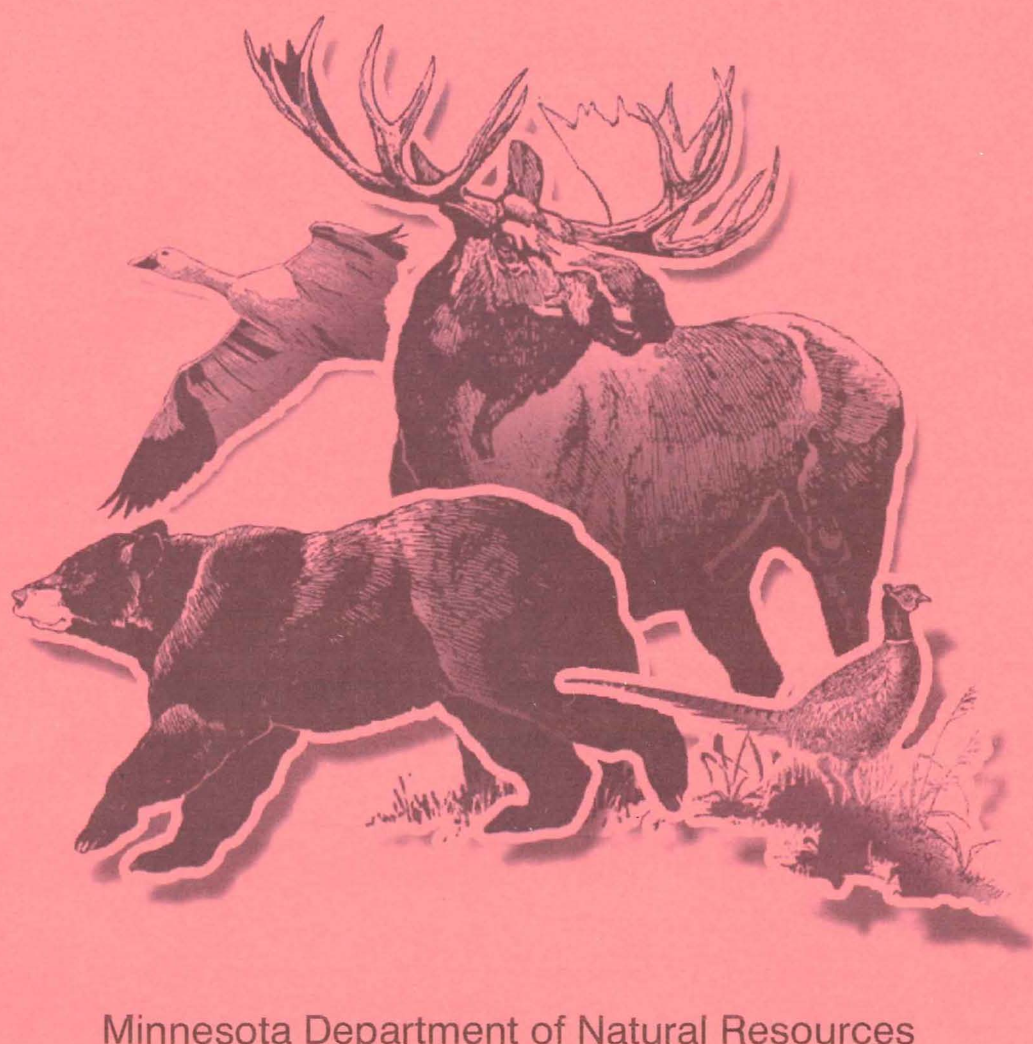
Summaries of Wildlife Research Findings

2009



Minnesota Department of Natural Resources
Division of Fish and Wildlife
Wildlife Populations and Research Unit

SUMMARIES OF WILDLIFE RESEARCH FINDINGS 2009

Edited by:
Jeffrey S. Lawrence
Richard O. Kimmel
Mark S. Lenarz

Minnesota DEPARTMENT OF NATURAL RESOURCES

Minnesota Department of Natural Resources
Division of Fish and Wildlife
Wildlife Populations and Research Unit
500 Lafayette Road, Box 20
St. Paul, MN 55155-4020
(651) 259-5203

September 2010 State of Minnesota, Department of Natural Resources

For more information contact:

DNR Information Center
500 Lafayette Road
St. Paul, MN 55155-4040
(651) 296-6157 (Metro Area)
1 888 MINNDNR (1-888-646-6367)

TTY (651) 296-5484 (Metro Area)
1 800 657-3929
http://www.mndnr.gov

This volume contains interim results of wildlife research projects. Some of the data and interpretations may change as a result of additional findings or future, more comprehensive analysis of the data. Authors should be contacted regarding use of any of their data.

Printed in accordance with Minn. Stat. Sec. 84.03

Printed on recycled paper containing a minimum of 30% post-consumer waste and soy-based ink.

TABLE OF CONTENTS

Forest Wildlife Research Group

Wetlands Wildlife Research Group

Wildlife Disease Program

Farmland Wildlife Research Group

Forest Wildlife Populations and Research Group
1201 East Highway 2
Grand Rapids, Minnesota 55744
(218) 327-4432

ECOLOGY AND POPULATION DYNAMICS OF BLACK BEARS IN MINNESOTA

David L. Garshelis and Karen V. Noyce

SUMMARY OF FINDINGS

During April 2009–March 2010, we monitored 45 radiocollared black bears (*Ursus americanus)* at 4 study sites representing contrasting portions of the bear's geographic range in Minnesota: Voyageurs National Park (VNP, northern extreme), Chippewa National Forest (CNF; central), Camp Ripley (southern fringe), and a site at the northwestern (NW) edge of the range. Hunting was the primary source (79%) of mortality in all areas, even though hunters were asked not to shoot radiocollared bears and bears cannot be legally hunted in 2 of the areas (but can be hunted when they wander outside). Reproduction was highest at the fringes of the bear range, at the NW study site followed by Camp Ripley, due largely to an abundance of oaks and hazelnut in these areas, as well as agricultural crops consumed by bears in the late summer–fall. Data from Global Positioning System (GPS)-radiocollars indicated that males in the NW made significant use of cropfields (corn, oats, sunflowers) from August–October (>30% of locations). Females in this area rarely used crops, but instead spent much of their time in shrublands. Continuation of this work will aim to explain this sex-related disparity in habitat use and predict whether further expansion of the bear range is possible.

INTRODUCTION

A lack of knowledge about black bear ecology and effects of harvest on bear populations spurred the initiation of a long-term telemetry-based research project on this species by the Minnesota Department of Natural Resources (MNDNR) in the early 1980s. For the first 10 years, the study was limited to the Chippewa National Forest (CNF), near the center of the Minnesota bear range (Figure 1). After becoming aware of significant geographic differences in sizes, growth rates, and productivity of bears across the state, apparently related to varying food supplies, we started other satellite bear projects in different study sites. Each of these began as a graduate student project, supported in part by the MNDNR. After completion of these student projects, we continued studies of bears at Camp Ripley Military Reserve, near the southern fringe of the Minnesota bear range, and in Voyageurs National Park (VNP), on the Canadian border (Figure 1).

These study sites differ enormously. The CNF is one of the most heavily hunted areas of the state, with large public (national, state, and county), heavily-roaded forests dominated by aspen (*Populus tremuloides, P. grandidentata)* of varying ages. Camp Ripley is unhunted, but bears may be killed by hunters when they range outside, which they often do in the fall, as the reserve is only 6–10 km wide. Oaks (*Quercus* sp.) are far more plentiful here than in the 2 study sites farther north, and agricultural fields (corn) border the reserve. VNP, being a national park, is also unhunted, but again bears may be hunted when they range outside. Soils are shallow and rocky in this area, and foods are generally less plentiful than the other sites.

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

OBJECTIVES

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

METHODS

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

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

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

During the non-denning period we monitored mortality of radio-instrumented bears from an airplane periodically through the summer. We listened to their radio signals, and if a pulse rate was in mortality mode (no movement of the collar in >4 hours), we tracked the collar on the ground to locate the dead animal or the shed radiocollar. If a carcass was located, we attempted to discern the cause of death. During the hunting season, hunters typically reported collared bears that they killed (but see Results).

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

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

RESULTS AND DISCUSSION

Radiocollaring and Monitoring

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

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

Mortality

Legal hunting has been the predominant cause of mortality among radiocollared bears from all study sites; 79% of mortalities that we observed were due, or likely due to hunting (Table 1). In earlier years of this study, hunters were encouraged to treat collared bears as they would any other bear so that the mortality rate of collared bears would be representative of the population at large. With fewer collared bears left in the study, and the focus now primarily on reproduction and habitat use rather than mortality, we sought to protect the remaining sample of bears. We asked hunters not to shoot radiocollared bears, and we fitted these bears with bright orange collars and colorful eartags so hunters could more easily see them. However, the mortality rate on collared bears has remained high even though some hunters reported avoiding them.

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

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

Natural mortality is a relatively minor cause of death among Minnesota bears >1 year old. Natural mortalities were most common in VNP (Table 1).

Reproduction

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

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

Average sex ratio of cubs shortly after birth was slightly, but consistently male-biased (pooled average across all areas = 52% male, n = 626 cubs examined). Observed year-to-year variation in cub sex ratios (Tables 3–6) was likely attributable to sampling error, although it is possible that some real year-to-year variation may occur as a result of varying food conditions.

Cub mortality was 21% for all areas pooled, but differences were observed among areas (range of means = 18–28%), with apparently the poorest survival in VNP (Tables 3–6). Across all areas, the mortality rate of male cubs was significantly (1.6x) higher than that of females (χ^2 = 6.7, P = 0.001), however, the predominant cause of cub mortality in Minnesota is not known.

Habitat Use of NW Bears

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

FUTURE DIRECTION

We plan to continue monitoring bears on these 4 study sites, although sample sizes have been greatly diminished by the exceedingly high harvest of collared bears in the past few years. Our main emphasis in the next few years will be at the NW study site, although for the past 2 years our data collection there has been limited by faulty GPS collars. We are hopeful that these issues have been solved for the coming year. In addition to gaining information from radiocollars, we have been and will continue to interview farmers to collect additional data on bear use of crops. This will yield an historical perspective on crop use, and provide insights into specific varieties of corn and sunflowers used by bears. Moreover, in the coming year we plan to obtain hair samples from hunter-killed bears in the NW for stable isotope analysis to ascertain the relative importance of corn in the diet, for males and females living in different parts of the study area. Ultimately we aim to create a habitat suitability map and thereby predict how far the bear population is likely to expand in this part of the state.

ACKNOWLEDGMENTS

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

LITERATURE CITED

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

Table 1. Causes of mortality of radiocollared black bears ≥1 year old from the Chippewa National Forest (CNF), Camp Ripley, Voyageurs National Park (VNP), and northwestern (NW) Minnesota, 1981–2009. Bears did not necessarily die in the area where they usually lived (e.g., hunting was not permitted within Camp Ripley or VNP, but bears were killed by hunters when they traveled outside these areas).

	CNF	Camp Ripley	VNP	NW	All combined
Shot by hunter	221	11	15	7	254
Likely shot by hunter[a]	8	1	0	2	11
Shot as nuisance	22	2	1	0	25
Vehicle collision	12	8	1	1	22
Other human-caused death	9	0	0	0	9
Natural mortality	7	3	4	0	14
Died from unknown causes	4	2	0	3	9
Total deaths	**283**	**27**	**21**	**13**	**344**

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

Table 2. Reproductive rates (cubs/female), mean litter size, and proportion of females with cubs (in all cases, counting only litters in which at least 1 cub survived 1 year) in winter dens (March) in 4 study sites (ordered from lowest to highest reproductive output): VNP (1997–2010), CNF (1981–2010), Camp Ripley (1991–2010), NW (2008-2010) (n = 4+ year-old female-years of observation).

	VNP (n = 62)			**CNF** (n = 409)			**Camp Ripley** (n = 55)			**NW** (n = 23)		
Age of female	Repro rate	Litter size	Prop w/ cubs	Repro rate	Litter size	Prop w/ cubs	Repro rate	Litter size	Prop w/ cubs	Repro rate	Litte r size	Prop w/ cubs
4–6 yrs	0.55	2.0	0.27	0.84	2.3	0.37	1.00	2.2	0.46	1.25	2.5	0.50
7–25 yrs	1.15	2.6	0.44	1.34	2.8	0.48	1.48	2.7	0.56	2.00	3.3	0.60
4–25 yrs	0.92	2.5	0.37	1.15	2.6	0.44	1.24	2.4	0.51	1.65	2.9	0.57

Table 3. Black bear cubs examined in dens of radiocollared mothers in or near the Chippewa National Forest during March, 1982–2010. High hunting mortality of radiocollared bears has reduced the sample size in recent years to the extent that the data are no longer suitable for monitoring.

Year	Litters checked	No. of cubs	Mean cubs/litter	% Male cubs	Mortality after 1 yr[a]
1982	4	12	3.0	67%	25%
1983	7	17	2.4	65%	15%
1984	6	16	2.7	80%	0%
1985	9	22	2.4	38%	31%
1986	11	27	2.5	48%	17%
1987	5	15	3.0	40%	8%
1988	15	37	2.5	65%	10%
1989	9	22	2.4	59%	0%
1990	10	23	2.3	52%	20%
1991	8	20	2.5	45%	25%
1992	10	25	2.5	48%	25%
1993	9	23	2.6	57%	19%
1994	7	17	2.4	41%	29%
1995	13	38	2.9	47%	14%
1996	5	12	2.4	25%	25%
1997	9	27	3.0	48%	23%
1998	2	6	3.0	67%	0%
1999	7	15	2.1	47%	9%
2000	2	6	3.0	50%	17%
2001	5	17	3.4	76%	15%
2002	0	0	—	—	—
2003	4	9	2.3	22%	0%
2004	5	13	2.6	46%	33%
2005	6	18	3.0	33%	28%
2006	2	6	3.0	83%	33%
2007	2	6	3.0	67%	17%
2008	1	3	3.0	100%	33%
2009	1	3	3.0	33%	33%
2010	1	4	4.0	100%	
Overall	**175**	**459**	**2.6**	**53%**	**18%**

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

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

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

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

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

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

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

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

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

[a] Cubs that were absent from their mother's den as yearlings were considered dead.
[b] Only one 5-cub litter was monitored, and all the cubs died (mother produced a litter of 4 cubs the next year).
[c] Excludes the total loss of the single 5-cub litter (which was not within the designated study area).

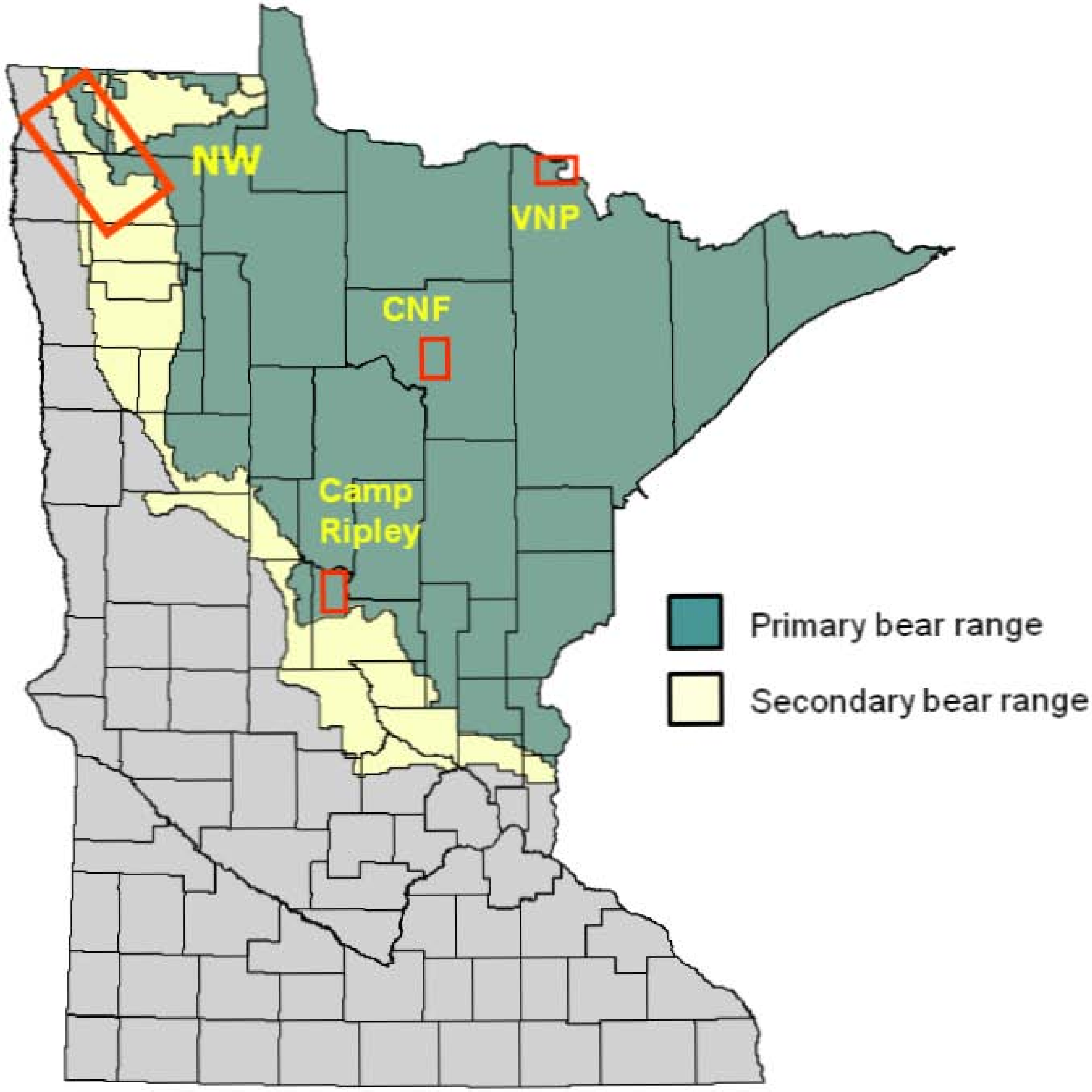


Figure 1. Location of 4 study sites within Minnesota's bear range: CNF (Chippewa National Forest, central bear range; 1981–2010); VNP (Voyageurs National Park, northern fringe of range; 1997–2010); Camp Ripley Military Reserve (near southern edge of range; 1991–2010); NW (northwestern fringe of range;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

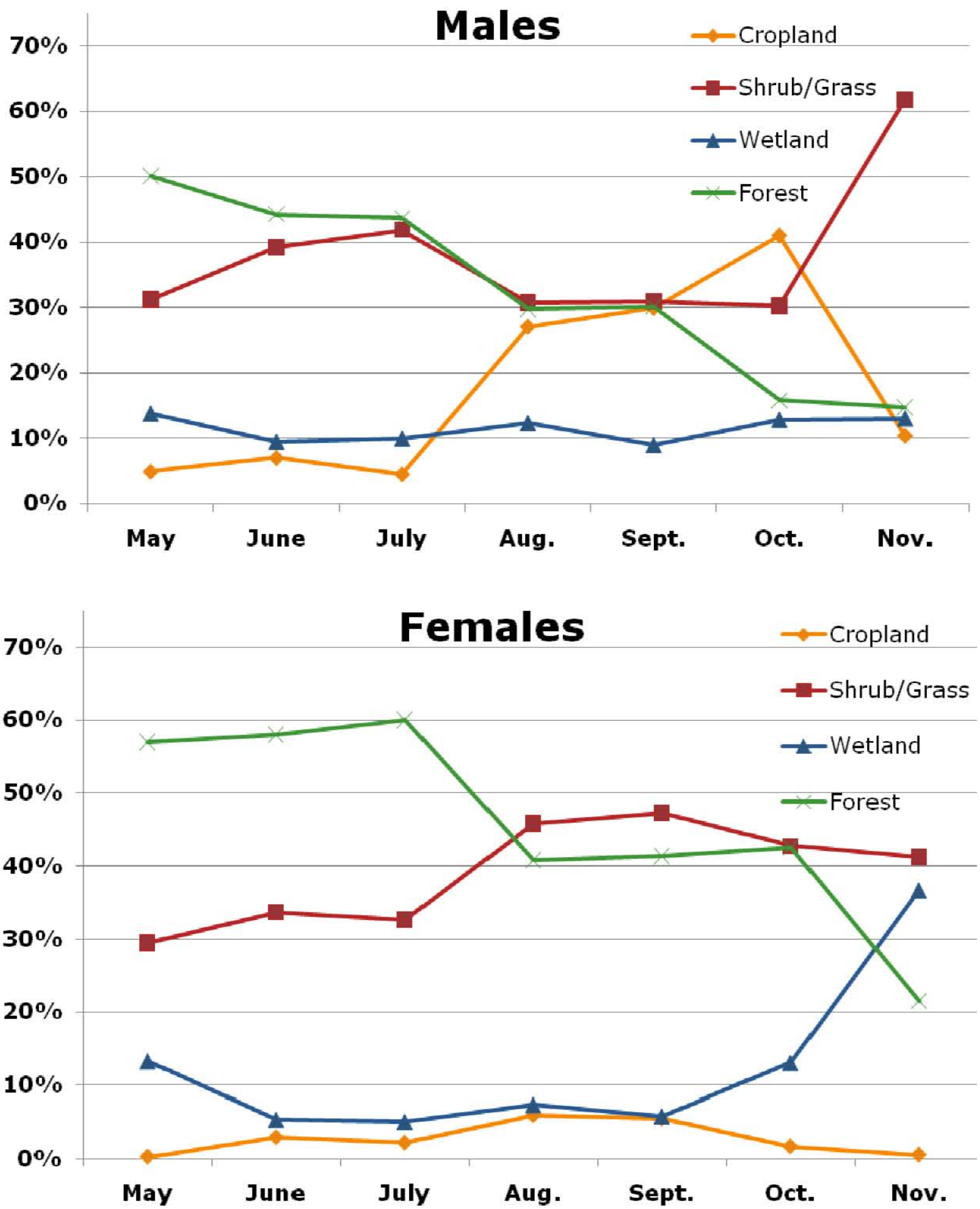


Figure 2. Trends in habitat use of black bears in Northwestern Minnesota, based on locations from GPS-radiocollars.

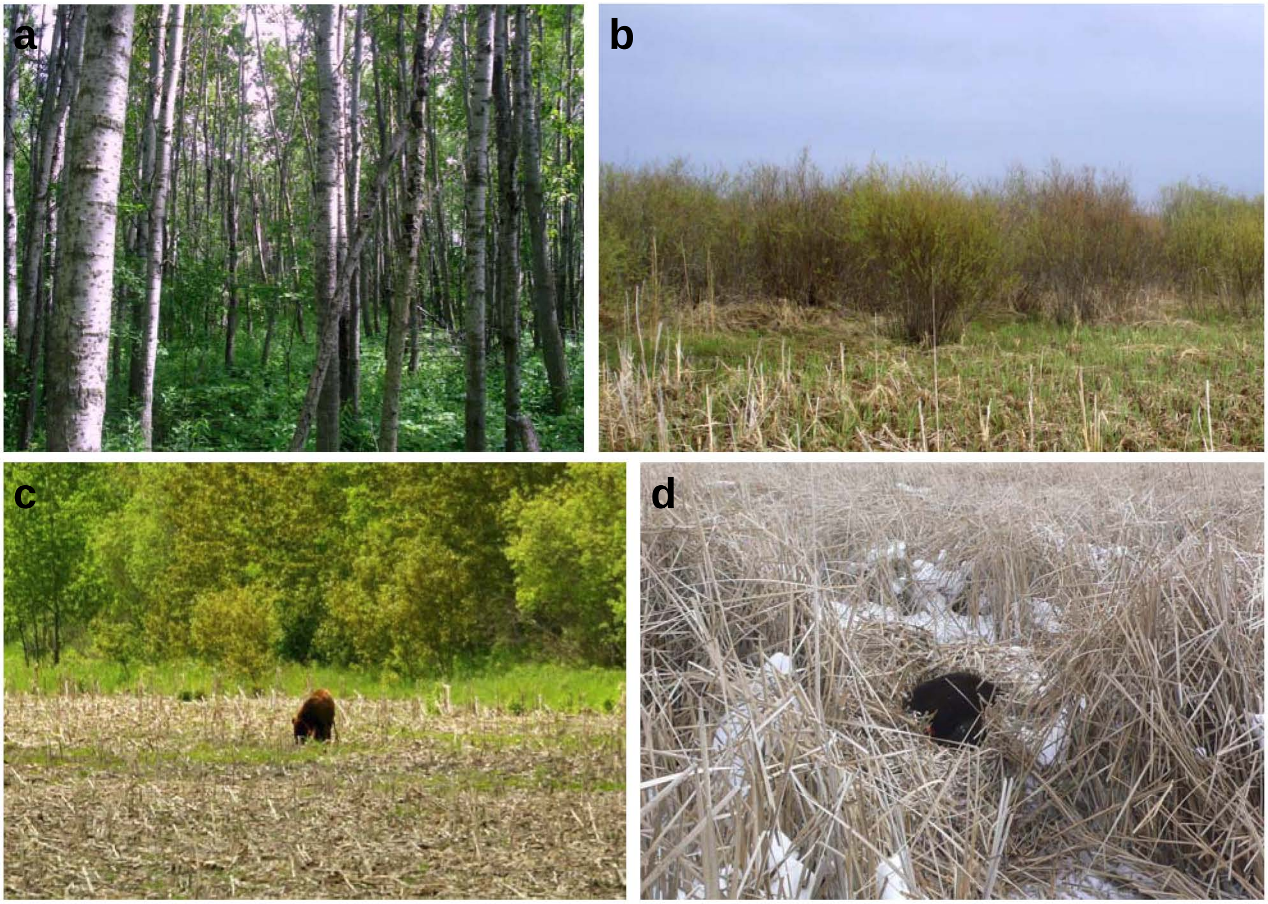


Figure 3. Examples of the main habitat types used by bears in the NW study site. (a) aspen forest; (b) lowland shrub, (c) cornfield (in this case, an unplowed field from the previous year, with a bear foraging on remnant cobs in June), (d) wetland, cattail swamp (used by denning bears). Photos: (a,b,c)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

REPRODUCTIVE ECOLOGY OF FISHER AND MARTEN IN MINNESOTA

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

INTRODUCTION

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

Legally, fisher and marten were unprotected in Minnesota prior to 1917, after which harvest season length restrictions were implemented. These protections were removed in the mid-1920s, and remained so until all harvest was prohibited in 1929. Seasons remained closed until 1977 for fisher and 1985 for marten, when limited harvests were reinstated. Since then, trapping zones and quotas have periodically increased to the current combined quota of 5 fisher/marten per trapper. While harvest is legal in approximately the northern 50% of the state, most marten harvest occurs in counties bordering Canada, particularly in northeast and north-central Minnesota. Fisher harvest occurs in most of the northern 50% of the state, though harvest is comparatively low in extreme northeast Minnesota (Lake and Cook counties), and lower, though perhaps increasing, in the Red River Valley (western Minnesota) and the highly fragmented transitional forests in central Minnesota. Peak harvest levels have been near 4,000 and 3,500 for marten and fisher, respectively. However, due to apparent multi-year population declines for both species, harvest seasons were reduced from 16 days to 9 days for the past 3 seasons, with harvests averaging 2,000 and 1,400 for marten and fisher, respectively.

