journal of Wildlife Management* 42:628-632.
- Lycke, A., L. Imbeau, and P. Drapeau. 2011. Effects of commercial thinning on site occupancy and habitat use by spruce grouse in boreal Quebec. *Canadian Journal of Forestry Research* 41:501-508.
- Michigan Department of Natural Resources. 2005. Michigan's wildlife action plan SGCN status and species-specific issues. <http://www.michigandnr.com/publications/pdfs/HuntingWildlifeHabitat/WCS/SGCN/Falci pennis canadensis.pdf> Last accessed July 24 2013.
- Pietz, P. J., and J. R. Tester. 1982. Habitat selection by sympatric spruce and ruffed grouse in north central Minnesota. *Journal of Wildlife Management* 46:391-403.
- Prasad, A. M., L. R. Iverson, S. Mathews, and M. Peters. 2007-ongoing. A climate change atlas for 134 forest tree species of the eastern United States. U.S. Forest Service Northern Research Station, Delaware, Ohio. <http://www.nrs.fs.fed.us/atlas/tree/> Accessed July 24 2013.
- Ratti, J. T., D. L. Mackey, and J. R. Aldredge. 1984. Analysis of spruce grouse habitat in north-central Washington. *Journal of Wildlife Management* 48:1188-1196.
- Robinson, W. L. 1969. Habitat selection by spruce grouse in northern Michigan. *Journal of Wildlife Management* 33:113-120.
- Rosenstock, S. S., D. R. Anderson, K. M. Giesen, T. Leukering, and M. F. Carter. 2002. Landbird counting techniques: current practice and alternative. *Auk* 119:46-53.
- Scheller, R. M., and D. J. Mladenoff. 2005. A spatially interactive simulation of climate change, harvesting, wind, and tree species migration and projected changes to forest composition and biomass in northern Wisconsin, USA. *Global Change Biology* 11:307-321.

- Schroeder, M. A., and D. A. Boag. 1989. Evaluation of a density index for territorial male spruce grouse. *Journal of Wildlife Management* 53:475-478.
- Thompson, W. L. 2002. Towards reliable bird surveys: accounting for individuals present but not detected. *Auk* 119:18-25.
- Whitcomb, S. D., F. A. Servello, and A. F. O'Connell, Jr. 1996. Patch occupancy and dispersal of spruce grouse on the edge of its range in Maine. *Canadian Journal of Zoology* 74: 1951-1955.
- White, G. C., and K. P. Burnham. 1999. Program MARK: survival estimation for populations of marked animals. *Bird Study* 46(Supplement): 120-138.
- Wisconsin Department of Natural Resources. 2004. Wisconsin endangered and threatened species laws and list. <http://dnr.wi.gov/topic/EndangeredResources/documents/WIETList.pdf> Last accessed 24 July 2013.
- Worland, M., K. J. Martin, and L. Gregg. 2009. Spruce grouse distribution and habitat relationships in Wisconsin. *Passenger Pigeon* 71:5-18.
- Zwickel, F. C., and J. F. Bendell. 1967. A snare for capturing blue grouse. *Journal of Wildlife Management* 31:202-204.

Figure 1. Survey points (yellow dots) for spruce grouse located along main roads and minimum maintenance roads that bisect or are adjacent to spruce grouse habitat at Red Lake Wildlife Management Area and Beltrami Island State Forest in northwestern Minnesota. Points were spaced ≥ 300 m apart to ensure independence among survey locations.