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

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

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

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

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

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

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

STUDY AREA

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

Fisher research will take place in 3 areas (Figure 1; Areas 1, 2, and 3). The work in Area 3 is a collaborative effort between Camp Ripley Military Reservation, Central Lakes Community College, and the Minnesota Department of Natural Resources. While we do include animals captured in that area in our basic summaries, we do not discuss other aspects of that project in this report. Area 2 (1075 km^2), our primary fisher study area, is composed of 74% deciduous forest, 11% open water, 5% lowland conifer or bog, 5% marsh and fen, 2% regenerating forest (deciduous and coniferous), 1% coniferous forest, 1% grassland, and 1% mixed forest. Area 2 is 67% public ownership, including portions of the Chippewa National Forest and State and county lands. Extremely few martens occupy Area 2.

METHODS

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

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

During processing, we placed animals on chemical hand warmers or heating pads connected to a power inverter and 12 volt battery. Portable shelters and propane heaters were also used to keep animals warm during processing. We monitored respiration, pulse, and rectal temperature during anesthesia. We weighed and sexed animals and typically removed a first pre-molar for aging. Morphological measurements taken included body length, tail length, hind foot length, and chest, neck, and head circumference. We removed guard hair samples for possible genotyping, and for evaluating the use of stable isotope analysis for deciphering food habits (Ben-David et al. 1997). To assist with determining which females would likely produce kits, blood samples were drawn when possible to measure serum progesterone level in each animal (Frost et al. 1997). All blood samples were sent to the University of Minnesota Veterinary Diagnostics Lab for progesterone analysis. Antibiotics were administered subcutaneously to all animals prior to release.

During the pilot year, we deployed several radiocollar designs on fisher, including an ATS M1585 zip-tie collar (~ 43 g), an ATS M1930 collar (~ 38 g), and a Lotec SMRC-3 collar (~ 61 g; deployed on adult males only). Since the pilot year, we have primarily deployed ATS M1940 (~ 43 g) or Sirtrack TVC-162 collars (~ 45 g) on fisher. The majority of martens have been fitted with Holohil MI-2 collars (~ 31 g). We retrofitted each collar with a temperature data logger, in part to assist with determination of exact parturition date.

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

When a natal or maternal den was confirmed, we recorded den location (above/on/below-ground) as well as various location-specific details (e.g., tree species, log/tree diameter, burrow entrance attributes, etc). We note that since birth is never observed, and kits may be moved to new dens within days following birth, distinguishing natal dens from maternal dens can rarely be done with certainty. Hence, while we report our best assessment of den type, our focus is ultimately on determining whether initial dens (be they natal or maternal) used early in the kit-rearing period (e.g., prior to June 1) are structurally different than dens used as kits get larger and more mobile. Hence, we organize our tabular reporting on the date at which the den was first documented to be in use.

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

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

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

RESULTS AND DISCUSSION

Including the pilot year of the study, a total of 86 martens (44F, 42M) and 45 fishers (25F, 20M) have been captured. Herein we provide a basic summary of data collected to date on den structures, pregnancy status, and litter size. Because tooth aging has not yet been completed for all animals, and some yet-to-be-aged females may be only 1 year of age (i.e., not capable of producing kits), we present results only for animals known to be ≥ 2 years of age during spring den visits, or those of unknown age but for which we have confirmed parturition at the time of this writing (i.e., until age is known, we do not include animals that we have confirmed to be nulliparous).

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

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

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

A total of 17 fisher natal or maternal dens have been located to date (Table 4). We have not confirmed sufficient numbers of dens used later in the kit rearing process (after 1 June) to evaluate whether the type of den structures they use changes as kits get older. Based on 16 fisher natal/maternal dens confirmed prior to 1 June of each year, 100% have been in tree cavities, primarily large-diameter aspen. Pooling all tree species, average DBH for natal and maternal den trees is ~ 22 inches. The only fisher maternal den confirmed after 1 June was in a large diameter hollow log on the ground (Table 4).

ACKNOWLEDGEMENTS

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

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.

Balser, 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.

Conaway, C. H. 1955. Embryo resorption and placental scar formation in the rat. Journal of Mammalogy 36: 516-532.

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.

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.

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. The 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. The 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.

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 status and litter size for radiocollared female marten[1] in Minnesota.

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

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

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

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

Table 3. Parturition status and litter size for radiocollared female fisher[1] in Minnesota.

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

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

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

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

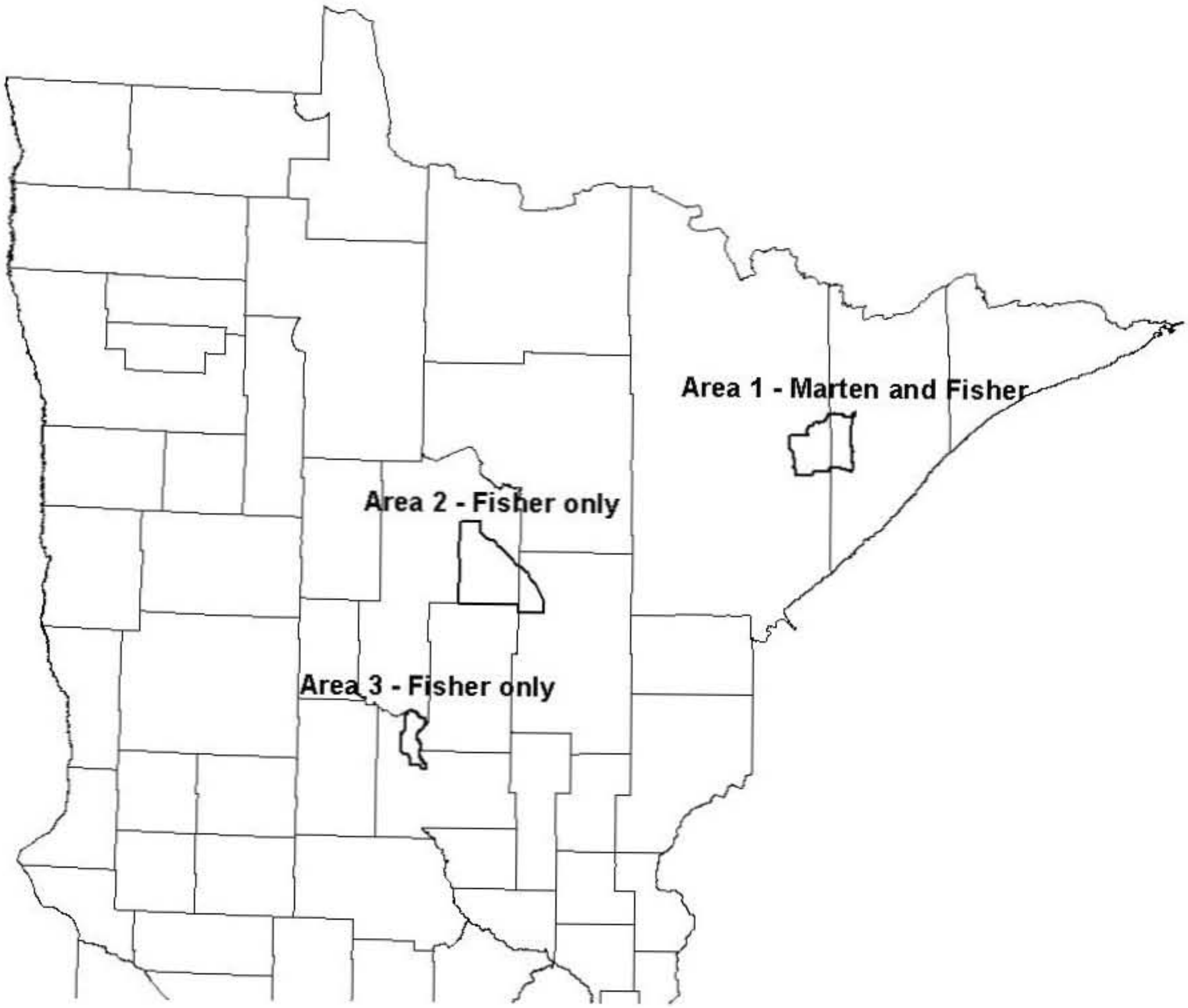


Figure 1. Fisher and marten study areas in Minnesota, 2008-2010.

SURVIVAL AND CAUSES OF MORTALITY FOR FISHER AND MARTEN IN MINNESOTA

John Erb, Barry Sampson, and Pam Coy

SUMMARY OF FINDINGS

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

INTRODUCTION

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

Legally, fisher and marten were unprotected in Minnesota prior to 1917, after which harvest season length restrictions were implemented. These protections were removed in the mid-1920s, and remained so until all harvest was prohibited in 1929. Seasons remained closed until 1977 for fisher and 1985 for marten, when limited harvests were reinstated. Since then, trapping zones and quotas have periodically increased to the current combined quota of 5 fisher/marten per trapper. While harvest is legal in approximately the northern 50% of the state, most marten harvest occurs in counties bordering Canada, particularly in northeast and north-central Minnesota. Fisher harvest occurs in most of the northern 50% of the state, though harvest is comparatively low in extreme northeast Minnesota (Lake and Cook counties), and lower, though perhaps increasing, in the Red River Valley (western Minnesota) and the highly fragmented transitional forests in central Minnesota. Peak harvest levels have been near 4,000 and 3,500 for marten and fisher, respectively. However, due to apparent multi-year population declines for both species, harvest seasons were reduced from 16 days to 9 days for the past 3 seasons, with harvests averaging 2,000 and 1,400 for marten and fisher, respectively.

While both species appear to have naturally re-colonized a significant portion of their historic range, Minnesota-specific information on survival and causes of mortality is limited. Except for harvest data, we are aware of only 1 published field study in Minnesota. Specifically, Mech and Rogers (1977) opportunistically radiocollared 4 marten and reported survival and home range information for those animals. This information is specific to marten, now nearly 30 years old, and based on a very limited sample size. Gathering cause-specific mortality information can be useful for informing population models, detecting unknown mortality agents, and guiding management remedies to any population declines of concern.

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

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

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

STUDY AREA

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

Fisher research will take place in 3 areas (Figure 1; Areas 1, 2, and 3). The work in Area 3 is a collaborative effort between Camp Ripley Military Reservation, Central Lakes Community College, and the Minnesota Department of Natural Resources. While we do include animals captured in that area in our basic summaries, we do not discuss other aspects of that project in this report. Area 2 (1075 km^2), our primary fisher study area, is composed of 74% deciduous forest, 11% open water, 5% lowland conifer or bog, 5% marsh and fen, 2% regenerating forest (deciduous and coniferous), 1% coniferous forest, 1% grassland, and 1% mixed forest. Area 2 is 67% public ownership, including portions of the Chippewa National Forest and State and county lands. Extremely few martens occupy Area 2.

METHODS

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

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

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

During the pilot year, we deployed several radiocollar designs on fisher, including an ATS M1585 zip-tie collar (~ 43 g), an ATS M1930 collar (~ 38 g), and a Lotec SMRC-3 collar (~ 61 g; deployed on adult males only). Since the pilot year, we have primarily deployed ATS M1940 (~ 43 g) or Sirtrack TVC-162 collars (~ 45 g) on fisher. The majority of martens in both years have been fitted with Holohil MI-2 collars (~ 31 g). While not discussed in detail here, we retrofitted each collar with a temperature data logger, in part to allow for determination of exact time of death.

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

RESULTS AND DISCUSSION

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

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

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

Of the 45 fishers radiocollared, 14 are still being monitored (8F, 6M), 3 are missing, 10 shed their collars due to belting design failures, and 1 collar was removed at the time of recapture due to neck abrasion. In addition, 3 juvenile males were ear-tagged only. Of the 14 known deaths (8F, 6M), 1 was struck by a vehicle, 1 was accidentally trapped out of season, 2 were legally trapped, 2 died from unknown but apparently natural causes, and 8 (6F, 2M) were killed by other predators (scavenging by an eagle can't be ruled out in 1 case).

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

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

ACKNOWLEDGEMENTS

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

LITERATURE CITED

Balser, D. S., and W. H. Longley. 1966. Increase of the fisher in Minnesota. Journal of Mammalogy 47: 547-550.

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.

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.
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., 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.
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., 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.
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. 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.

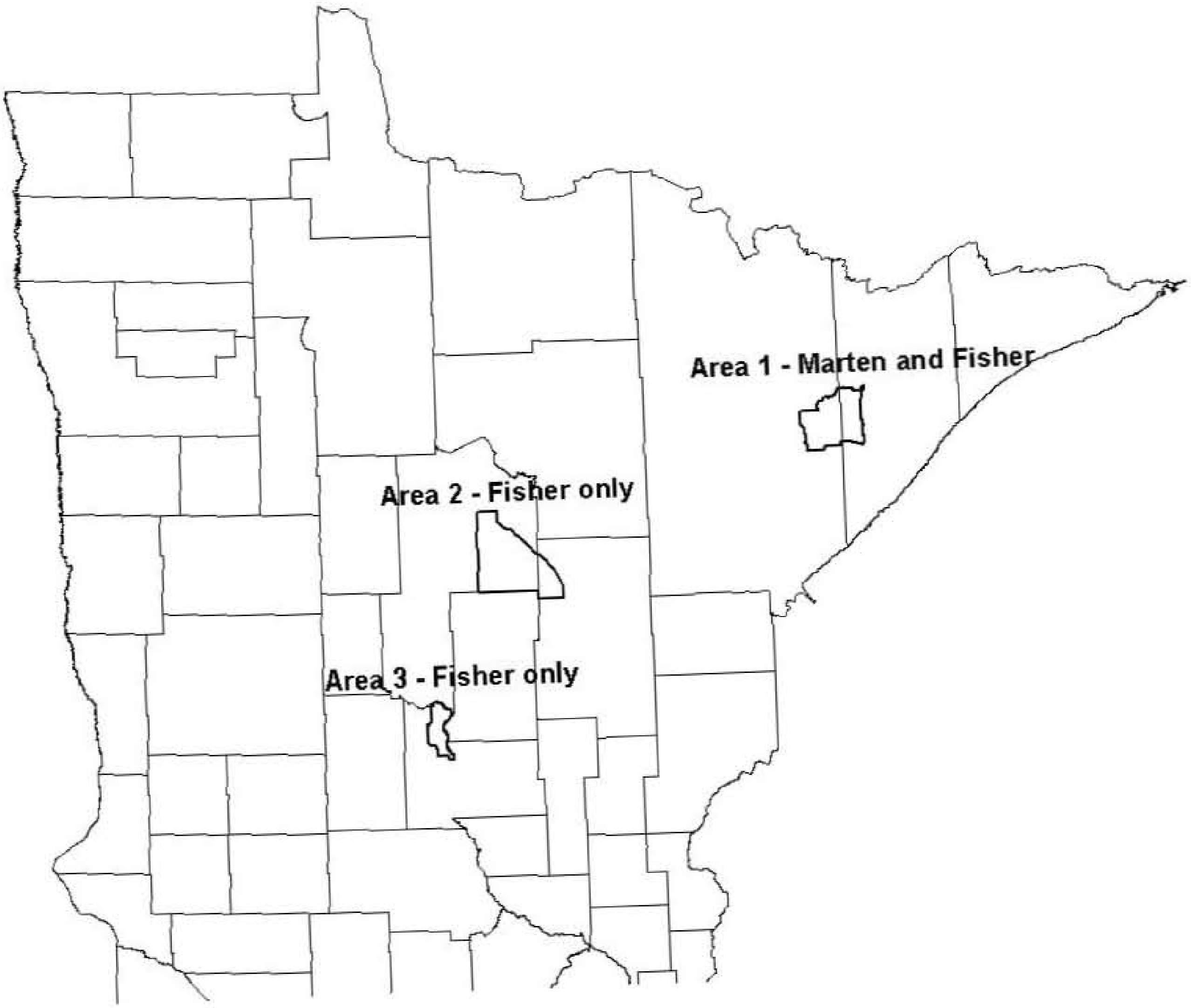


Figure 1. Fisher and marten study areas in Minnesota 2008-2010.

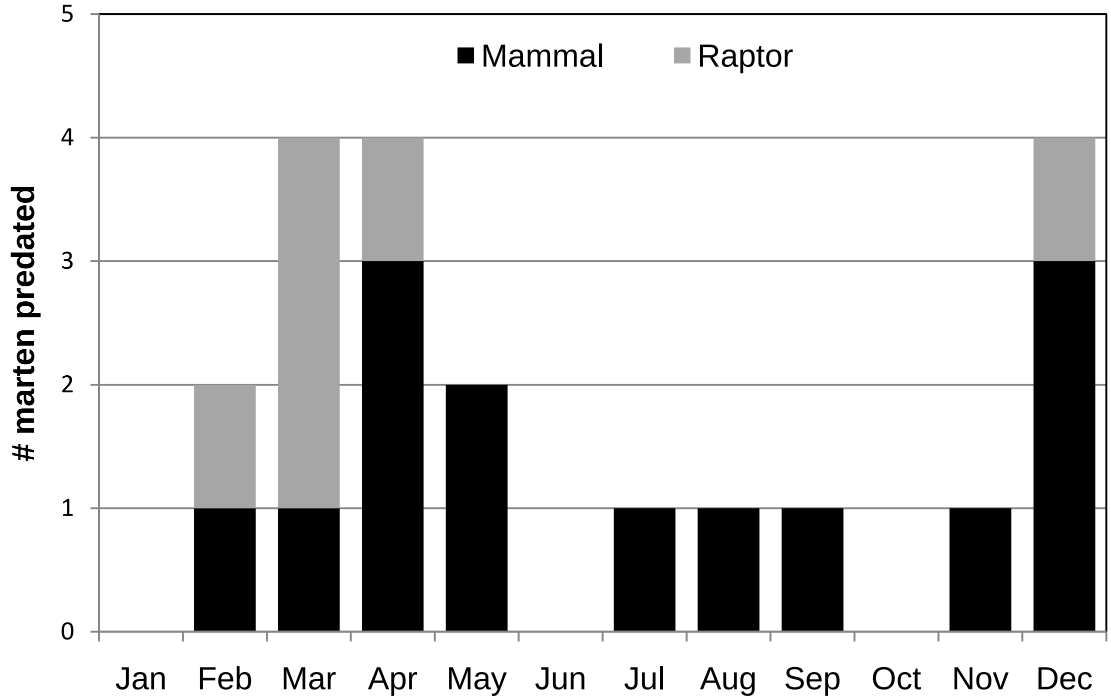


Figure 2. Seasonal timing of marten deaths attributable to predation in northeast Minnesota, 2007-2009.

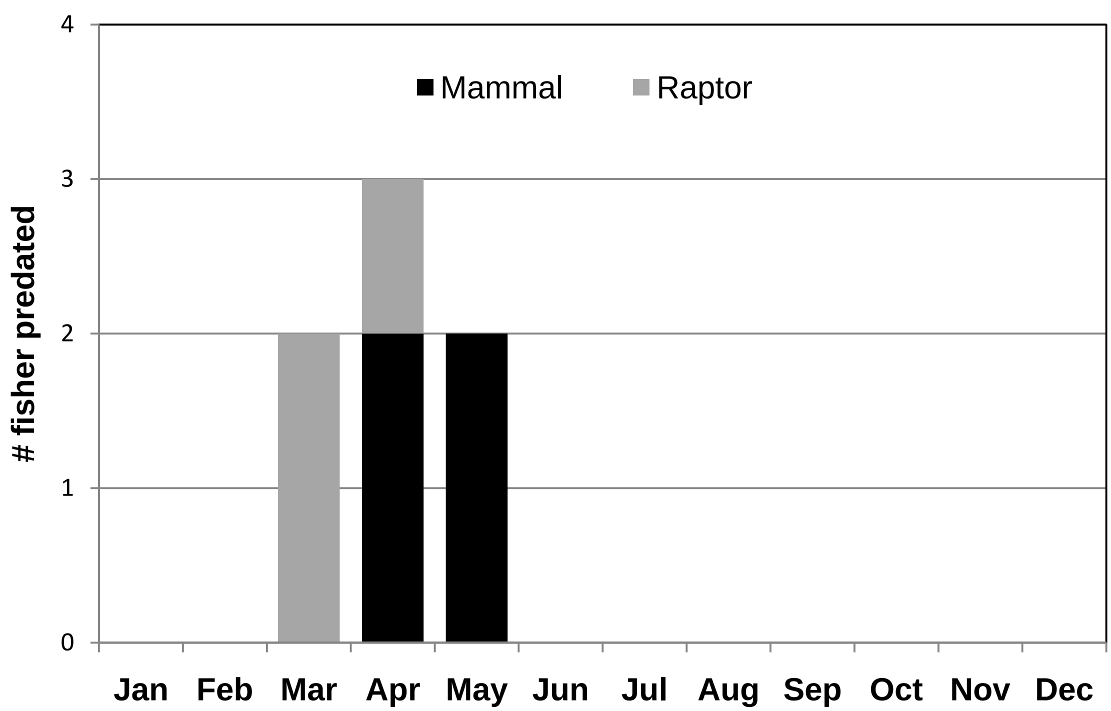


Figure 3. Seasonal timing of fisher deaths attributable to predation in north-central Minnesota, 2007-2009.

MOOSE POPULATION DYNAMICS IN NORTHEASTERN MINNESOTA

Mark S. Lenarz, Michael W. Schrage[1], Andrew J. Edwards[2], and Michael Nelson[3]

SUMMARY OF FINDINGS

We captured and radiocollared a total of 150 adult moose (*Alces alces*, 55 adult males and 95 adult females) between 2002 and 2008. As of 1 April 2010, 105 collared moose (48 adult males and 57 adult females) have died. Annual mortality rates varied among years, and generally were higher than found elsewhere in North America. Estimates of fertility for this population were also low compared with other North American moose populations. Data analyses from this research are progressing and 2 manuscripts are published, 1 manuscript is in press, and 2 other manuscripts are in preparation.

INTRODUCTION

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

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

METHODS

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

[1] Fond du Lac Resource Management Division, 1720 Big Lake Road, Cloquet, Minnesota, 55720, USA
[2] 1854 Authority, 4428 Haines Road, Duluth, Minnesota, 55811, USA
[3] United States Geological Survey, Northern Prairie Wildlife Research Center, Jamestown, North Dakota , 58401, USA

RESULTS AND DISCUSSION

As of 1 April 2010, 105 collared moose (48 adult males and 57 adult females) have died. In addition, 1 moose slipped its collar, 1 moose moved out of the study area, and we lost contact (apparent transmitter failure) with 2 moose. Moose that died within 2 weeks of capture (6) were designated as capture mortality. Hunters killed 17 moose, 2 were poached, and 11 were killed in collisions with vehicles (cars, trucks, or trains). The remaining mortality (69) was considered to be non-anthropogenic and causes included wolf predation (8), bacterial meningitis (1), or unknown (60).

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

Annual non-hunting mortality rates (1 June to 31 May) for adult moose averaged 18% for males (0 to 40%, SE = 5, n = 7) and 21% for females (5 to 30%, SE = 3, n = 7; Table 1). Sex did not contribute to the prediction of survival (χ^2 = 0.001, P = 0.98), which implies that there was no difference in survival rates (non-hunting) between adult male and female moose. Non-hunting mortality was substantially higher than documented for populations outside of Minnesota (generally 8 to 12%; Ballard, 1991, Bangs 1989, Bertram and Vivion 2002, Kufeld and Bowden 1996, Larsen et al. 1989, Mytton and Keith 1981, Peterson 1977) and similar to that observed for adult moose in northwestern Minnesota (21%; Murray et al. 2006).

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

Between 2004 and 2009, 197 radiocollared adult females gave birth to a minimum of 167 calves (96 singles, 34 twins, and 1 set of triplets; M. W. Schrage, Fond du Lac Resources Management Division, unpublished). The annual ratio of calves: radiocollared females ranged from 0.53 to 0.95 ($\bar{x}$ = 0.82, SE = 0.06, n = 6). These estimates were biased low because in 4 of 6 years, radiocollared females not accompanied by calves during the MJ survey were subsequently observed to be accompanied by a single calf (4 in 2004, 4 in 2005, 1 in 2007, 4 in 2008). It is also possible that post natal mortality occurred prior to the MJ survey. Nonetheless, these estimates are low compared with other locations in North America. Boer (1992), for example, reported estimates ranging from 0.88 to 1.24 calves/adult female, in moose populations above and below K carrying capacity, respectively.

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

ACKNOWLEDGMENTS

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

LITERATURE CITED

Ballard, W. B., J. S. Whitman, and D. J. Reed. 1991. Population dynamics of moose in south-central Alaska. Wildlife Monograph 114.

Bangs, E. E., T. N. Bailey, and M. F. Portner. 1989. Survival rates of adult female moose on the Kenai Peninsula, Alaska. Journal of Wildlife Management 53:557-563.

Bertram, M. R., and M. T. Vivion. 2002. Moose mortality in eastern interior Alaska. Journal of Wildlife Management 66:747-756.

Boer, A. H. 1992. Fecundity of North American moose (Alces alces): a review. Alces (Supplement) 1:1-10.

Cox, D. R. 1972. Regression models and life tables. Journal of the Royal Statistical Society, Series B 20:187-220.

Kaplan, E. L. and O. Meier. 1958. Non-parametric estimation from incomplete observations. Journal of the American Statistical Association 53:457-481.

Kufeld, R. C., and D. C. Bowden. 1996. Survival rates of Shiras moose (Alces alces shirasi) in Colorado. Alces 32: 9-13.

Larsen, D. G., D. A. Gauthier, and R. L. Markel. 1989. Cause and rate of moose mortality in the southwest Yukon. Journal of Wildlife Management 53:548-557.

Lenarz, M. S., M. E. Nelson, M. W. Schrage, and A. J. Edwards. 2009. Temperature mediated survival in northeastern Minnesota Moose. Journal of Wildlife Management. 73:503-510.

Murray, D. L., E. W. Cox, W. B. Ballard, H. A. Witlaw, 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.

Mytton, W. R., and L. B. Keith. 1981. Dynamics of moose populations near Rochester, Alberta, 1975-1978. Canadian Field-Naturalist 95:39-49.

Peterson, R. O. 1977. Wolf ecology and prey relationships on Isle Royale. National Park Service Scientific Monograph 11. Washington, D.C., USA.

Pollock, K. H., S.R. Winterstein, C.M. Bunck, and P.D. Curtis. 1989. Survival analysis in telemetry studies: the staggered entry design. Journal of Wildlife Management 53:7-15.

SAS Institute. 2008. Version 9.1, SAS Institute, Cary, North Carolina, USA.

Schwartz, C. C. 1997. Reproduction, natality and growth. Pages 141-172 *in* A. W. Franzmann and C. C, Schwartz , editors. Ecology and management of North American moose. Smithsonian Institution Press, Washington, D. C., USA.

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

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

[1] Period: 1 June – 31 May.

[2] Sample size as of 31 May

LIVING ON THE EDGE: VIABILITY OF MOOSE IN NORTHEASTERN MINNESOTA[1]

Mark S. Lenarz, John Fieberg, Michael W. Schrage[2], and Andrew J. Edwards[3]

ABSTRACT

North temperate species on the southern edge of their distribution are especially at risk to climate induced changes. One such species is the moose (*Alces alces*), whose continental United States distribution is restricted to northern states or northern portions of the Rocky Mountain cordillera. We used a series of matrix models to evaluate the demographic implications of estimated survival and reproduction schedules for a moose population in northeastern Minnesota, USA, between 2002 and 2008. We used data from a telemetry study to calculate adult survival rates and estimated calf survival and fertility of adult females using results of helicopter surveys. Estimated age- and year-specific survival rates showed a sinusoidal temporal pattern during our study and were lower for younger and old aged animals. Estimates of annual adult survival (when assumed to be constant for ages >1.7 yr old) ranged from 0.74 – 0.85. Annual calf survival averaged 0.40 and the annual ratio of calves born to radiocollared females averaged 0.78. Point estimates for the finite rate of increase (λ) from yearly matrices ranged from 0.67 to 0.98 during our 6-year study, indicative of a long-term declining population. Assuming each matrix to be equally likely to occur in the future, we estimated a long-term stochastic growth rate of 0.85. Even if heat stress is not responsible for current levels of survival, continuation of this growth rate will ultimately result in a northward shift of the southern edge of moose distribution. Population growth rate, and its uncertainty, was most sensitive to changes in estimated adult survival rates. The relative importance of adult survival to population viability has important implications for harvest of large herbivores and the collection of information on wildlife fertility.

---

1. Abstract of paper in press in Journal of Wildlife Management.
2. Fond du Lac Resource Management Division, 1720 Big Lake Road, Cloquet, Minnesota, 55720, USA.
3. 1854 Treaty Authority, 4428 Haines Road, Duluth, Minnesota, 58811, USA.

WINTER BODY CONDITION OF MOOSE (*ALCES ALCES*) IN A DECLINING POPULATION IN NORTHEASTERN MINNESOTA[1]

Glenn D. DelGiudice, Barry A. Sampson, Mark S. Lenarz, Michael W. Schrage[2], and Andrew J. Edwards[3]

ABSTRACT

Assessments of the condition of moose (*Alces alces*) may be particularly informative to understanding the dynamics of populations and other influential factors. During February-March 2003 to 2005, we assessed the nutritional condition of 79 moose (39 females, 40 males) in northeastern Minnesota by body condition scoring (BCS_F, scale of 0-10), and 67 of these by ultrasonographic measurements of rump fat (Maxfat), which was used to estimate ingesta-free body fat (IFBF) in all but 2 of these females. Scores of the BCS_F were related (r^2 = 0.34, P< 0.0001) to Maxfat. Body condition scores were not affected by sex X capture-year, capture-year, or age-at-capture, but the mean body condition score of males (6.5 ± 0.2 [SE], n = 40) was less ($P\leq$ 0.009) than that of females (7.4 ± 0.2, n = 39). Overall, Maxfat ranged from 0 to 4.6 and 0.3 to 2.8 cm in females and males, respectively, and was unaffected by age-at-capture. There was a sex X capture-year effect (P = 0.021) on Maxfat; mean values were stable for males during winters 2003 to 2005, but in females were lowest during 2003, consistent with lowest pregnancy rates and lowest winter and spring survival compared to 2004 and 2005. Based on estimates of % IFBF, late winter-early spring survival in 2003 of at least 6.1% of the collared animals assessed by Maxfat, 11.8% of the adult females specifically may have been seriously challenged directly by poor condition. Data from this study provide reference values, and assessments of body condition of moose will be an essential component of the additional comprehensive research needed to more closely examine and better understand relations of seasonal heat stress, nutrition, body condition, habitat use, and performance of this important remaining viable, but declining population. We will concentrate on improving the reliability of the BCS_F to extend the range of IFBF estimation (once rump fat is depleted) using an index that combines BCS_F scores and Maxfat measurements.

---

[1]Journal of Wildlife Diseases (2010) *Accepted*.
[2]Fond du Lac Resource Management Division, 1720 Big Lake Road, Cloquet, MN, USA 55720
[3]1854 Treaty Authority, 4428 Haines Road, Duluth, MN, USA 55811

HABITAT SELECTION BY MALE RUFFED GROUSE AT MULTIPLE SPATIAL SCALES

Meadow J. Kouffeld[1], Michael A. Larson, and R. J. Gutiérrez[1]

SUMMARY OF FINDINGS

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

INTRODUCTION

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

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

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

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

At the scale of a few forest stands, the preference of grouse for aspen in several age classes is well known (Gullion 1984, Rusch et al. 2000). Zimmerman (2006) found that variation in the number of drumming male grouse in individual forest stands was best explained by a model that included patch shape and 9 forest overstory types. More grouse were located in

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

young aspen stands and stands with low edge density, and fewer were in mixed hardwood-conifer stands and mature spruce-fir stands. Less is known, however, about the influence on grouse of the following patch and adjacency characteristics of forest stands: the presence of conifers in aspen stands, the presence of aspen clones in conifer stands, the relative importance of different age classes of aspen, and variation in the density of woody stems regenerating after harvesting aspen.

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

OBJECTIVES

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

STUDY AREA

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

Six of the 9 potential study sites had >7 times as much area in the aspen cover type than in conifer cover types. The other 3 potential study sites had the most conifer cover (10–24%, ratios of conifer:aspen area = 0.46–1.33). The site with the most conifer cover was adjacent to 1 of the 6 aspen sites, so we selected these 2 adjacent sites to comprise our study area. The study area is in portions of Red Lake Wildlife Management Area and adjacent Beltrami Island State Forest. We did not include Red Lake Band Tribal Lands in our study.

METHODS

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

Data Collection

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

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

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

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

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

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

Data Analysis

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

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

RESULTS

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

ACKNOWLEDGEMENTS

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

LITERATURE CITED

Archibald, H. L. 1975. Temporal patterns of spring space use by ruffed grouse. Journal of Wildlife Management 39:472–481.

Burnham, K. P., and D. R. Anderson. 2002. Model selection and multimodel inference: a practical information-theoretic approach. Second edition. Springer-Verlag, New York, USA.

Devers, P. K., D. F. Stauffer, G. W. Norman, D. E. Steffen, D. M. Whitaker, J. D. Sole, T. J. Allen, S. L. Bittner, D. A. Buehler, J. W. Edwards, D. E. Figert, S. T. Friedhoff, W. W. Giuliano, C. A. Harper, W. K. Igo, R. L. Kirkpatrick, M. H. Seamster, J. Spiker, Harry A., D. A. Swanson, and B. C. Tefft. 2007. Ruffed grouse population ecology in the Appalachian Region. Wildlife Monographs 168.

Gullion, G. W. 1984. Managing northern forests for wildlife. Ruffed Grouse Society, Coraopolis, Pennsylvania, USA.

Gullion, G. W. 1990. Ruffed grouse use of conifer plantations. Wildlife Society Bulletin 18:183–187.

Gullion, G. W, and A. A. Alm. 1983. Forest management and ruffed grouse populations in a Minnesota coniferous forest. Journal of Forestry 81:529–536.

Keating, K. A., and S. Cherry. 2004. Use and interpretation of logistic regression in habitat-selection studies. Journal of Wildlife Management 68:774–789.

MNDNR. 2001. Issues, strategies, desired future forest composition, and stand-selection criteria. Border Lakes Subsection Forest Resource Management Plan. St. Paul, Minnesota, USA.

MNDNR. 2003. Issues, strategies, desired future forest composition, and stand-selection criteria. Mille Lacs Uplands Subsection Forest Resource Management Plan. St. Paul, Minnesota, USA.

MNDNR. 2004 (revised 2006). Strategic direction document. North Shore area Subsection Forest Resource Management Plan. St. Paul, Minnesota, USA.

MNDNR. 2007. A strategic conservation agenda 2003-2007: measuring progress toward mission. Minnesota Department of Natural Resources, St. Paul, USA.

Rusch, D. H., S. DeStefano, M. C. Reynolds, and D. Lauten. 2000. Ruffed grouse. Number 515 *in* A. F. Poole, and F. B. Gill, editors. The birds of North America. Academy of Natural Sciences and American Ornithologists' Union, Philadelphia, Pennsylvania, USA.

USFS. 2004*a*. Land and resource management plan: Chippewa National Forest. USDA Forest Service, Eastern Region, Milwaukee, Wisconsin, USA.

USFS. 2004*b*. Land and resource management plan: Superior National Forest. USDA Forest Service, Eastern Region, Milwaukee, Wisconsin, USA.

Zimmerman, G. S. 2006. Habitat and population regulation of ruffed grouse in Minnesota. Dissertation, University of Minnesota, St. Paul, USA.

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

Michael A. Larson

SUMMARY OF FINDINGS

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

INTRODUCTION

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

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

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

OBJECTIVES

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

STUDY AREA

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

METHODS

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

ACKNOWLEDGEMENTS

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

LITERATURE CITED

Anderson, L. J. 1973. Habitat use, behavior, territoriality, and movements of the male Canada spruce grouse of northern Minnesota. Thesis, University of Minnesota, St. Paul, USA.

Bouta, R. P. 1991. Population status, historical decline, and habitat relationships of spruce grouse in the Adirondacks of New York. Thesis, State University of New York, Syracuse, USA.

Frelich, L. E., and P. B. Reich. 2009. Will environmental changes reinforce the impact of global warming on the prairie-forest border of central North America? Frontiers in Ecology and the Environment, URL: http://www.esajournals.org/doi/pdf/10.1890/080191

Gregg, L., B. Heeringa, and D. Eklund. 2004. Conservation assessment for spruce grouse (Falcipennis canadensis). USDA Forest Service, Eastern Region, Park Falls, Wisconsin, USA.

Haas, G. H. 1974. Habitat selection, reproduction, and movements in female spruce grouse. Dissertation, University of Minnesota, St. Paul, USA.

Healy, W. M., R. O. Kimmel, D. A. Holdermann, and W. Hunyadi. 1980. Attracting ruffed grouse broods with tape-recorded chick calls. Wildlife Society Bulletin 8:69–71.

Joyce, L. A., C. H. Flather, and M. Koopman. 2008. Analysis of potential impacts of climate change on wildlife habitats in the U.S. Final Project Report, USDA Forest Service, Rocky Mountain Research Station, Fort Collins, Colorado, USA.

Keppie, D. M. 1992. An audio index for male spruce grouse. Canadian Journal of Zoology 70:307–313.

MNDNR. 2006. Tomorrow's habitat for the wild and rare: an action plan for Minnesota wildlife. Division of Ecological Services, St. Paul, Minnesota, USA.

Pietz, P. J., and J. R. Tester. 1979. Utilization of Minnesota peatland habitats by snowshoe hare, white-tailed deer, spruce grouse, and ruffed grouse. Final Technical Report, Minnesota Department of Natural Resources, St. Paul, USA.

Ross, A. M., and G. Johnson. 2008. Spruce grouse in lowland boreal forests of New York State: distribution, movements and habitat. Final Report AMO5122, Grant T-2-1, New York State Department of Environmental Conservation, Albany, USA.

Williamson, S. J., D. M. Keppie, R. Davison, D. Budeau, S. Carrière, D. Rabe, and M. A. Schroeder. 2008. Spruce grouse continental conservation plan. Association of Fish and Wildlife Agencies, Washington, DC, USA.

LANDSCAPE CHARACTERISTICS ASSOCIATED WITH DANCING GROUNDS OF SHARP-TAILED GROUSE

Michael A. Larson and J. Wesley Bailey

SUMMARY OF FINDINGS

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

INTRODUCTION

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

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

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

OBJECTIVES

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

STUDY AREA

We defined the study area as occurring within both of 2 different boundaries for describing the geographic extent of sharp-tailed grouse range in Minnesota (Figure 1). One boundary encompassed the subsections of Minnesota's Ecological Classification System (ECS, following Cleland et al. 1997) where dancing grounds were observed during 1991–1993. The sample of dancing ground locations that we used is described and justified in the METHODS section below. The other boundary was the 85% kernel density estimate around observed dancing grounds. We selected the 85% kernel boundary because it encompassed 21% less area than the 95% kernel boundary and excluded only 1% of the used sites. The 80% kernel boundary encompassed 32% less area than the 95% kernel boundary, but we thought it excluded too many used sites (5%).

METHODS

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

Use-Availability Data

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

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

Landscape Data

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

Then we superimposed (i.e., replaced the MN-GAP data with) data from better sources for 3 of the cover types. Using the National Wetlands Inventory (NWI, Cowardin et al. 1979, Minnesota Land Management Information Center 2007) we selected scrub-shrub (broad-leaved deciduous and deciduous) and persistent emergent types that occurred within flooded, saturated, and seasonally flooded NWI water regime modifiers. We added the NWI scrub-shrub areas to our shrub cover type and the persistent emergent areas to our sedge meadow cover type, regardless of what the MN-GAP classification was. Then we added areas with herbaceous vegetation cover practices from the 1997 Conservation Reserve Program (CRP, Minnesota Natural Resources Conservation Service 2010) to our undisturbed grass cover type, regardless of what the MN-GAP or NWI classifications were.

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

Model Set

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

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

Model Fitting

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

RESULTS AND DISCUSSION

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

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

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

ACKNOWLEDGEMENTS

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

LITERATURE CITED

Berg, W. E. 1997. The sharp-tailed grouse in Minnesota. Minnesota Department of Natural Resources, Wildlife Report Number 10, St. Paul, USA.

Cleland, D. T., P. E. Avers, W. H. McNab, M. E. Jensen, R. G. Bailey, T. King, and W. E. Russell. 1997. National hierarchical framework of ecological units. Pages 181–200 *in* M. S. Boyce, and A. Haney, editors. Ecosystem management applications for sustainable forest wildlife resources. Yale University Press, New Haven, Connecticut, USA.

Cowardin, L. M., V. Carter, F. Golet, and E. LaRoe. 1979. Classification of wetlands and deepwater habitats of the United States. U.S. Fish and Wildlife Service, Washington, D.C., USA.

Dexter, M. H. 2009. 2008 small game hunter mail survey. Pages 157–171 *in* M. H. Dexter, editor. Status of wildlife populations, fall 2009. Division of Fish and Wildlife, Minnesota Department of Natural Resources, St. Paul, USA.

Hanowski, J. M., D. P. Christian, and G. J. Niemi. 2000. Landscape requirements of prairie sharp-tailed grouse *Tympanuchus phasianellus campestris* in Minnesota, USA. Wildlife Biology 6:301–307.

Johnson, C. J., S. E. Nielsen, E. H. Merrill, T. L. McDonald, and M. S. Boyce. 2006. Resource selection functions based on use-availability data: theoretical motivation and evaluation methods. Journal of Wildlife Management 70:347–357.

Keating, K. A., and S. Cherry. 2004. Use and interpretation of logistic regression in habitat-selection studies. Journal of Wildlife Management 68:774–789.

Landwehr, T. J. 1984. A summary of Minnesota game regulations, harvest, and population statistics, 1858–1983. Minnesota Department of Natural Resources, St. Paul, USA.

Larson, M. A. 2008. Grouse surveys in Minnesota during spring 2008. Pages 55–63 *in* M. H. Dexter, editor. Status of wildlife populations, fall 2008. Division of Fish and Wildlife, Minnesota Department of Natural Resources, St. Paul, USA.

_____. 2009. Grouse surveys in Minnesota during spring 2009. Pages 73–81 *in* M. H. Dexter, editor. Status of wildlife populations, fall 2009. Division of Fish and Wildlife, Minnesota Department of Natural Resources, St. Paul, USA.

Lele, S. R. 2009. A new method for estimation of resource selection probability function. Journal of Wildlife Management 73:122–127.

Lele, S. R., and J. L. Keim. 2006. Weighted distributions and estimation of resource selection probability functions. Ecology 87:3021–3028.

McGarigal, K., S. A. Cushman, M. C. Neel, and E. Ene. 2002. FRAGSTATS: spatial pattern analysis program for categorical maps. Computer software, University of Massachusetts, Amherst, USA. URL: www.umass.edu/landeco/research/fragstats/fragstats.html

Minnesota Land Management Information Center. 2007. National wetlands inventory, Minnesota: metadata summary web page. URL: http://www.mngeo.state.mn.us/chouse/metadata/nwi.html
Minnesota Natural Resources Conservation Service. 2010. CRP job sheets web page. URL: http://www.mn.nrcs.usda.gov/technical/ecs/agron/crp/planform.html
MNDNR. 2001. Full GIS metadata record for GAP land cover - raster web page. URL: http://jmaps.dnr.state.mn.us/mdreporter/dp_full_record.jsp?mpid=25000015&ptid=6&fcid=6&dsid=31
Niemuth, N. D., and M. S. Boyce. 2004. Influence of landscape composition on sharp-tailed grouse lek location and attendance in Wisconsin pine barrens. Ecoscience 11:209–217.
Solberg, K. 1999. Savannas suitable for restoration and management for sharp-tailed grouse. Final Technical Report for LCMR Project M32, Minnesota Department of Natural Resources, St. Paul, USA.

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

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

[a] McGarigal et al. (2002).

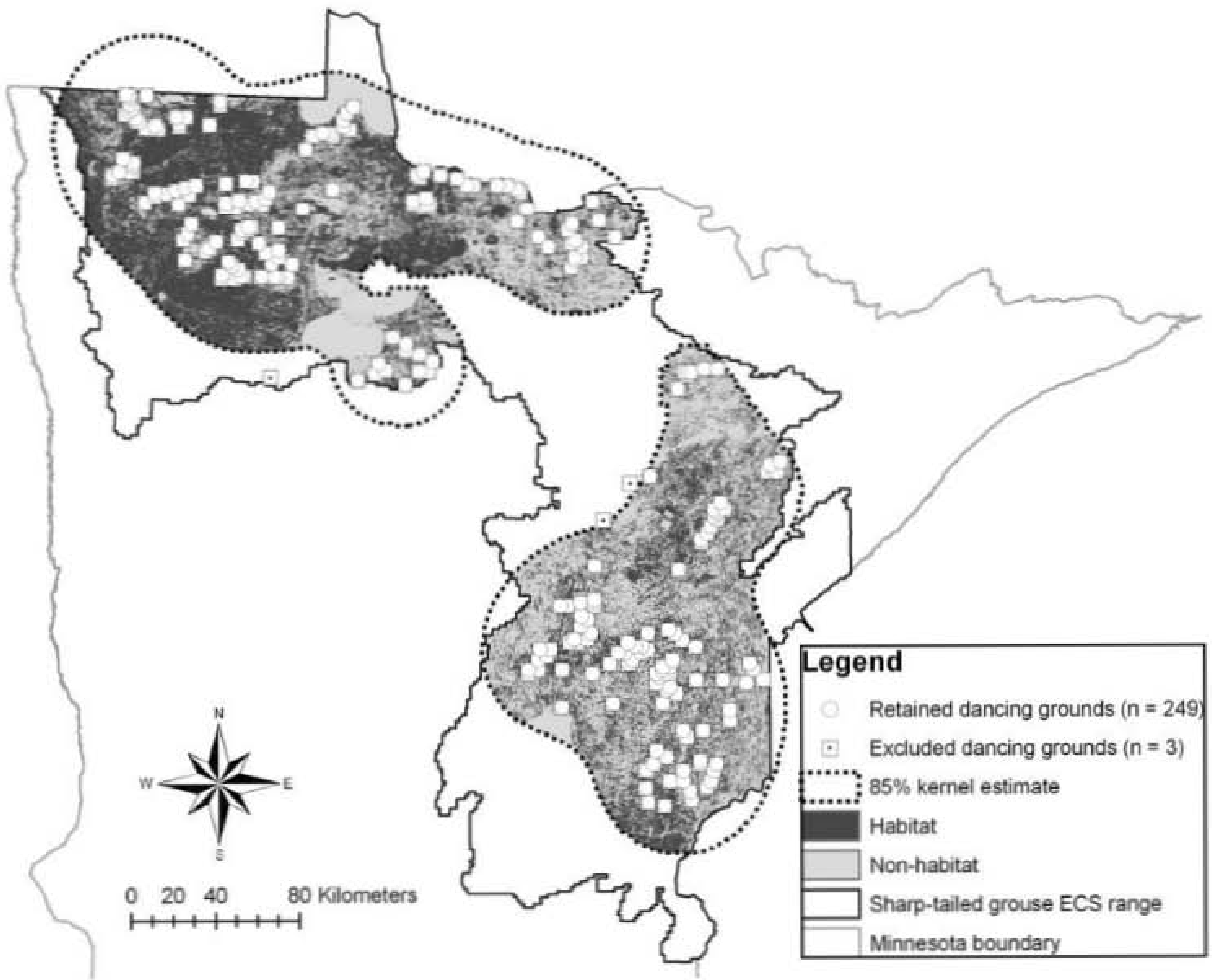


Figure 1. Map of the spatial extent of the habitat selection model for sharp-tailed grouse in northern Minnesota (shaded areas). The extent was defined as occurring within (1) occupied ECS subsections and (2) the 85% kernel estimate of space use, based upon the locations of dancing grounds that were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

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

J. Wesley Bailey

SUMMARY OF FINDINGS

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

INTRODUCTION

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

OBJECTIVES

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

STUDY AREA

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

METHODS

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

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

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

RESULTS

Initial modeling suggests connectivity varies among dancing grounds but is greatest among clusters of dancing grounds in northwest Minnesota. In east-central Minnesota, particularly in Aitkin and Carlton Counties, individual dancing grounds occur in highly connected clusters, but connectivity among individual clusters appears limited. Data analyses will include investigating how connectivity may affect dancing ground persistence and I will evaluate connectivity differences among high versus low count dancing grounds.

ACKNOWLEDGMENTS

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

LITERATURE CITED

Connelly, J. W., M. W. Gratson, and K. P. Reece. 1998. Sharp-tailed grouse (Phasianellus typanuchus). *In* A. Poole and F. Gill, editors. The Birds of North America, No. 354. The Academy of Natural Sciences, Philadelphia, Pennsylvania, USA and The American Ornithologists' Union, Washington, D. C., USA

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

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

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

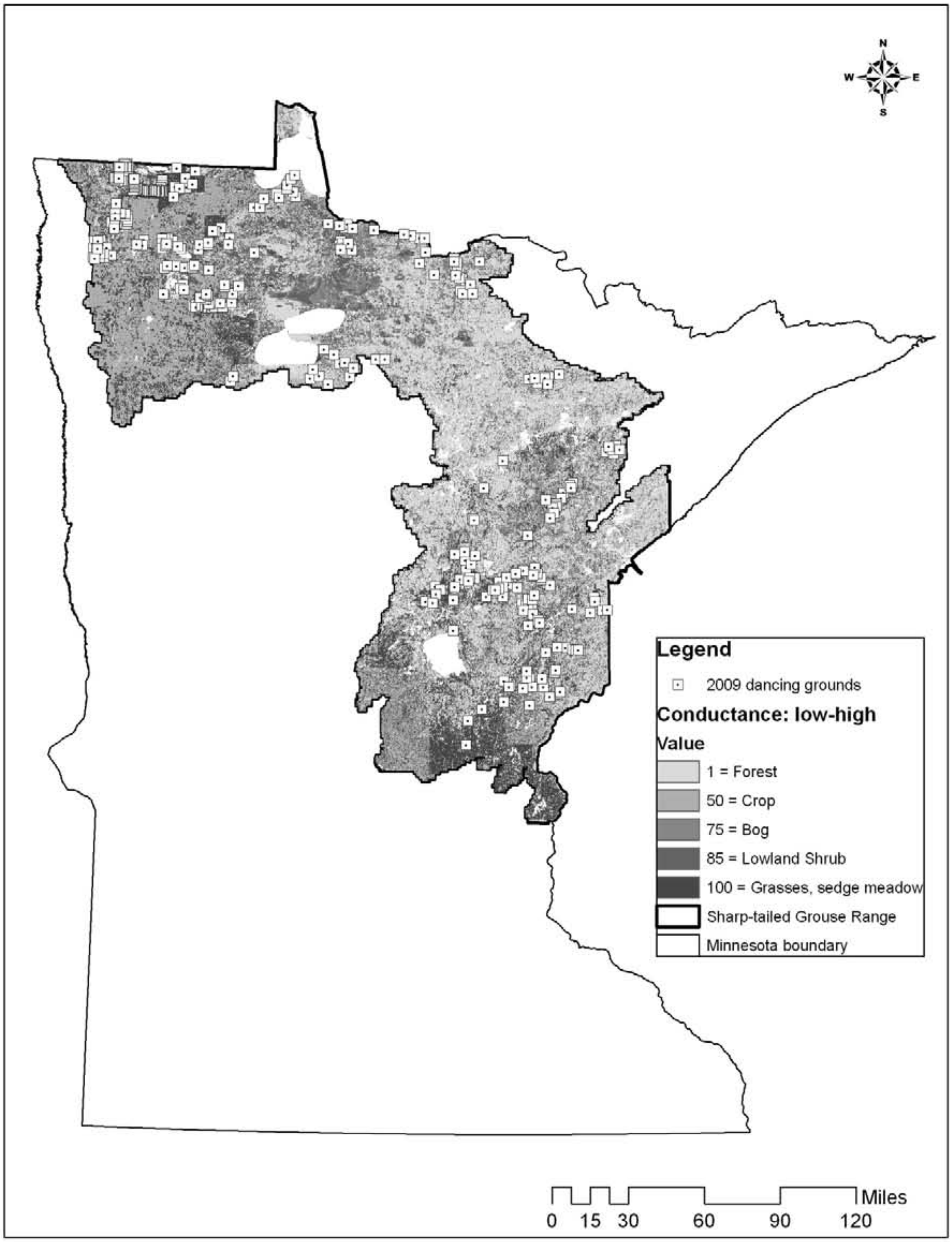


Figure 1. Sharp-tailed grouse dancing ground locations and landcover converted to conductance values to model connectivity in Minnesota, 2010.

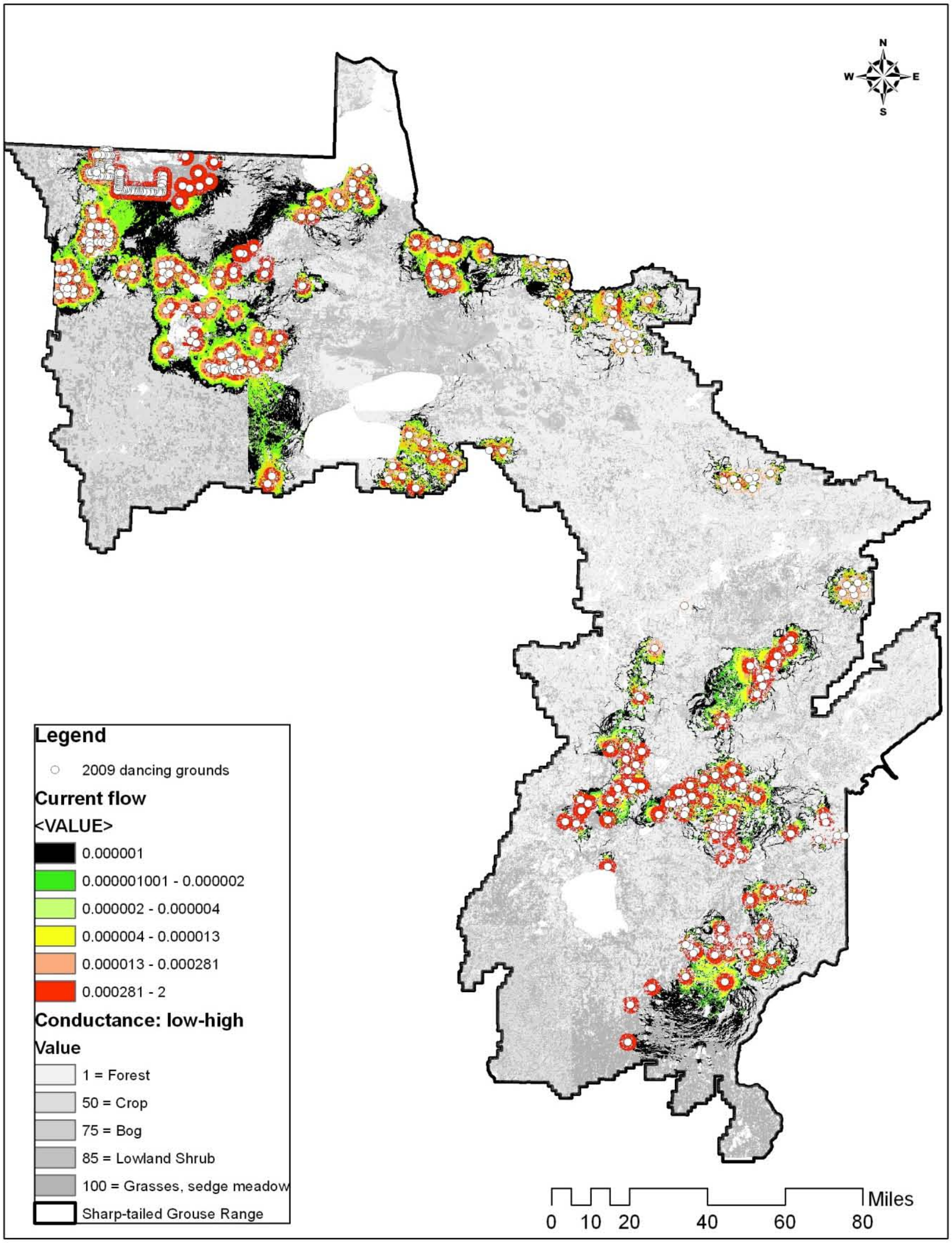


Figure 2. Preliminary output from Circuitscape modeling connectivity as current (low to high) flowing out from dancing grounds across the conductive landscape in Minnesota, 2010.

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

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

SUMMARY OF FINDINGS

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

INTRODUCTION

The goal of this long-term investigation was to assess the ecological value of conifer stands as winter thermal cover or snow shelter for white-tailed deer at the *population level*. This study was prompted directly by an increasing need of the Minnesota Department of Natural Resources' (MNDNR) wildlife managers for information regarding the habitat requirements of white-tailed deer in the forest zone of the state. Expanding our understanding of their habitat requirements and ecology during all seasons in relatively complex ecosystems impacted frequently by significant natural and human-related forces is critical to effective population management. It is also essential to the wildlife manager's ability to provide meaningful input to coordinated long-term forest management strategies and the short-term activities that immediately and dramatically alter deer habitat. Both white-tailed deer and the forests of the Great Lakes region are highly regarded for their recreational value and have notable positive impacts on local and state economies.

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

Historically, the availability of conifer stands has declined markedly relative to the increasing numbers of deer in Minnesota and elsewhere in the Great Lakes region, and this in part, has increased management's need for a better understanding of the value of this cover type to deer. The level of logging of all tree species collectively, and conifer stands specifically, has recently reached the estimated allowable harvest. Land management agencies and commercial landowners commonly restrict harvests of conifers compared to hardwoods, because of evidence at the *individual-level* indicating the seasonal value of this vegetation type to white-tailed deer and other wildlife species. However, agencies anticipate increased pressure to allow more liberal harvests of conifers in the future. Additional information is needed to assure future management responses and decisions are ecologically sound. This need has been reinforced by increasing information about the potential effects of climate change on northern forest ecosystems in Minnesota, including a shift northward of spruce-fir forests (Iverson and Prasad 2001, Hansen et al. 2003), as well as a pronounced decline in lowland coniferous forests and the potential benefits they afford as snow shelter and thermal cover. According to MNDNR (2008), "wildlife associated with coniferous forests may be under the greatest threat of extirpation from Minnesota due to climate change."

OBJECTIVES

Expecting that environmental variation, particularly in winter weather conditions, would have biologically significant influences on various aspects of deer ecology, we knew a long-term study would enhance our ability to examine and understand these influences and the importance of conifer cover as a habitat component (DelGiudice and Riggs 1996). We hypothesized that winter severity and conifer availability affect the use of moderately dense (40-69% canopy closure [Class B]) and dense (≥70% canopy closure [Class C]) conifer stands on winter range by female white-tailed deer as thermal cover or snow shelter, deer movements (i.e., migration) and distribution. Further, we hypothesized that nutrition is likely the mechanistic thread between this environmental variation and the population performance (survival and reproduction) of deer. Relative to varying winter severities, the objectives of the comprehensive approach of this study have been to:

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

STUDY DESIGN AND PROGRESS

This study (1991-2005) included 4 winter range study sites (Willow, Inguadona, Shingle Mill, and Dirty Nose), located in the Grand Rapids-Remer-Longville area of north-central Minnesota; they range from 13 to 23.6 km^2 (5-9.1 mi^2) in area (Table 1). Conifer stands on the sites primarily included balsam fir (*Abies balsamea*), northern white cedar (*Thuja occidentalis*), black spruce (*Picea mariana*), and jack and red pine (*Pinus banksiana* and *P. resinosa*). Common browse species were beaked hazel (*Corylus cornuta*), mountain maple (*Acer spicatum*), sugar maple (*A. saccharum*), red-osier dogwood (*Cornus stolonifera*), and ironwood (*Ostrya virginiana*). The study began with the Willow and Inguadona sites during winter 1990-1991. The Shingle Mill and Dirty Nose sites were included beginning in winter 1992-1993. We applied an experimental treatment (timber harvest) to reduce moderately dense and dense conifer stands (good and optimum thermal cover/snow shelter, respectively) to what is considered poor cover (< 40% canopy closure [Class A]) on the Inguadona and Shingle Mill sites midway through the study; limited, unplanned decreases of conifer cover occurred on all 4 sites over the 15-year period (Figure 1). Mean area of conifer canopy closure classes A, B, and C differed markedly among the 4 sites (Table 1). During the 15-year study, availability of dense conifer cover was greatest on the Willow (23.2%) and Shingle Mill (16.5%) sites (Figure 1). The most pronounced reduction (percentage) in dense conifer cover as the study progressed occurred on the Inguadona site (Figure 1). The temporal variations in conifer cover and differences among sites are proving to be of notable value to many of our analyses.

Data collected on all 4 study sites included the following: (1) descriptive quantification of deer habitat by color infrared air photointerpretation, digitizing, and application of a geographic information system (GIS, ArcMap 9.3.1) for temporal and spatial analyses; (2) monitoring of ambient temperature, wind velocity, snow depth, and snow penetration (index of density) in various habitat types (e.g., openings versus dense conifer cover) by automated weather data-collecting systems, minimum/maximum thermometers, and conventional hand-held measurements; (3) deer capture, chemical immobilization, and handling data (e.g., rectal temperature, response times to immobilizing chemicals); (4) age determination by last incisor extraction and cementum annuli analysis; (5) data generated by laboratory analyses of physiological samples of all captured and recaptured female deer, including complete blood cell counts (CBCs), serum profiles of approximately 20 constituents, (e.g., reproductive and metabolic hormones, chemistries), urine chemistry profiles, and partial and complete body composition determination by isotope-dilution and ultrasonography; (6) morphological measurements; (7) physiological assessment of winter nutritional restriction by sequential collection and chemical analysis of snow-urine; (8) seasonal migrations and other movements via VHF and GPS radiocollars; (9) habitat use; (10) annual and seasonal cause-specific mortality; (11) age-specific survival rates; (12) pregnancy determination; (13) winter food habits; and (14) movements, territory size, survival, and cause-specific mortality of radiocollared wolves. See DelGiudice and Sampson (2008), other previous issues of the Minnesota Department of Natural Resources' annual "Summaries of Wildlife Research Findings," and associated publication lists for further details of this study.

Winter Severity, Use of Conifer Cover, Nutrition, and Survival of White-Tailed Deer

Weather is one of the strongest environmental forces impacting wildlife populations. Our 15-year study period allowed us to capture a wide breadth of variation in the severity of winter weather conditions, including 2 back-to-back historically severe winters (1995-1996, 1996-1997), followed by 3 consecutive, unprecedented mild winters in more than 100 years of weather data collection (P. Boulay, Minnesota State Climate Office, personal communication), as well as many of mild to average conditions. The MNDNR's maximum winter severity index (WSI, calculated by accumulating 1 point for each day with an ambient temperature $\leq$ -17.7^{o}C and 1 point for each day with snow cover $\geq$ 38 cm during November-May) ranged from 42 to 195. This long-term variation in winter weather provided a valuable broader context for data examination, interpretation, and understanding than would have been possible in a typical short-term study.

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

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

Our 14-year monitoring of winter food habits of white-tailed deer on the 4 sites showed that, overall, relative to the number of plant species, their diet was highly diverse; however, beaked hazel, mountain maple, and red-osier dogwood accounted for 81.9 and 89.3% of their diet during mild-average and severe winters, respectively (Table 3). Most typically, mountain maple and red-osier dogwood were selected (proportion of overall use was >overall proportional availability) for by deer, whereas, beaked hazel, although co-dominant in their diet, was used in proportion to availability (Table 3). The category "other species" consisted of about 24 browse species, and on average accounted for 28.8-35.4% and 17.7-33.8% of their diet during mild-average winters and severe winters, respectively, on the 4 sties (Table 3). Diet diversity is critical to the deer's ability to maintain its nutritional status during winter (Verme and Ullrey 1972). Use of the "other species" category decreased ($P \leq 0.05$) by 48% and 42% during severe winters compared to mild-average winters on the Willow and Dirty Nose sites, respectively, suggesting that the diversity of their diet decreased during severe winters. At Willow, where deer were most likely to be in dense conifer cover during severe winters of deep snow, mean proportional use of mountain maple also declined (28%, $P \leq 0.05$), as did use of red-osier dogwood (47%), although not significantly so due to greater variability. Deer made significantly ($P \leq 0.05$) greater use (up to 71%) of beaked hazel at all sites during severe winters, except at Willow, where the increase was less pronounced. Deer typically made relatively low use of paper birch (*Betula papyrifera*) and trembling aspen (*Populus tremuloides*) at all 4 sites, but in proportion to their availabilities.

Winter nutritional restriction or deprivation of white-tailed deer and other northern ungulates can be assessed by sequential collection and chemical analysis of fresh urine voided in snow (DelGiudice et al. 1988, 1989, 1997, 2001; Ditchkoff 1994; and others). Overall, we documented significant (P = 0.057 and P = 0.013) relationships between maximum WSIs and percent of snow-urine samples collected during each winter with urea nitrogen:creatinine (UN:C) ratios indicative of severe nutritional restriction (UN:C ≥3.5 mg:mg, Figure 4) and between the latter and percent winter mortality of radiocollared deer (Figure 5). Interestingly, we noted significant ($P \leq 0.05$) differences between mean UN:C ratios during mild-average versus severe winters on all 4 sites when we included winters 1992-1993 (WSI = 124) and 1993-1994 (WSI = 126) in the mild-average category (Table 4), as opposed to including these winters in the severe winter category. Additionally, the differences in the percentage of samples collected that were indicative of severe nutritional restriction was more apparent when winters 1992-94 and 1993-94 were categorized as mild-average (Table 4). So from the perspective of the deer's physiological response to winter conditions during these 2 winters, we would consider WSIs of 124-126 to be reflective of conditions less than severe. Serious nutritional restriction was most common (indicated by UN:C ratios of 18-20% of snow-urine samples, Table 4) during severe winters at the Willow and Shingle Mill sites where dense conifer cover was most available (Figure 1) and where deer were most likely to be using this cover (Figure 2).

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

ACKNOWLEDGMENTS

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

LITERATURE CITED

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.
DelGiuduce, 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., L. D. Mech, and U. S. Seal. 1988. Comparison of chemical analyses of deer bladder urine and urine collected from snow. Wildlife Society Bulletin 16:324-326.
DelGiudice, G. D., L. D. Mech, and U. S. Seal. 1989. Physiological assessment of deer populations by analysis of urine in snow. Journal of Wildlife Management 53:284-291.
DelGiudice, G. D., R. A. Moen, F. J. Singer, and M. R. Riggs. 2001. Winter nutritional restriction and simulated body condition of Yellowstone elk and bison before and after the fires of 1988. Wildlife Monographs No. 147. 60pp.
DelGiudice, G. D., R. O. Peterson, and W. M. Samuel. 1997. Trends of winter nutritional restriction, ticks, and numbers of moose on Isle Royale. Journal of Wildlife Management 61:895-903.
DelGiudice, G. D., and M. R. Riggs. 1996. Long-term research of the white-tailed deer-conifer thermal cover relationship: aligning expectations with reality. Transactions of the North American Wildlife and Natural Resources Conference 61:416-428.
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., and B. A. Sampson. 2008. Assessing the relationship of conifer thermal cover to winter distribution, movements, and survival of female white-tailed deer in north central Minnesota. Pages 389-395 *in* M. W. DonCarlos, R. O. Kimmel, J. S. Lawrence, and M. S. Lenarz, editors. Summaries of wildlife research findings, 2007. Minnesota Department of Natural Resources, St. Paul, Minnesota.
DelGiudice, G. D., M. S. Lenarz, and M. Carstensen Powell. 2007. Age-specific fertility and fecundity in northern free-ranging white-tailed deer: evidence for reproductive senescence? Journal of Mammalogy 88:427-435.
Ditchkoff, S. S. 1994. Nutritional status and food availability of white-tailed deer during winter in Maine. M. S. Thesis. University of Maine, Orono, Maine, USA.
Fieberg, J., D. W. Kuehn, and G. D. DelGiudice. 2008. Understanding variation in autumn migration of northern white-tailed deer by long-term study. Journal of Mammalogy 89:1529-1539.
Hansen, L. J., J. L. Biringer, and J. R. Hoffman. 2003. Buying time: a user's annual for building resistance and resilience to climate change in natural systems. World Wildlife Fund, Berlin, Germany.
Iverson, L. R., and A. M. Prasad. 2001. Potential changes in tree species richness and forest community types following climate change. Ecosystems 4:186-199.
MNDNR. 2008. Climate change: preliminary assessment for the Section of Wildlife of the Minnesota Department of Natural Resources. Prepared by the Wildlife Climate Change Working Group, Division of Fish and Wildlife, Minnesota Department of Natural Resources, St. Paul.
Verme, L. J., and D. E. Ullrey. 1972. Feeding and nutrition of deer. Pages 275-291 in D. C. Church, editor. Digestive physiology and nutrition of ruminants. Volume 3, Practical nutrition of ruminants. Oregon State University, Corvallis.

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

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

Table 2. Nearest distance of radiocollared, female white-tailed deer to conifer cover with canopy closures of at least 70% (“C”) or 41-69% (“B”) on 4 study sites during mild to average winters versus severe winters, north-central Minnesota, winters 1993-1994 to 2004-2005.[1]

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

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

Table 3. Browse availability and use by white-tailed deer on 4 study sites during mild-average versus severe winters, north-central Minnesota, winters 1991-1992 to 2004-2005.[1]

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

Table 3. Continued.

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

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

Table 4. Mean urea nitrogen:creatinine (UN:C) ratios in urine recently voided (≤ 72 hr) in snow by white-tailed deer and percent of samples indicative of severe nutritional restriction (UN:C ≥3.5 mg:mg) on 4 study sites during mild-average versus severe winters, north-central Minnesota, winters 1992-1993 to 2004-2005.[1]

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

[1]Winters of mild-average severity (winter severity indices [WSI] ≤ 126) included winters 1994-1995, 1998-1999, 1999-2000, and 2001-2002 to 2004-2005, and severe winters (WSIs ≥153) included winters 1993-1994, 1995-1996, 1996-1997, and 2000-2001.
[2]These percentages were recalculated with winters 1992-1993 and 1993-1994 included as severe winters, rather than as mild-average winters.

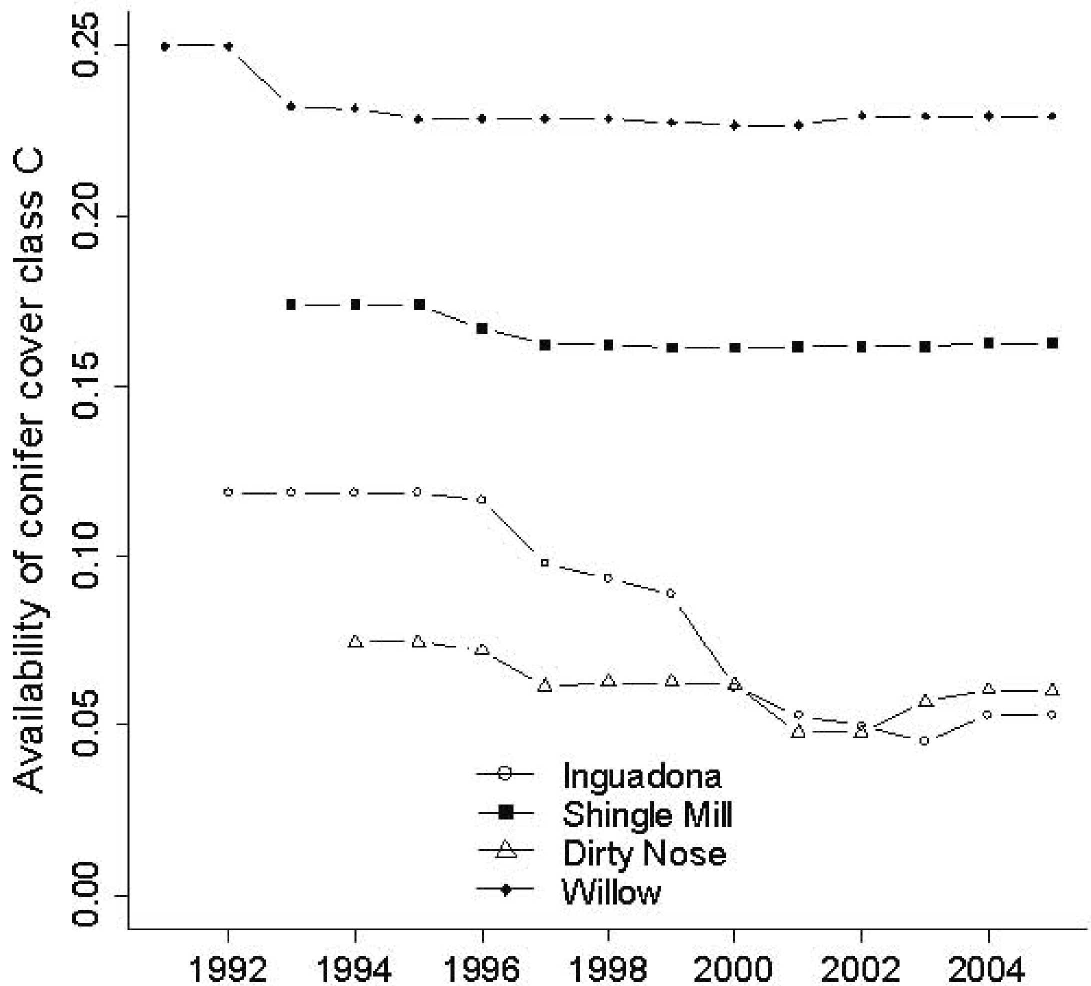


Figure 1. Changes in the availability of conifer cover with canopy closure of at least 70 percent within the 4 study sites of the white-tailed deer/winter cover study, north-central Minnesota, winters 1990-1991 to 2004-2005.

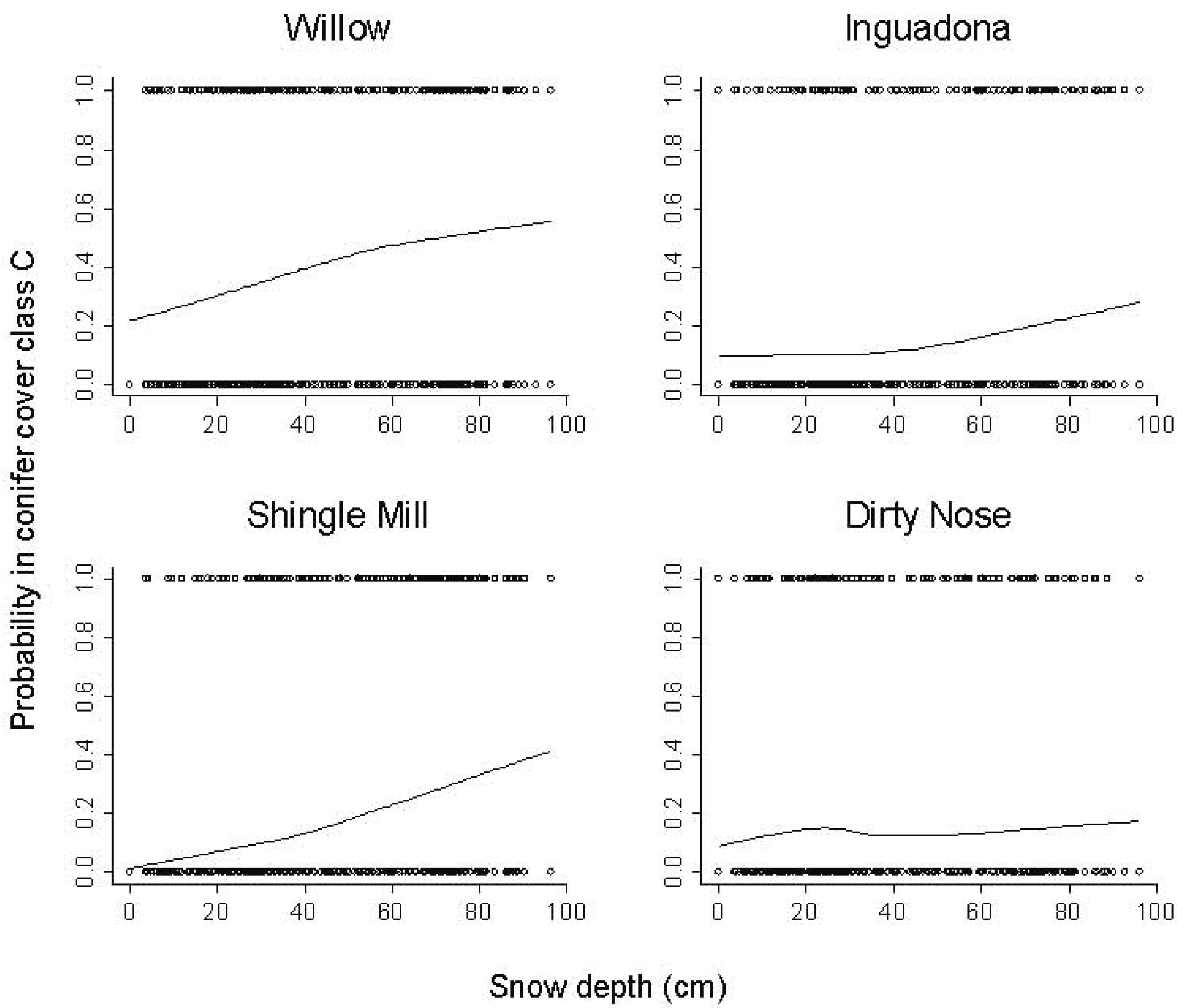


Figure 2. Probability of radiocollared deer being in conifer cover with canopy closures of at least 70% as a function of snow depth on the 4 study sites, north-central Minnesota, winters 1993-1994 to 2004-2005. (Small circles at the bottom and top of graphs represent the density of data collected.)

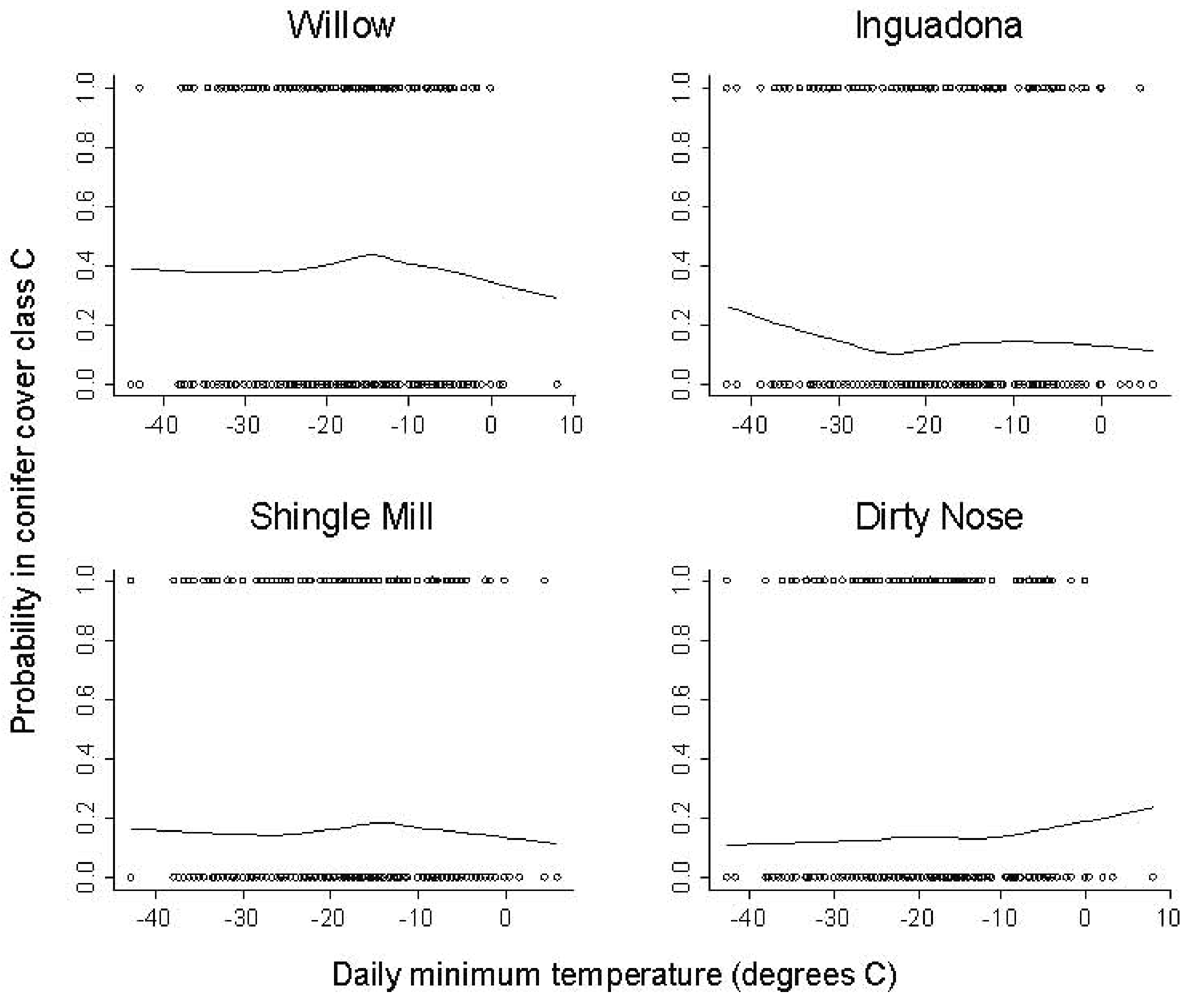


Figure 3. Probability of radiocollared deer being in conifer cover with canopy closures of at least 70% as a function of daily minimum temperature on the 4 study sites, north-central Minnesota, winters 1993-1994 to 2004-2005. (Small circles at the bottom and top of graphs represent the density of data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

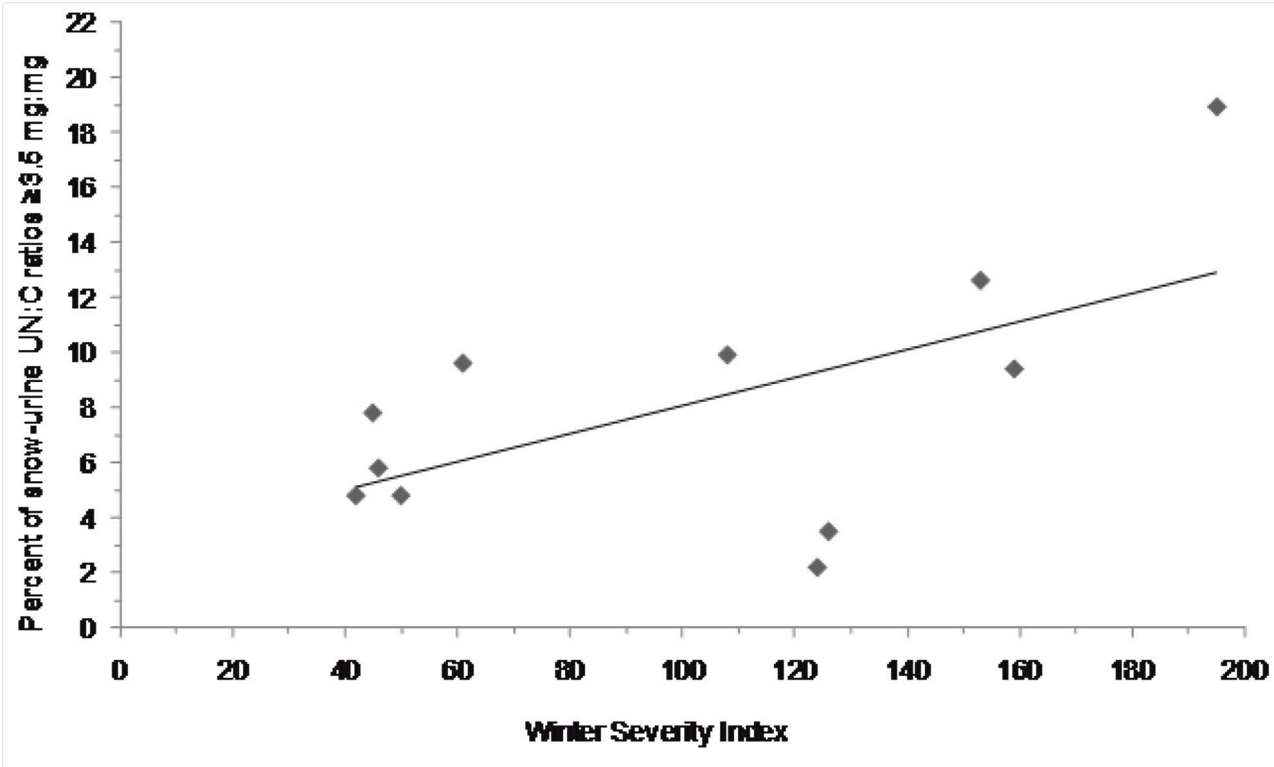


Figure 4. Relationship (r^2 = 0.35, y = 2.958 + 0.051*x*, *P* = 0.057) of the annual maximum winter severity index (see text for definition) to the percent of urine samples in snow (snow-urine) of white-tailed deer with urea nitrogen:creatinine (UN:C) ratios indicative of severe nutritional restriction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

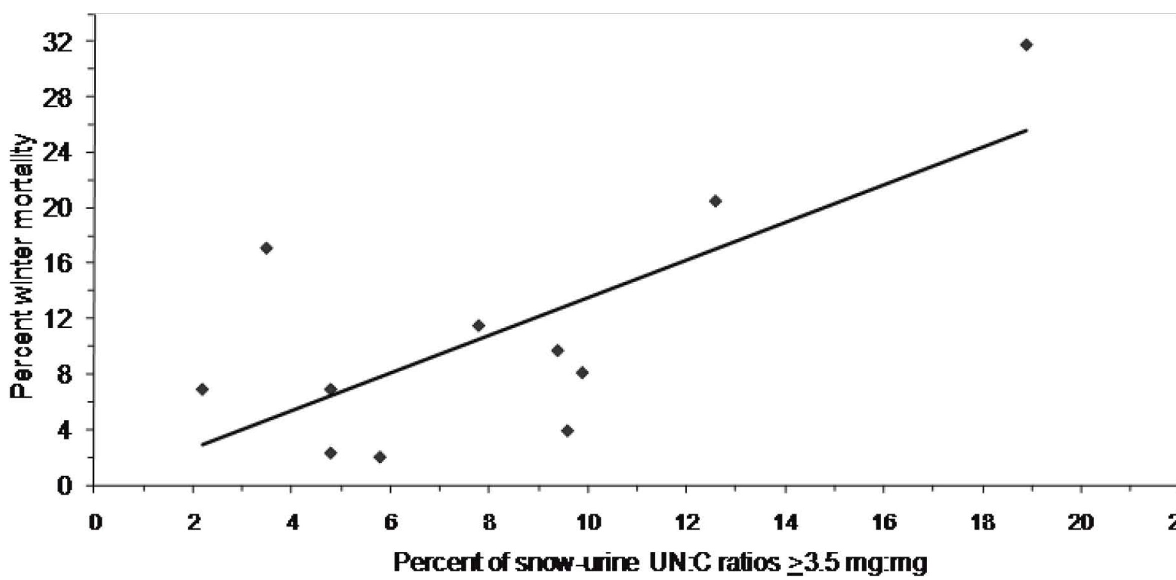


Figure 5. Relationship (r^2 = 0.52, y = 3.942 + 0.381*x*, *P* = 0. 013) of the annual percent of urine samples in snow (snow-urine) of white-tailed deer with urea nitrogen:creatinine (UN:C) ratios indicative of severe nutritional restriction (≥3.5 mg:mg) to percent winter mortality, all study 4 study sites (pooled), north-central Minnesota, winters 1992-1993 to 1998-1999, 2000-2001, 2001-2002, 2003-2004, and 2004-2005.

Wetlands Wildlife Populations and Research Group
102 - 23rd Street NE
Bemidji, Minnesota 56601
(218) 308-2282

COMPARISON OF NATIVE GRASSLAND MANAGEMENT TREATMENTS TO SPRING PRESCRIBED BURNS

David Rave, Kevin Kotts, and John Fieberg

SUMMARY OF FINDINGS

We conducted a pilot study in 2008 to measure the response of restored native grasslands to: (1) grazing; (2) fall biomass harvest; and (3) spring prescribed burning. Among field variability was substantial in the pilot study, suggesting the need to control for this variability when making treatment comparisons. Therefore, in 2009, we dropped the grazing element of the study, and added 6 additional sites using a split plot design, in which matched subplots were biomass harvested in fall 2008, or burned in spring 2009. Fields were located on Wildlife Management Areas (WMAs) or Waterfowl Production Areas (WPAs) in Working Lands Initiative Focus Areas of Chippewa, Grant, Kandiyohi, Lac Qui Parle, Renville, Stevens, and Swift counties. We conducted visual obstruction measurements, Daubenmire frame analysis, and we measured litter depth and vegetation height in all study fields. We also examined temporary and seasonal wetlands in bioharvested fields and recorded wetland type, and waterfowl presence. Biomass harvested and burned subplots appeared similar in most vegetative characteristics in both 2008 and 2009. In 2010, we intend to survey vegetation in additional plots in which biomass harvest/burn treatments are applied, and using these additional data, will determine whether to continue the project in 2011.

INTRODUCTION

Minnesota's Department of Natural Resources' (MNDNR) Draft Grassland Biomass/Bioenergy Harvest on WMAs and Aquatic Management Areas (AMAs) management document states, "Grassland biomass harvest from WMAs and AMAs shall be in concert with fish and wildlife habitat management activities, consistent with the habitat or wildlife species management goals and habitat management objectives for each individual WMA/AMA." Further, Sample and Mossman (1997) found that differences in habitat structure are likely more important to bird communities than differences in vegetative species composition. They recommend that the following features of grassland habitat are important to grassland nesting birds: vegetation height and density, height and cover of woody vegetation, litter depth and cover, standing residual (dead) and live herbaceous cover, and ratio of grass vs. forb cover. However, the response of native grassland stands on WMAs and AMAs to grassland biomass harvest is unknown. We conducted this study with the following objectives:

- to determine vegetative response to biomass harvest;
- to determine whether vegetative response to fall biomass harvest is similar to vegetative response to spring controlled burning; and
- to determine whether fall biomass harvest can be used by Wildlife Managers to maintain restored prairie grasslands.

STUDY AREA

The study was conducted in Chippewa, Grant, Kandiyohi, Lac Qui Parle, Renville, Stevens, and Swift counties, within the prairie portion of Minnesota (Figure 1), and was targeted at Working Lands Initiative (MNDNR unpublished brochure http://files.dnr.state.mn.us/assistance/backyard/privatelandsprogram/working-lands-ini.pdf) Focus Areas. Fields sampled were all located on state managed WMAs or federally managed WPAs. Sites in 2009 consisted of 9 fields with bioharvest and burn subplots, and 6 sites with only a bioharvest subplot. Spring burns on these latter 6 fields were not accomplished.

METHODS

We compared the response of restored native grasslands to fall biomass harvest (hayed) and spring prescribed burning (control) using paired subplots and a split-plot design (Steel et al. 1997). Visual obstruction measurements (VOMs, Robel et al. 1970) were taken every 2 weeks from early June through mid-August in hayed and control subplots of each field following methods described by Zicus et al. (2006). Three VOM sample stations were established at the 3 quarter points along the longest straight-line transect across each subplot within a field (hereafter the VOM transect). GIS locations were permanently marked with stakes to define starting and sampling points along the VOM transect. Each station had 4 sampling points located 20 m north, east, south, and west of a starting point. At each field sampling point, vegetation height and density was measured in each cardinal direction. This provided 48 VOMs for each treatment from each field on a given date.

A Daubenmire square (Daubenmire 1959) was used to determine coverage by various species across hayed and burned subplots. We sampled at 10 locations along the VOM transect in all subplots of each field every 2 weeks. The 1m^2 Daubenmire frame was placed on the ground approximately 10 meters from the VOM transect every tenth of the entire transect distance determined using a GPS. Each plant species (and % coverage within the frame) that comprised $\geq$ 10% of the total number of individual plants within the frame was recorded.

Litter depth (nearest 1mm) and vegetation height (nearest 0.5 dm) were also measured at 10 locations, each 1 tenth of the entire transect distance as determined using a GPS, on the VOM transect in all subplots of each field every 2 weeks. While walking the VOM transect, all exotic and woody species present were recorded. The amount of these species in each field will be estimated using distance sampling (Buckland et al. 2004).

We also examined seasonal and temporary wetlands in mid-April that had vegetation removed, primarily cattails, during biomass harvest the previous fall. For each wetland, we recorded wetland type (Stewart and Kantrud 1971), waterfowl numbers, and waterfowl pair status.

RESULTS

Vegetative characteristics were largely similar in hayed and burned subplots (Figures 2-6). The most notable exception was Klason in 2008. At this site (in 2008), vegetation was taller (with larger VOM readings), litter depth was greater, and a higher number of species were located in the hayed treatment subplot than the burned subplot; however, these differences were largely absent the next year. In 2009, litter depths again varied in subplots hayed in fall 2008 and burned in spring 2009 (Beaver Falls WMA, Danvers WMA, Lac Qui Parle WMA, and Towner WMA), whereas other vegetative characteristics were similar between treatment subplots.

We examined 12 seasonal and temporary wetlands in mid-April that had been at least partially harvested during the biomass treatment in fall 2007. Cattail growth in summer of 2008 filled in these wetlands, and there were no waterfowl pairs using the wetlands in spring 2009.

DISCUSSION

Recently, the cost of fossil fuels has increased as their supply tightened. Alternative sources of energy are being sought. Wind, solar, and other renewable energy sources are being developed. One potential source is biomass energy derived from agricultural or other cellulose residues. Based on estimates from 2005, there is approximately 194 million tons of biomass available each year from the agricultural sector (Perlack et al. 2005). However, the United States Department of Agriculture projects that to replace 30% of petroleum use by 2030 will require over 1 billion tons of biomass. To acquire this amount of biomass, new sources of biomass will need to be developed. One possible source of biomass is native grass. However, the effects of biomass harvest on vegetation in native grass fields and the birds that nest in those fields are unknown.

The Minnesota Department of Natural Resources acquires and manages Wildlife Management Areas primarily to establish and maintain optimal population levels of wildlife while maintaining ecological diversity; maintaining or restoring natural communities and ecological processes; and maintaining or enhancing populations of native species (including uncommon species and state- and federally-listed species; The Draft Grassland Biomass/Bioenergy Harvest on WMAs & AMAs directive, unpublished MNDNR publication). Prior to settlement and implementation of agriculture, natural disturbance in the form of fire and grazing maintained native grassland diversity and productivity (Anderson 1990). Wildlife managers have traditionally used spring prescribed burns to simulate these natural disturbances (K. Kotts, personal communication). However, there are a variety of management options available to wildlife managers to create disturbances in native grass stands. These options are not typically the first choice of managers; likely because there is little known about the response of native grass stands to these treatments. Our study is designed to compare the vegetative response of a biomass harvest for disturbing native grass stands, and compare the response to that from a spring controlled burn.

After 2 field seasons, there appears to be little difference in vegetation characteristics between bioharvested and burned subplots. We will monitor all subplots again in 2010 to look for any vegetative differences among subplots that may occur with time. Further, the removal of wetland vegetation in the fall is a promising way to open choked wetlands, making them available to waterbirds such as dabbling ducks, geese, swans, and shorebirds. Fall wetland conditions play an important role in determining how successful this technique will be. Wetlands must be fairly dry when the haying occurs to allow equipment to harvest vegetation within the wetland basin. Basins that were harvested in 2007 contained open water areas in spring 2008, and were utilized by migrating and nesting waterfowl. However, cattail growth in summer of 2008 was sufficient enough to eliminate most of the open water in these basins, and they were not utilized by waterfowl in spring 2009.

ACKNOWLEDGEMENTS

Funding for this study was provided by a Working Lands Initiative grant. M. Tranel and K. Haroldson helped develop the original study design. L. Dahlke, J. Miller, R. Olsen, J. Strege, C. Vacek, S. Vacek, K. Varland, and J. Zajac helped with study logistics. A. Bochow, H. Curtis, J. Gregory, and B. Stenberg were interns on this project, and collected most of the field data. The University of Minnesota at Morris and Minnesota Alfalfa Producers harvested grass from our treatment fields.

LITERATURE CITED

Anderson, R. C. 1990. The historic role of fire in the North American grassland. Pages 8-18 *in* Fire in North American Tallgrass Prairies (S. L. Collins and L. L. Wallace, Eds). University of Oklahoma Press, Norman.

Buckland, S. T., D. R. Anderson, K. P. Burnham, J. L. Laake, D. L. Borchers, and L. Thomas. 2004. Introduction to distance sampling: estimating abundance of biological populations. Oxford University Press 416pp.

Daubenmire, R.F. 1959. A canopy-cover method of vegetational analysis. Northwest Science, 33:43-46.

Perlack, R. D., L. L. Wright, A. F. Turhollow, R. L. Graham, B. J. Stokes, D. C. Erbach. 2005. Biomass as feedstock for bioenergy and Bioproducts industry. The technical feasibility of a billion ton annual supply. US Department of Energy, Document Number: DOE/GO-102995-21 35.

Robel, R. J., J. N. Briggs, A. D. Dayton, and L. C. Hulbert. 1970. Relationships between visual obstruction measurements and weight of grassland vegetation. Journal of Range Management 23:295-297.

Sample, D. W., and M. J. Mossman. 1997. Managing Habitat for Grassland Birds. A Guide for Wisconsin. Department of Natural Resources, Madison, Wisconsin, 154pp.

Steel RGD, Torrie JH, and Dickey D. 1997. Principles and procedures of statistics: a Biometrical Approach, 3rd edition. McGraw-Hill, Boston, MA, USA.

Stewart, R. E., and H. A. Kantrud. 1971. Classification of natural ponds and lakes in the glaciated prairie region. U.S. Fish and Wildlife Service Resource Publication 92, 57p.

Zicus, M. C., D. P. Rave, A. Das, M. R. Riggs, and M. L. Buitenwerf. 2006. Influence of land use on mallard nest-structure occupancy. Journal of Wildlife Management, 70(5): 1325-1333.

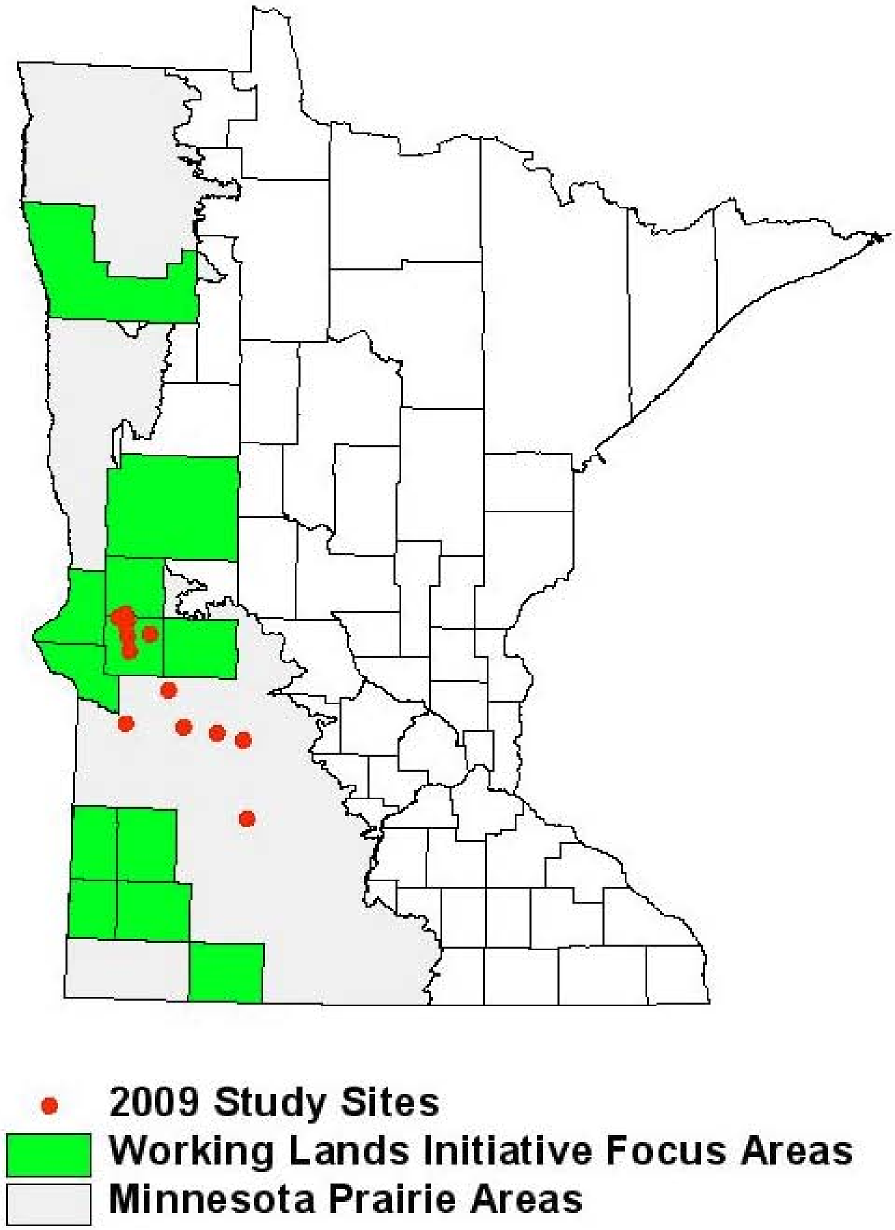


Figure 1. Minnesota counties showing prairie areas and Working Lands Initiative focus areas, 2009.

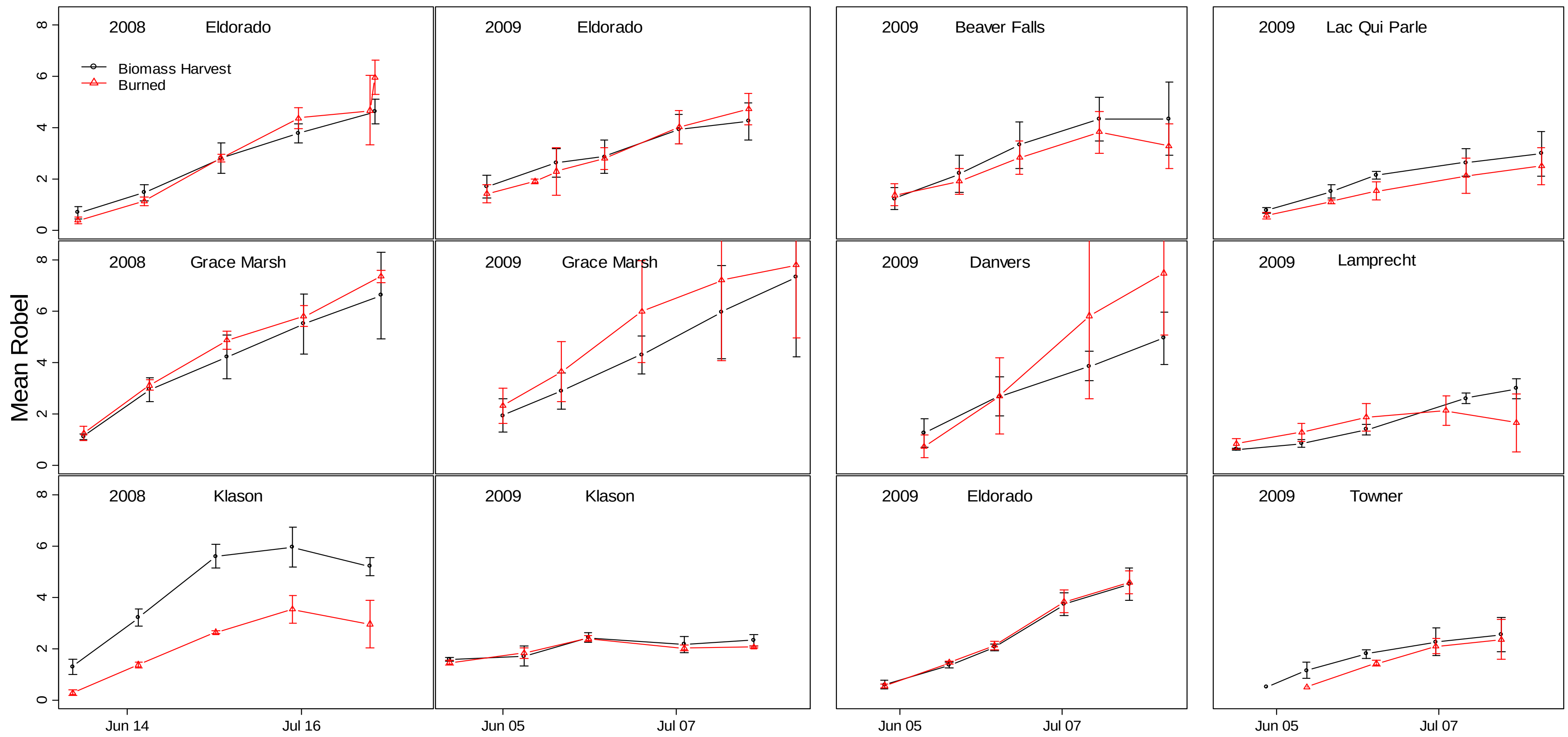


Figure 2. Comparison of mean Robel measurements (dm) and 95% confidence intervals between 2 treatment subplots (a fall biomass harvest and a prescribed burn the following spring) within the same restored native grass field on 3 State Wildlife Management Areas in west-central Minnesota, in both summer 2008 and summer 2009 (leftmost two columns), and on 5 State Wildlife Management Areas and 1 Federal Waterfowl Production area in west-central Minnesota, in only summer 2009 (rightmost two columns).

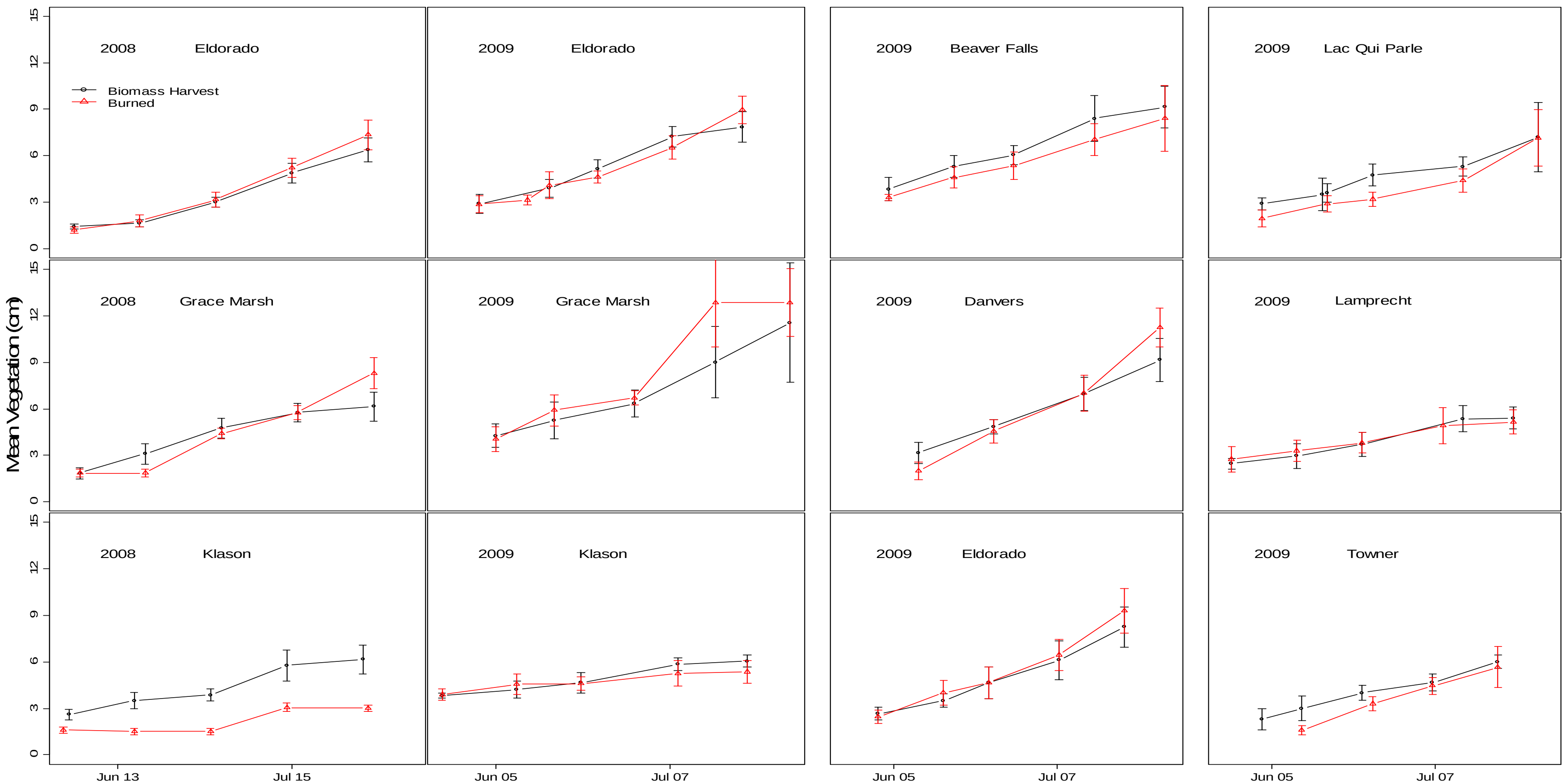


Figure 3. Comparison of mean vegetation height (dm) and 95% confidence intervals between 2 treatment subplots (a fall biomass harvest and a prescribed burn the following spring) within the same restored native grass field on 3 State Wildlife Management Areas in west-central Minnesota, in both summer 2008 and summer 2009 (leftmost two columns), and on 5 State Wildlife Management Areas and 1 Federal Waterfowl Production area in west-central Minnesota, in only summer 2009 (rightmost two columns).

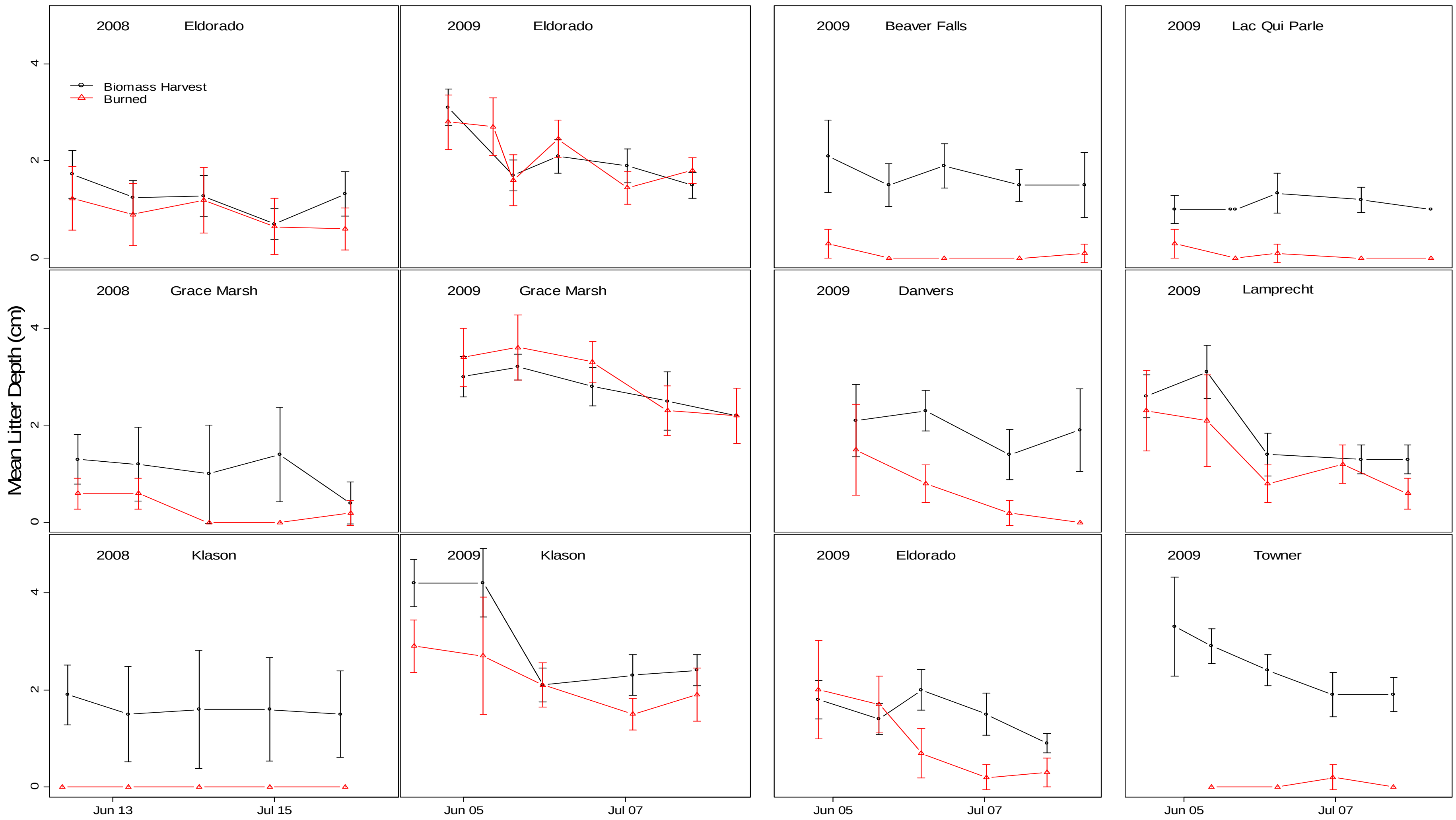


Figure 4. Comparison of mean litter depth (dm) and 95% confidence intervals between 2 treatment subplots (a fall biomass harvest and a prescribed burn the following spring) within the same restored native grass field on 3 State Wildlife Management Areas in west-central Minnesota, in both summer 2008 and summer 2009 (leftmost two columns), and on 5 State Wildlife Management Areas and 1 Federal Waterfowl Production area in west-central Minnesota, in only summer 2009 (rightmost two columns).

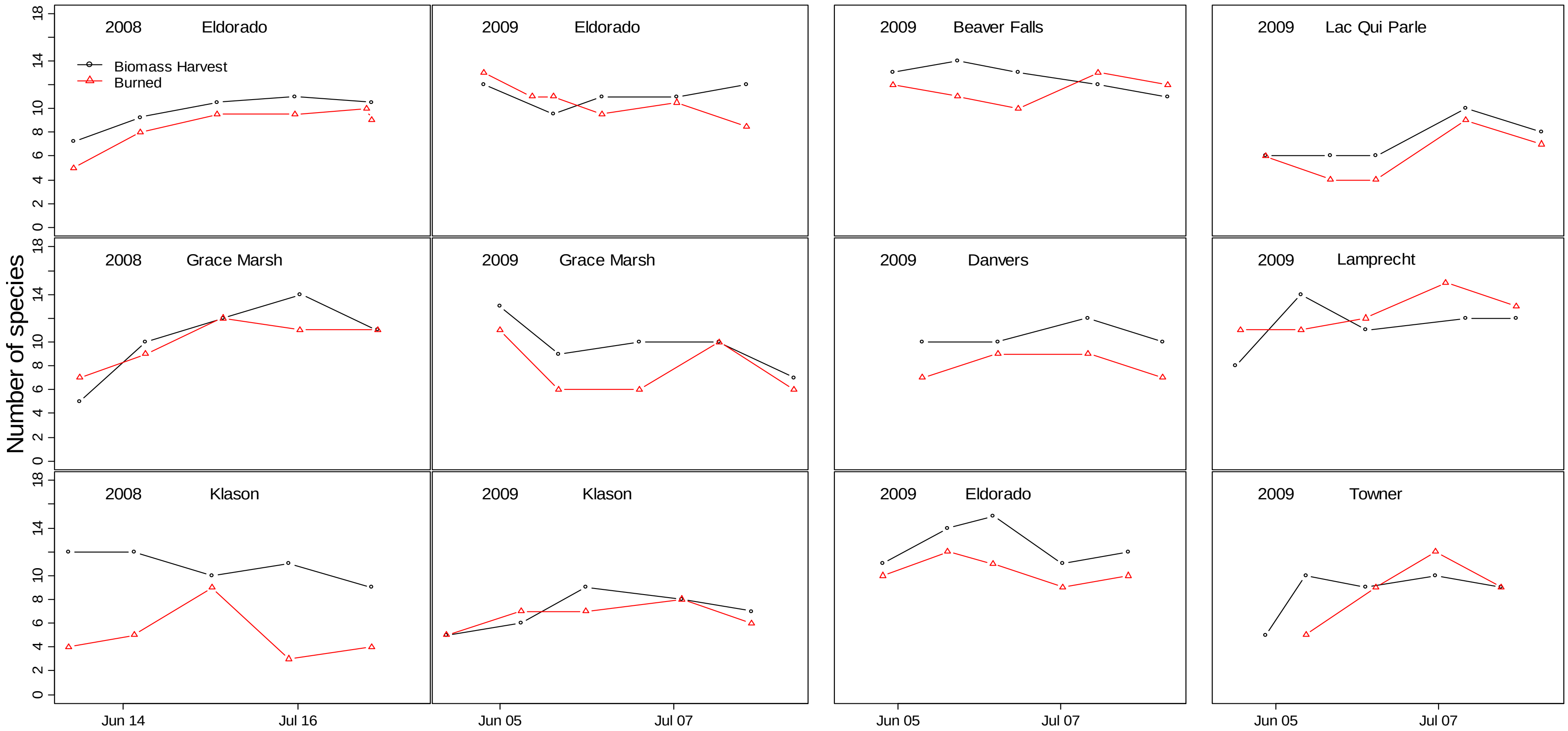


Figure 5. Comparison of mean number of plant species per transect between 2 treatment subplots (a fall biomass harvest and a prescribed burn the following spring) within the same restored native grass field on 3 State Wildlife Management Areas in west-central Minnesota, in both summer 2008 and summer 2009 (leftmost two columns), and on 5 State Wildlife Management Areas and 1 Federal Waterfowl Production area in west-central Minnesota, in only summer 2009 (rightmost two columns).

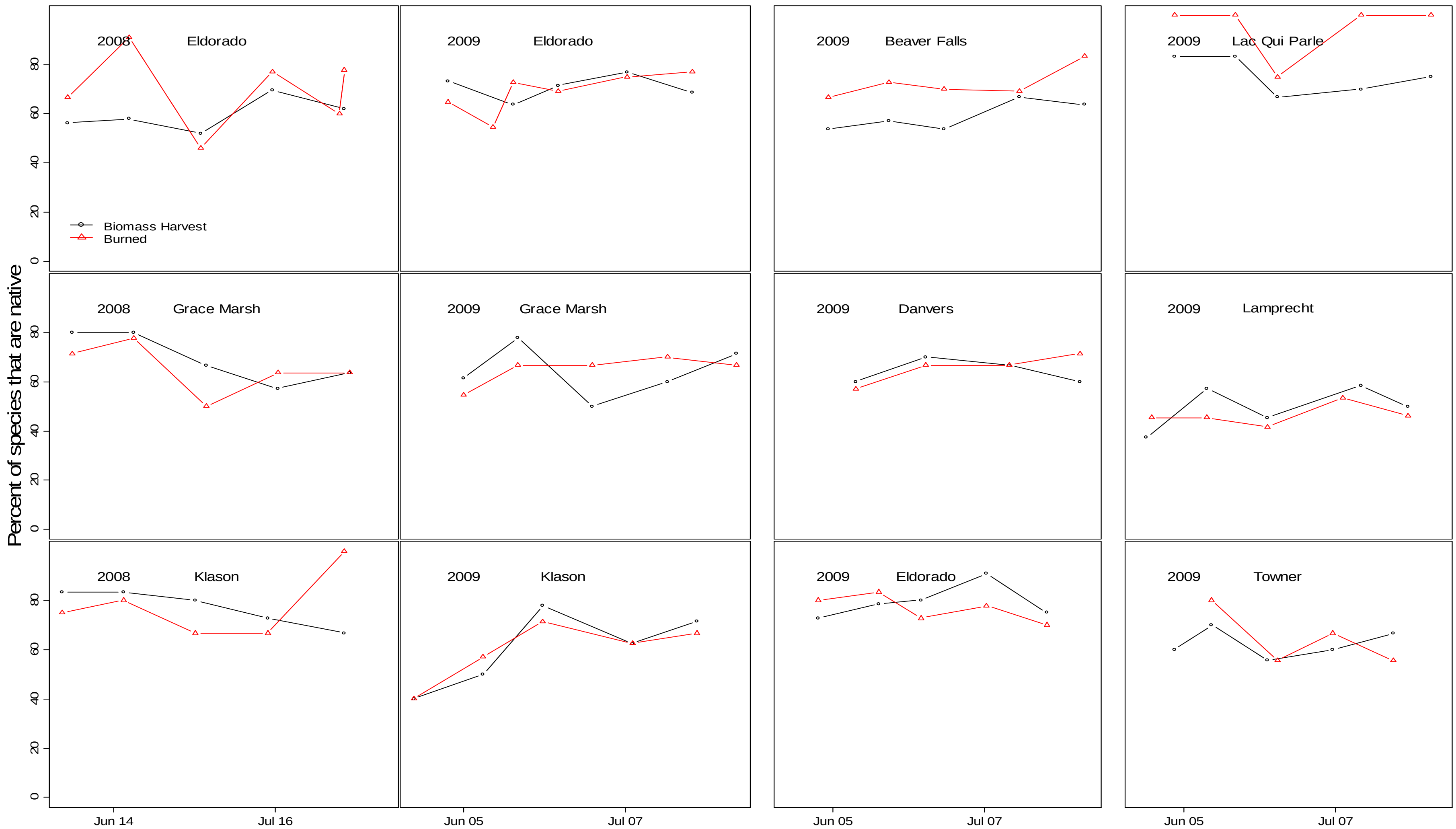


Figure 6. Comparison of the percent of native plant species per transect between 2 treatment subplots (a fall biomass harvest and a prescribed burn the following spring) within the same restored native grass field on 3 State Wildlife Management Areas in west-central Minnesota, in both summer 2008 and summer 2009 (leftmost two columns), and on 5 State Wildlife Management Areas and 1 Federal Waterfowl Production area in west-central Minnesota, in only summer 2009 (rightmost two columns).

MOVEMENTS, SURVIVAL, AND REFUGE USE BY RING-NECKED DUCKS AFTER FLEDGING IN MINNESOTA

Charlotte Roy, Christine Sousa, David Rave, Wayne Brininger[1], and Michelle McDowell[2]

SUMMARY OF FINDINGS

The Minnesota Department of Natural Resources (MNDNR) is conducting a study that examines use and survival benefits of waterfowl refuges to locally produced ring-necked ducks (*Aythya collaris*). During 2007 – 2009, we captured and implanted 176 flightless ring-necked ducks with radiotransmitters. Ducklings were tracked weekly by aircraft and from telemetry receiving stations located on 14 waterfowl refuges. The distance between weekly locations averaged ~8 km in all years. Young ring-necked ducks used state and federal waterfowl refuges, but this use was not evenly distributed among refuges; 3 refuges received the majority of use and 1 refuge has yet to be used by marked birds. Refuge use was also higher during hunting season than prior to the season opening. Additional data collection in 2010 will be aimed at increasing sample sizes to address survival benefits of refuge use to young birds.

INTRODUCTION

Sizable populations of resident breeding ducks were recognized as a cornerstone to improving fall duck use in the MNDNR Fall Use Plan. Although breeding ring-necked duck populations have been increasing continentally, they may be declining in Minnesota (Zicus et al. 2005). Furthermore, hunter harvest of ring-necked ducks has declined markedly in Minnesota in the last 40 years (U.S. Fish and Wildlife Service, Harvest Surveys, unpublished data), even as numbers of these birds staging on most traditional ring-necked duck refuges in the fall have increased in the state (MNDNR, unpublished data). Efforts to better understand population status began in 2003 with development of a ring-necked duck breeding-pair survey.

Factors influencing resident populations of ring-necked ducks are poorly understood. Further, the Fall Use Plan identified a need to better understand the role of refuges in duck management. The influence of north-central Minnesota refuges on the distribution and survival of resident ring-necked ducks is unknown.

The intent of this project was to determine whether refuges benefit locally produced ring-necked ducks and increase survival. Additionally, post-fledging ecology of many waterfowl species has not been documented, and this study provides information for an important Minnesota species. Understanding movements and refuge use in the fall may provide valuable insights into the distribution of refuges required to meet management objectives for ring-necked ducks in Minnesota.

OBJECTIVES

1. Characterize post-fledging movements of local ring-necked ducks prior to their fall departure;
2. Estimate survival of locally produced birds before migration; and
3. Relate survival of locally produced birds to the proximity between natal lakes and established refuges (Federal and State) and refuge use in north-central Minnesota.

[1] U.S. Fish and Wildlife Service, Tamarac National Wildlife Refuge, Rochert, MN
[2] U.S. Fish and Wildlife Service, Rice Lake National Wildlife Refuge, McGregor, MN

STUDY AREA

The study area lies primarily in the Laurentian mixed forest province of Minnesota (Figure 1). This area is characterized by mixed coniferous and hardwood forest interspersed with lakes, many of which are dominated by wild rice (*Zizania palustris*). The study area is ~200 x 135 km in size and encompasses a significant portion of the core of ring-necked duck breeding range in Minnesota and numerous important refuges for ring-necked ducks. Two federal and 12 state refuges were included in the study (Table 1). Lakes we monitored with remote receiving stations in this study were not open to public hunting, thus providing "refuge" for ducks during the fall migration.

METHODS

Night-lighting techniques were employed to capture flightless ring-necked ducks during July and August in 2007 – 2009. Duckling age (Gollop and Marshall 1954) and sex were determined at capture. We implanted radiotransmitters dorsally and subcutaneously on class IIb (~25 – 30 days old) and IIc (~31 – 38 days old) ring-necked ducklings following techniques developed by Korschgen et al. (1996), with 1 modification; we attached mesh to the back of transmitters (D. Mulcahy, US Geological Survey (USGS), Alaska Science Center, personal communication) to improve transmitter retention and minimize dehiscing that occurred during a pilot study in 2006. Ducks were then allowed several hours to recover from surgery before release at their capture location. We also marked ducklings with nasal saddles in 2007 to allow examination of natal philopatry in the spring, but because few birds were resighted, we discontinued marking with nasal saddles in 2008 and 2009.

By early September, radiotelemetry stations were established at each refuge as a means of quantifying refuge use. Receivers were programmed to scan each of the established frequencies periodically each hour, 24 hours/day. Data were downloaded weekly from data-loggers from mid-September through early November. Reference radiotransmitters were stationed permanently at each refuge to ensure receivers and data-loggers functioned properly.

Aerial flights with telemetry equipment were also conducted once weekly throughout the fall to document the locations and survival of radiomarked birds within the study area. Additional location and survival information came from USGS Bird Banding Lab banding and harvest reports. These reports include the hunters' names and the dates and locations of harvest.

RESULTS

Capture and Tracking

We captured 52 ducklings with night-lighting techniques between 4 August and 3 September 2007. In 2008, we captured 56 ducklings between 29 July and 26 August, and in 2009 we captured 68 ducklings between 27 July and 25 August. Capture locations were distributed throughout the study area, but more ducklings were captured on the western half of the study area in all years (31 in 2007, 32 in 2008, and 46 in 2009 in western counties compared to 21, 24, and 22 in each respective year in eastern counties, Table 2 and Figure 2).

The number of locations per bird varied from 1 to 14 (mean = 7.46, SE = 0.24) for the 176 marked birds. On average, 67% in 2007, 82% in 2008, and 82% in 2009 of birds were located weekly during aerial surveys beginning when the first bird was marked and continuing through early November. Success locating birds from aerial flights, however, was higher before hunting season (87% in 2007, 95% in 2008, 95% in 2009) than the week hunting opened in all years (66% in 2007, 83% in 2008, 83% in 2009). Success locating birds also declined as birds began moving more in preparation for migration.

Average weekly movements tended to increase as the fall progressed until mid to late October when birds started leaving the study area. For the tracking period, average weekly movements were 8.5 ± 1.9 km (mean ± SE) in 2007, 8.3 ± 2.1 km in 2008 and 7.2 ± 1.8 km in 2009, but average weekly movements prior to the start of hunting, after birds started moving (6.4 ± 1.1 km in 2007, 6.8 ± 1.6 km in 2008, 7.1 ± 1.7 km in 2009) were shorter than after hunting season opened (14.5 ± 3.0 km in 2007, 16.6 ± 3.5 km in 2008, 14.4 ± 2.4 km in 2009) in all years. All but 3 birds left their natal lake before hunting opened over the 3 year period.

Mortalities

In 2007, 15 radiomarked birds (*n* = 52) *were* known to have died by the end of the monitoring period (8 Nov); 5 were shot and retrieved by hunters (all in Minnesota), and 10 were depredated. Four of the 5 hunter-harvested birds were harvested during the first 2 days of the waterfowl hunting season (29 and 30 Sept). Evidence obtained at the recovery site indicated that radioed birds were either depredated or scavenged by mink (*Mustela vison*) and other mammals (7), or great-horned owls (*Bubo virginianus*) or other raptors (3). Six transmitters retrieved from open water in 2007 were thought to have dehisced; thus the fate of these birds was unknown. Six additional birds were harvested after the monitoring period ended; 3 were harvested during the 2007 hunting season (2 in Louisiana and 1 in Illinois), 2 were harvested in 2008 (1 in South Carolina and 1 in Arkansas), and 1 was harvested in 2009 (Arkansas).

In 2008, 25 radiomarked birds (*n* = 56) were known to have died by the end of the monitoring period (Nov 18); 8 were harvested by hunters (all in Minnesota), 11 were depredated, and 6 died of unknown causes. Four of the 8 hunter-harvested birds were shot during the first 2 days of the waterfowl hunting season (Oct 4 and 5). Radioed birds were either depredated or scavenged by mink, raccoon (*Procyon lotor*) and other mammals (5), raptors (1), and unknown sources (5) based on evidence at the recovery site. A cause of mortality could not be determined for 6 birds whose transmitters were found with no additional evidence at the site. Five radios were thought to have dehisced in 2008; and 2 of the birds which lost their radios were subsequently harvested (1 in 2008 in Oklahoma and 1 in 2009 in Cuba). Four additional birds were harvested after the monitoring period ended; 3 were harvested during the 2008 hunting season (2 in Louisiana, and 1 in South Carolina), and 1 was harvested during 2009 (Minnesota).

In 2009, 29 radiomarked birds (*n* = 68) were known to have died by the end of the monitoring period (Nov 9); 6 birds were shot by hunters (all in Minnesota), 12 were depredated, 10 died of unknown causes, and 1 may have died as a result of surgery. Two of the 6 harvested birds were shot during the youth opener (Sept 19) and 1 was shot during the first 2 days of the waterfowl hunting season (Oct 3 and 4). Radioed birds were either depredated or scavenged by mink, river otter (*Lontra canadensis*) and other mammals (9), raptors (1), and unknown sources (2). Four transmitters were thought to have dehisced in 2009, and the fate of these individuals was unknown. At the time of production of this document, 5 additional birds were harvested after the monitoring period ended during the 2009 hunting season (1 each in Alabama, Florida, Illinois, Missouri, and Texas).

Possible losses to predation prior to hunting season (7 in 2007, 12 in 2008, 21 in 2009) were similar or slightly higher than those during hunting (3 in 2007, 5 in 2008, 1 in 2009). Depredation earlier in the study period was expected to be higher, because during the first few weeks after marking, many ducklings are incapable of flight and more susceptible to predation. During hunting, some of the birds that appeared to have been depredated may have been wounded by hunters and later scavenged by predators. We x-rayed 2 birds that were recovered during the hunting season and found definitive shot pellets in 1 bird.

Refuge Use

Overall, 44 (25%) birds were documented at refuges based on aerial surveys and tower detections (17 in 2007, 11 in 2008, and 16 in 2009, Table 1). Although refuges were used before hunting season, use by radiomarked birds increased markedly with the onset of hunting (Figure 3). Numbers of ducks using refuges prior to hunting was less than during the hunting season. However, many birds were not capable of flight the first few weeks after capture.

Most refuges were used at least once during the study (Table 1); however not all refuges were used equally. The most heavily used refuges (based on number of marked birds) were Mud Goose (15), Drumbeater (12), and Tamarac NWR (10, Table 1). Rice Lake NWR has never been used by radiomarked ducklings, but this refuge was outside the capture area, and we expected use of this refuge by radiomarked birds to be less than for refuges located within the capture area. Most birds visited only 1 refuge (29 of 44 birds); however, a number of birds used more than 1 refuge during the fall period (Table 3).

Although use of individual refuges varied each year, a number of refuges were used every year: Mud Goose, Drumbeater, Tamarac NWR, and Rice Pond. In 2007, refuge use was documented for 17 radiomarked birds from both aerial and tower data. Six refuges were used by marked birds, but the most heavily used refuges based on number of birds located there were Mud Goose (6), Tamarac NWR (6), Fiske and Blue Rock (4), and Drumbeater (3). Several state refuges also received no documented use by radiomarked birds in 2007 (Table 1). A similar pattern was observed in 2008 with 11 radiomarked birds using 8 refuges. The most heavily used refuge was Mud Goose (6 birds; Table 1). In 2009, refuge use was documented for 16 radiomarked birds at 11 refuges during the fall migration. The most heavily used refuge in 2009 was Drumbeater (7 birds). The tower data are challenging to interpret and the number of birds detected by towers is subject to revision as we continue to analyze the data.

From the tower data, we also determined diurnal versus nocturnal use. Refuges could also be classified as day use (7:00 am and 6:00 pm), night use (7:00 pm to 6:00 am), and 24-hour use, based on the majority of observations occurring at various times during a 24 hour period (Table 4, Figure 4).

DISCUSSION

One more field season is anticipated. Methods in 2010 will be similar to those of 2008 and 2009. More formal analyses will be conducted at the conclusion of the study. Results and discussion of these analyses will be included in future Summaries of Wildlife Research Findings.

ACKNOWLEDGMENTS

M. Zicus was instrumental in planning, development, and initiation of this project. J. Berdeen helped with planning, surgeries, night-lighting, and hiring interns. N. Besasie, S. Bischof, D. Borhart, P. Christensen, B. Ferry, R. Hanauer, J. Kennedy, K. Kerfeld, J. Marty, T. Peterson, E. Sievers, E. Smith, and E. Zlonis helped capture and radiotrack birds as well as help establish remote towers. S. Cordts, J. Lawrence, K. Noyce, and B. Sampson helped with surgeries. J. Lawrence and S. Cordts also helped in the field, especially with aerial telemetry. J. Heineman flew telemetry flights, and A. Buchert flew us to Drumbeater Lake and helped set up the remote tower. J. Fieberg provided statistical advice. Dr. A. Piller and M. Kelly of Bemidji Veterinary Hospital ordered surgical equipment and gave technical advice. Dave Brandt, USGS Northern Prairie Research Center, loaned us 2 pairs of crimping pliers. John Finn and Martin Robinson of Leech Lake Division of Resource Management guided us into Drumbeater Lake and retrieved equipment from Drumbeater Lake each winter. R. Lego allowed access to the Leech River, and D. Barrett, North Country Regional Hospital Surgery Department, sterilized transmitters. Partial funding provided by the U.S. Fish & Wildlife Service, Region 3 Migratory Bird Division.

LITERATURE CITED

Gollop, J. B. and W. H. Marshall. 1954. A guide to aging duck broods in the field. Unpublished report. Mississippi Flyway Council.

Korschgen, C. E., K. P. Kenow, W. L. Green, M. D. Samuel, and L. Sileo. 1996. Technique for implanting radiotransmitters subcutaneously in day old ducklings. Journal of Field Ornithology 67:392 – 397.

Zicus, M. C., D. P. Rave, J. Fieberg, J.Giudice, and R. Wright. 2005. Minnesota's ring-necked ducks: a pilot breeding pair survey. Pages 137 – 158 *in* P. J. Wingate, R. O. Kimmel, J. S. Lawrence, and M. S. Lenarz, editors. Summaries of Wildlife Research Findings 2004. Minnesota Department of Natural Resources, St. Paul, Minnesota, USA.

Table 1. National Wildlife Refuges and Minnesota State Refuges included in the study area, approximate location of the refuges, peak numbers of ring-necked ducks during fall migration, number of recording telemetry stations established on each refuge, and the use of each refuge by radiomarked post-fledging ring-necked ducks during 2007 – 2009. Note that the tower data are challenging to interpret and the number of birds detected by towers is subject to revision as we continue to analyze the data.

Refuge	Location	~Peak numbers	Stations	No. radiomarked birds using Refuge		
				2007	2008	2009
National Wildlife Refuge						
Rice Lake	5 mi SSW of McGregor	120,000	4	0	0	0
Tamarac	16 mi NE Detroit Lakes	50,000	3	6	1	3
State Waterfowl Refuge/State Game Refuge						
Donkey Lake	6 mi SW Longville	350	1	1	0	1
Drumbeater Lake	2 mi N of Federal Dam	280,000	1	3	2	7
Fiske and Blue Rock Lakes	8 mi SE Northhome	40,000	1	4	0	0
Gimmer Lake	10 mi SE Blackduck	3,500	1	0	3	0
Hatties and Jim Lakes	13 mi SE Blackduck	0	1	0	0	1
Hole-in-Bog Lake	2 mi SW Bena	4,000	1	0	0	4
Mud Goose	4 mi SSW of Ballclub	4,000	1	6	6	3
Lower Pigeon Lake	4 mi S Squaw Lake	700	1	0	1	3
Pigeon River Flowage	6 mi S Squaw Lake	700	1	0	1	3
Preston Lakes	22 mi ENE of Bemidji	1,800	1	0	2	2
Round Lake	8 mi N Deer River	11,000	1	0	0	2
Rice Pond	9 mi E of Turtle River	32	1	2	2	2

Table 2. Ring-necked duckling captures per county in Minnesota during 2007 – 2009.

County	Captures		
	2007	2008	2009
Aitkin	1	0	2
Becker	6	1	4
Beltrami	17	7	17
Cass	9	10	7
Clearwater	5	15	13
Hubbard	3	7	7
Itasca	9	10	11
Koochiching	2	4	2
Polk	0	2	3
Wadena	0	0	2

Table 3. Number of ring-necked ducklings that used 1 or more refuges, Minnesota 2007 – 2009.

No. refuges visited	No. birds
1	29
2	8
3	4
4	2
5	0
6	1

Table 4. Minnesota refuges classified as day use, night use, and 24-hour use based on data collected by monitoring equipment established to detect refuge use by radiomarked post-fledging ring-necked ducklings.

Day use	Night use	24-h2our use	Not used
Donkey	Pigeon River	Mud Goose	Rice Lake NWR
Drumbeater	Rice Pond	Round	Tamarac NWR - Chippewa
Fiske Blue Rocks		Tamarac NWR – Little Flat	
Gimmer			
Hatties and Jim			
Hole-in-Bog			
Lower Pigeon			
Preston Lakes			
Tamarac NWR – Flat			

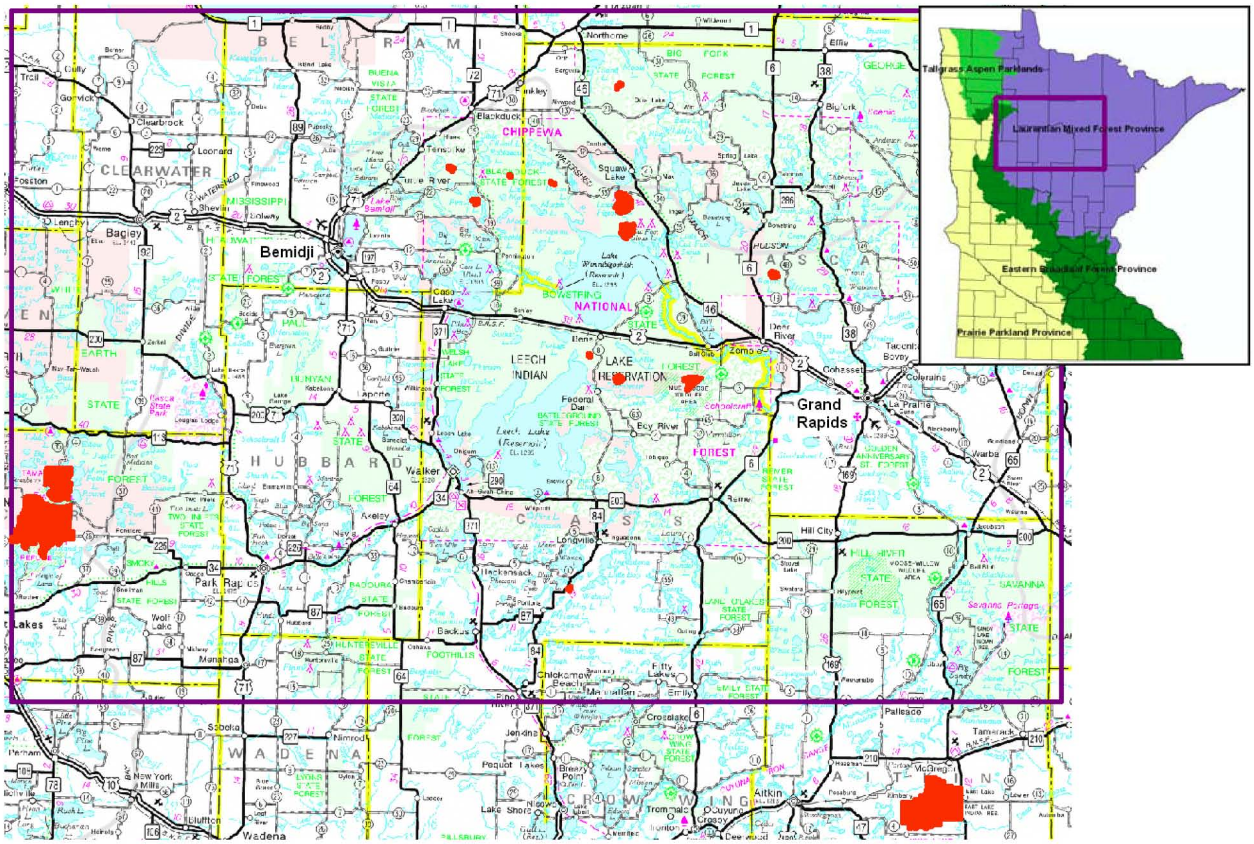


Figure 1. Ring-necked duck study area in Minnesota during 2007 – 2009 with 12 state waterfowl/game refuges and 2 National Wildlife Refuges depicted in red.

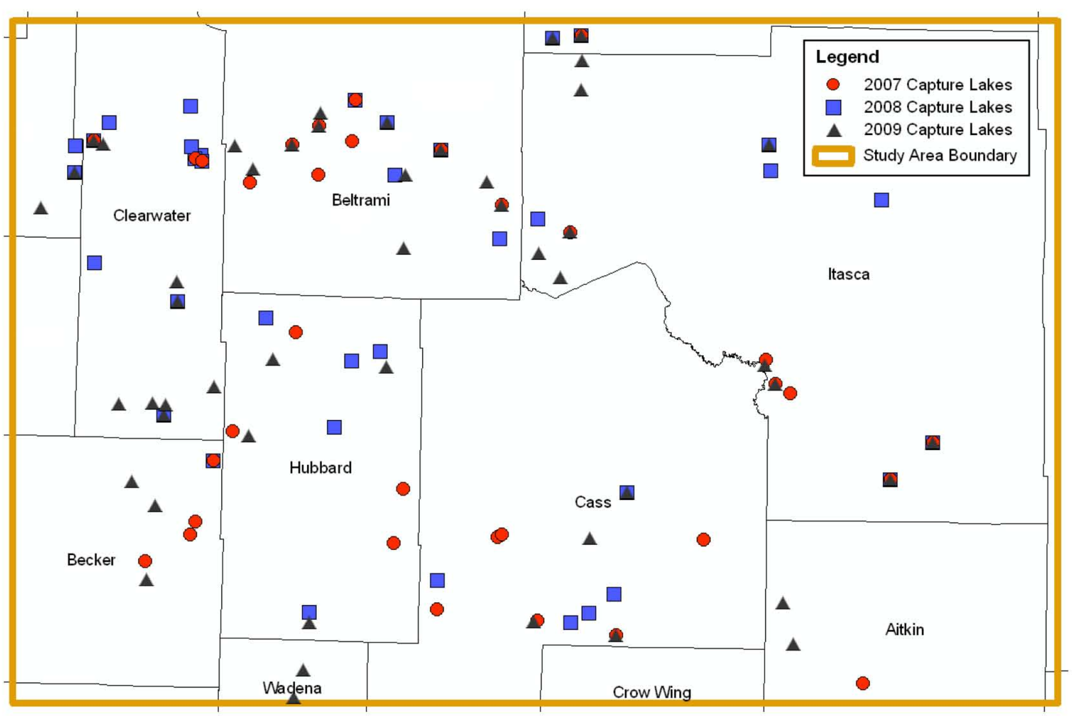


Figure 2. Capture locations for ring-necked duck ducklings in Minnesota during 2007 – 2009.

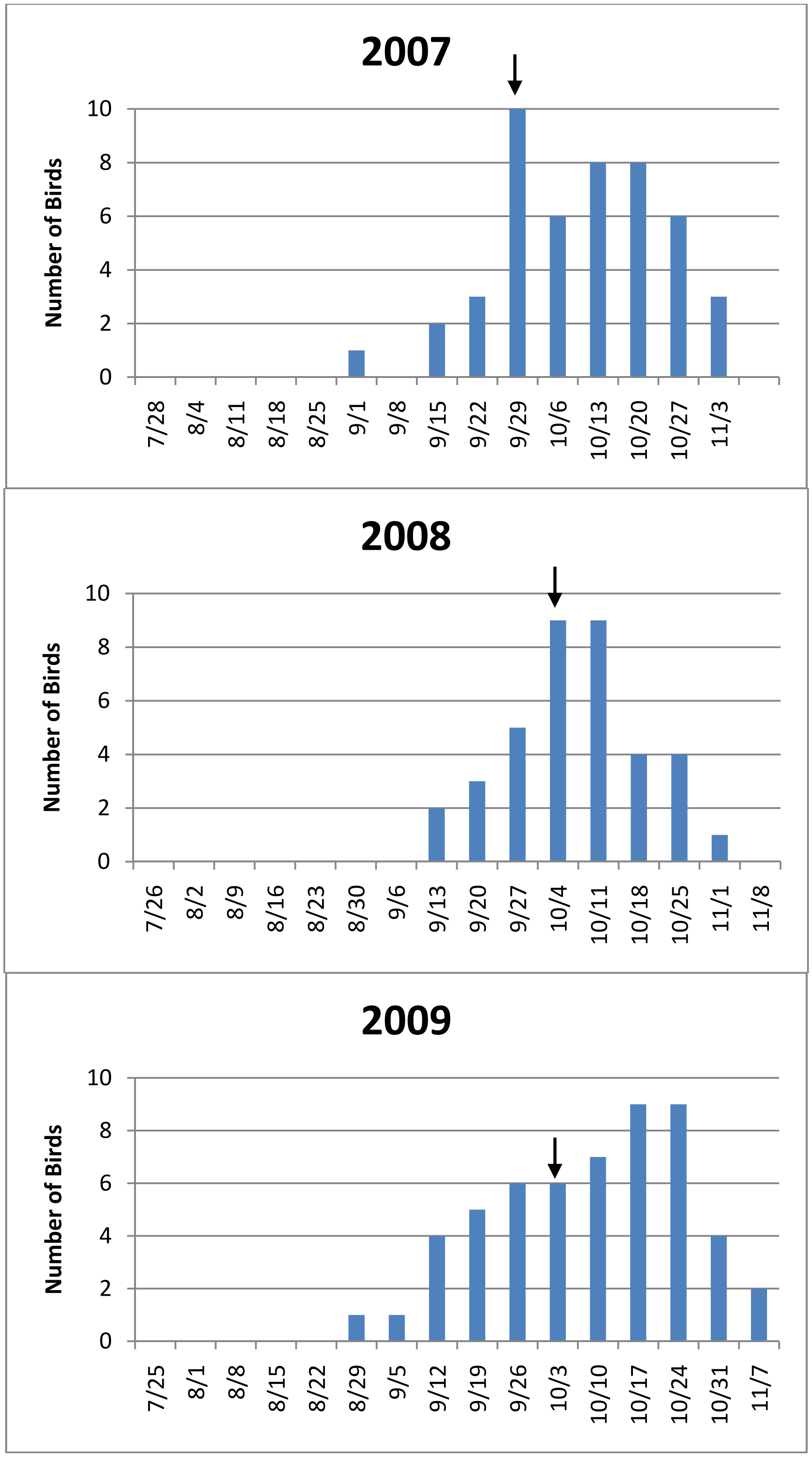


Figure 3. Weekly use of refuges by post-fledging ring-necked ducks before and during hunting season in 2007 – 2009 in Minnesota. Weeks are from Saturday through Friday with the Saturday date shown. Arrows indicate the week waterfowl hunting opened.

A
B
C

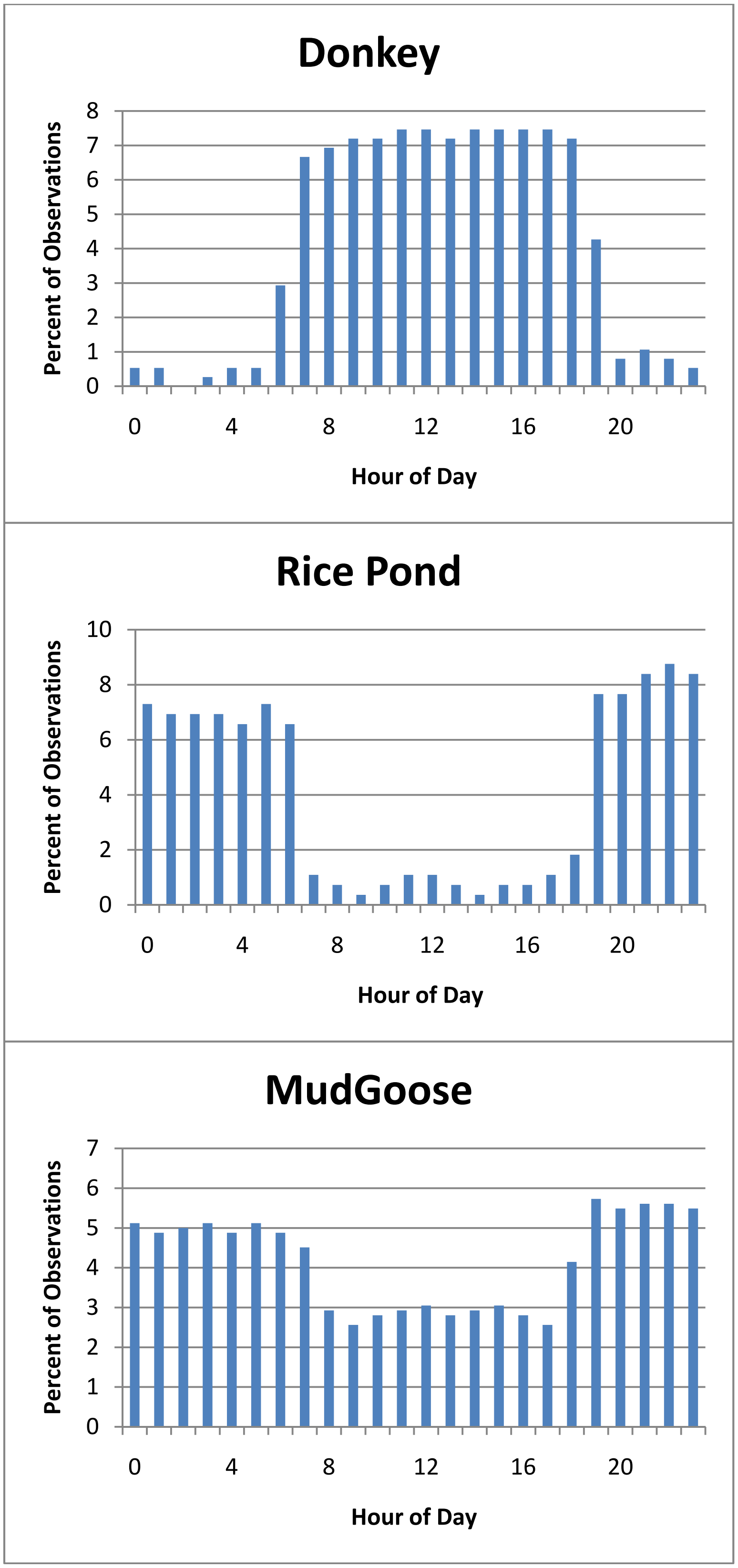


Figure 4. Example refuge use data to show the difference among day use (A), night use (B), and 24-hour use (C) refuges in Minnesota during 2007 - 2009.

REGIONAL COMPARISONS OF RELATIONSHIPS AMONG LANDSCAPE SETTING, AMBIENT NUTRIENTS, LAND USE, FISH COMMUNITIES, AND ECOLOGICAL CHARACTERISTICS OF SHALLOW LAKES - PRELIMINARY EFFORTS – 2009

Mark A. Hanson, Brian R. Herwig[1], Jerry A. Younk[1], Kyle D. Zimmer[2], Sean R. Vaughn[3], Robert W. Wright[4], Shane Bowe[5], Jim Cotner[6], and Patrick G. Welle[7]

SUMMARY

Minnesota's shallow lakes provide numerous valuable ecosystem services and habitat for native species along with direct human benefits including clean water, recreational opportunity, and carbon sequestration. Unfortunately, water and habitat quality of Minnesota's shallow lakes have deteriorated dramatically during the past century. Conversion from native upland covers, widespread wetland drainage, and surface-water consolidation to facilitate agricultural and urban/residential development have been implicated as major causes for these changes. We are studying approximately 136 shallow lakes in 5 ecological regions of Minnesota to: (1) identify major factors leading to deterioration, (2) evaluate results of specific lake restoration approaches, including cost-effectiveness of various combinations of lake management strategies; and (3) assess the impacts of increased surface water connectivity on fish invasions and resulting habitat quality. Our efforts include extensive sampling of shallow lakes to identify direct and indirect causes of deterioration, evaluation of approximately eight lakes currently undergoing rehabilitation, and economic analyses to determine which restoration strategies are likely to produce the greatest improvements in water quality and other lake characteristics per unit cost. Ultimately, our results will allow municipalities, state, county, and local governments, and private organizations to identify cost-effective approaches for maintaining and restoring ecological integrity of shallow lakes throughout Minnesota. Special attention will be directed towards development of regionally-specific recommendations for sustainable lake management.

BACKGROUND

Minnesota has approximately 4,000 lakes characterized by mean depth $\leq$ 15 ft and surface area > 40 acres (Nicole Hansel-Welch, personal comm.) and many thousands of smaller waters technically classified as "prairie wetlands"; the latter are functionally indistinguishable from the larger analogues (Potthoff et al. 2008). Collectively, these shallow lakes represent an international resource, providing critical waterfowl habitat and ecological benefits within Minnesota and the Mississippi Flyway. Currently, only 40 of these lakes > 40 acres are formally designated for wildlife management, however many others are focus areas for various wildlife habitat and conservation practices. Due to concerns over shallow lake water

---

[1]Fisheries Research, Populations and Community Ecology Group, Minnesota Department of Natural Resources, 2114 Bemidji Avenue, Bemidji, MN 56601
[2]Department of Biology, University of St. Thomas, 2115 Summit Avenue, St. Paul, MN 55105
[3]Division of Waters, Minnesota Department of Natural Resources, 800 Oak Savanna Lane SW, Cambridge, MN 55008
[4]Wildlife GIS/IT Unit, Minnesota Department of Natural Resources, Carlo-Avery Game Farm, 5463-C West Broadway, Forest Lake, MN 55025
[5]Red Lake Department of Natural Resources, 15761 High School Drive, Red Lake, MN 56671
[6]Department of Ecology, Evolution, and Behavior, University of Minnesota, 100 Ecology Building, 1987 Upper Buford Circle, St. Paul, MN 55108
[7]Department of Economics, DH20 # 30, Bemidji State University, Bemidji, MN 56601

quality, seasonal duck abundance and habitat use, and hunter satisfaction, MNDNR recently proposed a collaborative plan to Recover Ducks, Wetlands, and Shallow Lakes (http://files.dnr.state.mn.us/outdoor_activities/hunting/waterfowl/duck_plan_highlights.pdf). This plan targets restoration of 1,800 shallow lakes in Minnesota. At the same time, restoration strategies available to shallow lake managers remain limited and often ineffective; in addition, reliable data on baseline conditions of shallow lake characteristics and regional patterns of variability are often unavailable, especially for northern areas. This means that lake and wildlife managers are frequently unsure of the current status of lakes they manage, and whether ecological characteristics of these areas may be limiting use by waterfowl and other wildlife. In general, managers receive little technical guidance useful for management and restoration of these lakes, or for implementation of rules for managing increased development and other anthropogenic influences in these areas.

Ecological characteristics of shallow lakes, along with their suitability for ducks and other wetland wildlife species, result from integrated influences of within-site and landscape-mediated processes. Effects of key variables operate at multiple spatial scales, sometimes result from off-site influences, and no doubt vary regionally throughout the state. Ecologists have long held that prairie wetlands (including our “shallow lakes”) are strongly influenced by gradients of hydrology (or hydrogeomorphic setting) and climate (especially precipitation) (Euliss et al. 2004). However, within boundaries established by hydrology and climate, biological interactions, especially wetland fish communities, also exert major structuring influences on communities and characteristics of prairie wetlands and shallow lakes (Hanson et al. 2005). This is not surprising given robust improvements known to follow removal of undesirable fishes from shallow Minnesota lakes such as Christina (Hanson and Butler 1994), and smaller “prairie pothole” wetlands (Zimmer et al. 2001).

As evidenced by whole-lake manipulations such as those summarized above, shallow lake food webs often differ dramatically in response to density and community structure of associated fish populations. Fish-mediated influences on invertebrate community structure and water transparency are often pronounced (Bendell and McNicol 1987; Zimmer et al. 2000, 2001). Recent studies in Minnesota’s Prairie Pothole Region (PPR) documented the strong negative influences of fathead minnows on invertebrate populations (Zimmer et al. 2000, 2001, 2002). Consequent reductions in herbivorous zooplankton (resulting from fish predation) allowed increases in phytoplankton densities and turbidity consistent with predictions of the models of Scheffer et al. (1993) and Scheffer (1998). These models propose that shallow-water ecosystems exist in one of two alternative conditions, either a clear-water, macrophyte-dominated state, or a turbid-water, phytoplankton-dominated state (Scheffer et al. 1993). Minnesota PPR wetlands largely conform to a binomial distribution (clear or turbid), rather than a normal distribution of features along a theoretical continuum (Zimmer et al. 2001; Herwig et al. 2004; Zimmer et al. 2009a).

Composition of fish assemblages may also mitigate the relative influence of fish on shallow lake communities, and may dictate the success of remediation efforts. For example, stocking of piscivorous fish often results in a reduction of planktivorous fish (especially soft-rayed minnows), which may indirectly increase water transparency (Walker and Applegate 1976; Spencer and King 1984; Herwig et al. 2004). Similarly, in small lakes in northern Wisconsin containing natural fish communities, piscivores (largemouth bass *Micropterus salmoides* or northern pike *Esox lucius*) and cyprinids often occupy unique and separate assemblages (Tonn and Magnuson 1982; Rahel 1984). This pattern is thought to reflect the elimination of minnows via predation, and further suggests that biotic interactions can be important in structuring fish assemblages. In contrast, populations of large-bodied benthivorous fish species (e.g., black bullhead *Ameiurus melas*, white sucker *Catostomus commersoni*, and common carp *Cyprinus carpio*) are often resistant to predation, and are frequently associated with high turbidity and loss of rooted aquatic plants (Hanson and Butler 1994; Braig and Johnson 2003; Parkos et al. 2003). Due to the important but very different influences of planktivorous and benthivorous fishes on water quality, and the potential for restoration success given different fish assemblages, managers would benefit from tools that linked fish assemblages to landscape features and environmental characteristics of shallow lakes themselves.

Many lake and wetland studies have reported that landscape setting directly influences characteristics of embedded waters. For example, the watershed position sets boundaries on a variety of physical, chemical, and biological attributes of both deep lakes (Kratz et al. 1997) and prairie wetlands (Euliss et al. 2004). These lake properties include potential responses to drought, predominant groundwater interactions, water chemistry and concentrations of dissolved constituents, and biological communities. Other landscape features that have been found to influence lake water quality are wetland extent in the lake watershed (Detenbeck et al. 1993; Prepas et al. 2001), and extent of agricultural land use, the latter being correlated with higher trophic state index in associated lakes (Detenbeck et al. 1993). In many cases, off-site influences probably interact with site-level wetland features and processes so that observed community characteristics reflect simultaneous influences operating within the local context of lake nutrient status (Scheffer et al. 1993; Bayley and Prather 2003; Jackson 2003), surface area (Hobæk et al. 2002), depth (Scheffer et al. 1993), and biological properties such as abundance of macrophytes (Scheffer et. al. 1993; Paukert and Willis 2003; Zimmer et al. 2003).

Our previous work (2005-06) confirmed that landscape characteristics can influence lake communities, interact with within-basin processes, and may be important determinants of shallow lake characteristics in Minnesota. These landscape effects are direct and indirect. For example, both presence of downstream fish sources and depth were useful for predicting fish presence/absence (Herwig et al. 2010), and landscape control on distribution of fish species limited the ability of predatory fish to control prey fish and improve water quality conditions (Friederichs et al. In revision). Extent of agriculture in upstream lake watersheds interacted with fish mass in our best models and together these attributes were useful for predicting algal biomass in adjacent shallow lakes (Gorman et al. In prep.), and fish variables were always included in best models for predicting amphibian site occupancy and abundance in shallow lakes (Herwig et al. In Prep.). In addition, results from our previous study helped elucidate mechanisms associated with important in-lake processes such as identifying thresholds at which shallow lakes shift from turbid- to clear-water regimes, and clarifying roles of benthivorous fish in these well-known lake dynamics (Zimmer et al. 2009a). Preliminary results from earlier work indicate that fish abundance and community structure exert major influences on shallow lake invertebrates, yet this relationship varies widely across ecological region. We are also comparing relative influences of within-site and landscape-scale characteristics on shallow lake invertebrate communities. Contributions from Sean Vaughn (Division of Waters, MNDNR) and Robert Wright (Section of Wildlife, MNDNR) provided new spatial analysis tools (delineating lake watershed boundaries, spatial analysis, etc.) that were not only critical for the recently-completed study, but will have direct application to questions and hypotheses posed in this current effort.

Major goals of our previous study were to develop conceptual and empirical models linking landscape features, environmental influences and wetland fish assemblages, to assess influences of these factors on the community characteristics in shallow lakes, and to clarify specific influences of within-lake processes that influence ecological characteristics of shallow lakes. An overarching finding of the prior work was that regional differences typically constituted the largest source of variance in characteristics of shallow Minnesota lakes. This is not unexpected given findings of others studying deeper lakes (Carpenter et al. 2007), or observations from MNDNR shallow lakes program staff indicating that baseline characteristics of shallow lakes differ dramatically across regions of the state (Nicole Hansel-Welch, pers. comm.). Regional differences not only contribute to major variability in obvious lake characteristics such as water clarity, but they probably influence extent and nature of lake responses to landscape constraints such as surface-water connectivity, and within-lake processes in regime responses to thresholds of phytoplankton and fish mass. For example, it is likely that combinations of increased benthivorous fish mass and/or decreased macrophytes will often induce regime shifts in prairie lakes, and these changes probably portend shifts to turbid-water states. However, we speculate that increased fish mass is much less likely to induce turbid-states in north-central Minnesota lakes, and turbid states may not even be possible in northern lakes where low ambient nutrient levels prevail. Additional work is needed to document extent and patterns of regional variation, and to assess how it influences key structuring mechanisms such as surface connectivity, fish community characteristics, stability of phytoplankton- and macrophyte-dominated states, and proportion of lakes in clear- vs. turbid-water states.

RESEARCH APPROACHES

Extensive Lakes

We are currently gathering data from, and characterizing watershed features of, 128 shallow lakes (hereafter Extensive lakes) from 6 regions of Minnesota. Lakes will be sampled once each July in 2009-11 to assess general ecological features and determine whether basins exhibit characteristics of clear- or turbid-water regimes. Lake watershed characteristics associated with each study lake will also be determined by creating and applying numerous lake watershed variables via GIS technology and interpretation of aerial photography. Resulting data will be used to develop models to identify combinations of variables that explain most variability in shallow lake characteristics, especially water quality features and lake regime status (turbid or clear). Special attention will be given to assessing influences of resident fish populations, extent of surface-water connectivity associated with study lakes, and proportion of agriculture in lake watersheds because these are believed to be major determinants of water quality in Minnesota's shallow lakes. Resulting data will help identify and estimate magnitude of major factors responsible for deterioration of water quality and ecological characteristics in our regional subsets of study lakes.

Intensive Lakes

During 2010-11 we will also evaluate responses of 8 shallow lakes (hereafter Intensive lakes) currently undergoing lake restoration treatments such as draw downs or fish community manipulation. Ecological characteristics of Intensive lakes will be sampled monthly from May-August each year, including all components measured in the 128 Extensive sites. Identical landscape-level analyses will be conducted on these areas to determine upland cover and surface-water connectivity in lake watersheds using GIS analysis and interpretation of aerial photographs. Combining results and data from Intensive and Extensive lakes, we will estimate water quality improvements in response to various combinations of rehabilitation treatments including upland restoration and within-lake-basin measures such as fish community manipulation. Specific efforts will be directed to evaluating responses of the Intensive lakes to management efforts applied on each lake.

Connectivity Emphasis

Ecological health of shallow lakes is a reflection of their upstream and downstream watersheds and the hydrologic connectivity within those flow networks. Increased surface water connectivity due to drainage, ditching, road construction, and other anthropogenic activities is known to increase the transfer of organisms, especially undesirable fishes, among shallow lakes in Minnesota. Such connectivity probably also provides major pathways for the spread of invasive species, which threaten native communities.

We will identify, delineate and digitize unmapped natural and human-induced water conveyance features that constitute present-day surface water connectivity. Using data from the Extensive (128) and Intensive lakes (8), we propose to document water quality, biodiversity, habitat characteristics, and measure lake responses to various surface water connectivity scenarios. This will allow the development of models useful for assessing probable results from increased surface water connectivity within the watersheds. We believe this will provide useful data and guidance for natural resource managers who frequently evaluate requests for landscape modifications that increase surface-water connectivity, runoff and channelized flow.

Economic Analysis

An economic analysis will be conducted using the empirical data from all study lakes in identifying water quality improvements (such as cost per unit of algae reduced [µg/L chlorophyll *a*]) resulting from various application of various management options being utilized or considered within Minnesota. We plan to quantify the costs of applying various combinations of upland vegetation restoration (conversion of agriculture to grass) and in-lake habitat enhancements (fish removal, installation of barriers, etc.) to achieve a given measure of lake water quality improvement. We expect that costs of management options will vary widely among ecological regions due to regionally variability in lake characteristics, lakesheds, upland easement costs, property values, and other attributes of lakes and adjacent uplands.

Comparison of restoration costs will be informative and will help elucidate trade-offs on temporal and spatial scales. Some options may generate quick results but may need to be repeated frequently, so that variations in long-run costs (over multiple decades) will be important to consider. Easement costs for land to be restored to vegetative buffers are known to vary across regions of the state. Cost data for the management options being studied are known to be currently available or obtainable.

Working Hypotheses

Our overall, general working hypothesis is that 6 fundamental "drivers" are ultimately responsible for most of the variation in ecosystem characteristics of Minnesota shallow lakes: climate, ambient nutrient levels, fish abundance and community type, landscape features, land use, and morphometric features of individual lakes. These 6 factors, in turn, induce strong, predictable spatial gradients in shallow lake characteristics across Minnesota. Thus, we expect shallow lakes will exhibit wide ranges of features (and responses to lake management) at a statewide scale as the influence of some drivers increase while others decrease. Additionally, inter-annual and regional variability in precipitation and temperature will have strong influences on shallow lakes. Thus, we hypothesize these drivers generate predictable spatial and temporal patterns in shallow lakes across the state of Minnesota. Overall, we believe that understanding and predicting ecosystem characteristics of shallow lakes (fish, plant and invertebrate communities, water quality, carbon cycling, etc.), along with lake responses to rehabilitation efforts, requires understanding the influence of these drivers, as well as synergistic combinations of influences arising from two or more drivers. Within-lake interactions, such as those associated with fish, have strong influences on shallow lakes (Scheffer et al. 2006; Verant et al. 2007; Potthoff et al. 2008). However, we hypothesize that strengths of these interactions are also a function of our main drivers such that within-lake interactions will contribute to spatial and temporal patterns that can be predicted from these influences.

We believe it is also especially important to test further hypotheses regarding stability regimes in shallow lakes. Previous work (Hanson and Butler 1994) suggests that shallow lakes in MN conform to general models of alternative states developed for European lakes (Scheffer et al. 1993, Scheffer 1998) and these relationships have recently been confirmed from our prior work on Minnesota lakes (Zimmer et al. 2009a). However, in Minnesota, it is likely that regime dynamics and stability thresholds will vary along regional gradients. We expect that companion models may need to be developed that extend concepts of lake regimes to include patterns of variance in invertebrate communities and other lake characteristics. Results from all study lakes will be used to estimate the magnitude of major factors responsible for deterioration of shallow lakes within the 6 study regions. Comparisons among management outcomes on 8 Intensive lakes will allow generalizations about relative usefulness of these lake rehabilitation approaches. Using a combination of data and outcomes from Extensive and Intensive lakes, our economic analysis will compare cost-effectiveness of various management approaches and will provide guidelines useful for maximizing future lake restoration and management decisions, including suggestions as to how more cost-effective approaches vary across the state. Finally, all resulting data will be used to assess extent to which surface connectivity among surface waters influences ecological characteristics of shallow study lakes.

METHODS

Study Areas

A key goal of our study is to increase understanding of spatial patterns of shallow lake characteristics across Minnesota. Shallow lakes here occur across a wide range of lake watershed characteristics (agriculture and urban land uses, native cover types, etc.), phosphorus concentrations, and water transparency gradients. We used an aquatic ecoregion approach for characterizing shallow lake features (sensu Heiskary et al. 1987). We used classifications based on Omerik's (1987) Level III ecoregion delineations denoting areas of general similarity in the type, quality, and quantity of environmental resources. Under this approach we established a study area (or "study landscape"), each containing a cluster of study sites, within each of the 5 ecoregions that collectively encompass the vast majority of lakes and wetlands in Minnesota: Northern Minnesota Wetlands (NMW), Northern Lakes and Forests (NLF), Northern Glaciated Plains (NGP), Western Corn Belt Plains (WCP), and North Central Hardwood Forests (CHF). As previously mentioned, there are large gradients in in-lake phosphorus (P) and nitrogen concentrations across these ecoregions. For example, in a survey of 1,062 lakes, Heiskary et al. (1987) found median P concentrations of 23 ppb (NLF), 50 ppb (CHF), 121 ppb (WCP), and 177 ppb (NGP). No information was available for NMW, but we expect lower P concentrations here, perhaps intermediate between NLF and CHF. Cover types also vary widely, ranging from heavily forested, with some marshlands (NLF) to nearly level marsh, containing both boreal vegetation and expansive swamps (NMW), to principally cropland agriculture (WCP & NGP), to a mosaic of cover types, including forests, wetlands and lakes, cropland agriculture, pasture, grasses, and urban development (CHF) (Omerik 1987).

Our study focuses on 6 landscape areas distributed across 5 aquatic ecoregions within Minnesota as follows: (1) the NMW study area (hereafter "Red Lake") is located within the boundaries of the Red Lake Indian Reservation in far northern Clearwater and west-central Beltrami counties, (2) a NLF study area (hereafter "Itasca") is positioned within and around Itasca State Park in south-eastern Clearwater County, (3) a second NLF study area with sites located in western portions of the Chippewa National Forest in far western Itasca County (hereafter "Chippewa"), (4) the NGP study landscape (hereafter "Elbow Lake") located in the southern portions of Grant County, extending into the northern and western margins of Stevens and Douglas counties, respectively (we have a long time series here, dating back the mid 1990's), (5) the WCP study area (hereafter "Windom") centered around Windom, MN, and thus roughly split between Cottonwood and Jackson counties, (6) the CHF study landscape (hereafter "Twin Cities") located in the Hennepin-Carver county metro area (Figure 1).

Our study landscapes are also positioned in several different major river watersheds. In some cases, study areas fall within two or more drainages. For example, Red Lake is entirely within the Red River drainage, Itasca is entirely within the Upper Mississippi River drainage, but Twin Cities is within both the Upper Mississippi River and Minnesota River drainages. Similarly, Windom is within the Minnesota River and Lower Mississippi River drainages, and Elbow Lake is within the Red River and Minnesota River drainages.

Individual Study Sites

Within each study landscape, we are studying up to 24 shallow lakes, measuring fish assemblages, wetland characteristics, and surrounding landscape attributes. Study lakes were distributed across both public and private ownerships, and all lakes are of semipermanent or permanent (type IV or V) duration of flooding (Shaw and Fredine 1956; Stewart and Kantrud 1971). Within these broad classifications, shallow lakes span a range of values of surface area, depth, and adjacent upland cover types.

General Data Collection Approaches

Development of land use and lake watershed variables using GIS and air photo interpretation

GIS data layers will be used to derive metrics that characterize features of the landscape associated with each study site, including proportions of the dominant cover types at the watershed-scale as well as upstream and downstream hydrological connectivity. Lake watershed boundaries will be delineated for each site using the delineation methods of Sean Vaughn (2009). Existing land cover layers (perhaps MN GAP or land cover layers developed by the University of Minnesota's Remote Sensing and Geospatial Laboratory: http://land.umn.edu) will then be overlaid and summarized for the individual lake watersheds. Data summaries will be developed as needed and will primarily include connectivity attributes and watershed characteristics (e.g. surface area of different cover types, inter-lake surface connection distances, watershed:lake area ratios).

Landscape/watershed connectivity analyses may include but are not limited to the following: (1) presence of upstream/downstream connections to surface waters capable of supporting fish populations; (2) modeled upstream/downstream connections of surface water from digital elevation models (DEM) to surface waters capable of supporting fish populations; (3) distances to represent "as the fish swims" to surface waters capable of supporting fish populations (horizontal and vertical dimensions); and (4) rank variable for type and degree of connectivity to other surface waters (also a potential proxy for geomorphic setting).

Fish assemblages

Fish species composition and relative abundance (biomass per unit effort) will be determined using a combination of gears deployed overnight. All fish sampling will be done during July and August each year. Three mini-fyke nets (6.5 mm bar mesh with 4 hoops, 1 throat, 7.62 m lead, and a 0.69 X 0.99 m rectangular frame opening into the trap) will be set overnight in the littoral zone of each lake. One experimental gill net (61.0 m multifilament net with 19, 25, 32, 38, and 51-mm bar meshes) will be set along the deepest depth contour available in lakes less than 2 m deep or along a 2 m contour in lakes with sufficient depth. The protocol outlined above has been shown to be effective in sampling fish assemblages in shallow lakes in Minnesota (Herwig et al. 2010) as well as small lakes from other regions (Tonn and Magnuson 1982; Rahel 1984; Jackson and Harvey 1989; Robinson and Tonn 1989). This should enable us to capture both small- and large-bodied fish, and species from all of the major trophic guilds (e.g., plankivores, benthivores, piscivores) potentially present in the study wetlands. All fish sampled will be sorted by species, rated (counts per unit weight), and weighed in bulk. Fish data will likely be quantified as the summed total biomass of each species collected in all four nets. Voucher specimens will be collected and returned to the laboratory for identification when field identification cannot be made.

Aquatic invertebrates

Zooplankton will be sampled once per year in July concurrent with fish sampling by collecting two replicate vertical column samples (Swanson 1978) at 5 locations in each wetland. Estimates will be made of density and taxon richness. Relative abundance of macroinvertebrates will be sampled concurrent with other sampling in July using sweep net samples (Murkin et al. 1983) at 0.75m depth at 5 randomly selected locations in each lake. Abundance and taxon richness of macroinvertebrates will be measured.

Nutrients, specific conductance, light attenuation, and phytoplankton

Surface (dip) water samples will be taken from the center of each lake once during July concurrent with other sampling. Samples will be frozen and transported to the University of St. Thomas for analysis of chlorophyll *a*, total nitrogen, total phosphorus and total dissolved phosphorus. Turbidity will be measured in the field with a portable nephelometer. Phytoplankton biomass will be estimated from chlorophyll *a* (Strickland and Parsons 1972). Collection of samples for chlorophyll *a* simultaneously with measurement of turbidity will allow assessment of the contribution of phytoplankton to turbidity, and ultimately to light attenuation.

Submerged macrophytes

Abundance of submerged macrophytes and *Chara* spp. will be assessed using modified techniques of Jessen and Lound (1962), and Deppe and Lathrop (1992). In each lake, submerged macrophytes will be sampled at 15 stations located equidistant along four transects running the width of each basin in July or August of each year. Two throws of a weighted plant rake will be made at each station, and dragged along 3 m of lake bottom. Plants collected on the first throw will be weighed (all taxa combined) and frequency of occurrence (1 = sampled on one throw, or 2 = sampled on both throws) will be recorded for each plant species sampled. Plant data will be summarized as mass and frequency of occurrence (all taxa combined) summed across the total number of throws used for each metric.

Earthworms

We will study earthworm effects on shallow lakes in only one region due to lack of facilities and personnel for examining this phenomenon elsewhere. Earthworms will be collected from uplands within 50 m of all study lakes in our Itasca core area. Near each lake, 10 35 cm x 35 cm areas will be cleared of surface duff and flooded with a saturated solution of mustard (after methods of Laurence and Bowers 2002). Extracted worms will be collected, preserved in 75% ethanol, and identified according to an ecological classification system of Hale et al. (2005). Data will be used to develop a relative abundance estimate for earthworms in catchment areas immediately adjacent to study lakes.

We will correlate earthworm abundance and ecological classifications with the nutrient concentrations, chlorophyll *a*, and other water quality characteristics in adjacent study basins. Earthworm collections will be restricted to lakes within our Itasca core area due to relatively uniform forest composition in this ecological region (enabling earthworm effects to be assessed independent of other factors) and because related measurements require laboratory facilities available at the University of Minnesota field station in Itasca State Park. It is also important to note that students (using non-project funds provided to J. Cotner) will be collecting ancillary data on forest characteristics and soils in this region.

Intensive sampling

In consultation with Minnesota Ducks Unlimited staff, we recently identified 8 case study lakes to evaluate effectiveness of restoration strategies typically used by state, federal, and private organizations working on shallow lake management. We plan to assess effectiveness of various combinations of lake rehabilitation approaches including installation of fish barriers, water level draw downs, rotenone, and perhaps other measures commonly used by lake managers in Minnesota. Study sites include lakes that have been restored by drawdown and re-flooding during the past 2-4 years.

We will measure the effectiveness of the various management activities by assessing changes in ecosystem features following the specific manipulation to the lake. Variables to be assessed include water clarity, nutrient levels in the water column, and abundance and species composition of phytoplankton, submerged macrophytes, zooplankton, macroinvertebrates, and fish. The relative improvement in each of these variables will be assessed using data from our larger Extensive study for lakes in the same ecoregion. For example, it is difficult to quantify the degree of “successfulness” following lake-drawdown when duration of improvements and responses of submerged plants cannot be predicted. However, the Extensive (128) lake study will help to quantify this change if lakes shift from turbid-to clear-water states following drawdown. Interpreting lake response in the context of natural regional variability should also facilitate assessment of success across ecoregions where lake features naturally vary.

Data Analysis

We anticipate applying a suite of analysis strategies to evaluate the various hypotheses outlined above. This is necessary because no single approach we are aware of allows for identification and measurement of multiple complex linkages discussed above. Our approach will include gradient analysis (ter Braak 1995; ter Braak and Smilauer 1998; McCune and Grace 2002), classification and regression tree techniques (Breiman et al. 1984, De’ath and Fabricus 2000), variance partitioning (Borchard et al. 1992; ter Braak and Wiertz 1994), mixed effect linear models (Littell et al. 2006), piecewise regression (Toms and Lesperance 2003), information-theoretic model selection techniques (Burnham and Anderson 1998; Anderson et al. 2000), and traditional parametric approaches (SLR, ANOVA) (Zar 1999). Collectively, our analyses are intended to provide evidence whether ecological features of study lakes differ in predictable ways (thus whether lakes can be grouped) and, if so, whether fish communities, landscape and lake watershed features, cover types, ambient nutrients, lake basin morphology, and climate and other regional patterns account for observed differences among groups. Analyses will likely include situations where data are pooled from all landscapes to ensure a considerable range of values in both predictor and response variables, and situations where analyses will be developed for each study landscape separately, especially if separate modeling improves predictive ability, or if region-specific prediction and models are required.

Synthesis and Expected Research Products

We will use data from 8 Intensive and 128 Extensive lakes and from characterization of associated watersheds to address our working hypotheses. Along with results from our economic analysis, we will suggest management guidelines for shallow lakes based on data and outcomes from specific ecological regions of the state. Study results will be synthesized and distributed in the form of several peer-reviewed manuscripts and a project summary, the latter to be developed specifically for shallow lake managers in Minnesota.

PROGRESS TO DATE

During July 2009, we gathered data from 128 Extensive lakes in our 6 study regions (Figure 1). In each lake, we measured water transparency and lake depth, and collected surface-dip samples for water-column concentrations of phosphorus and nitrogen in major pools, and phytoplankton biomass (chlorophyll *a*). At the same time, we gathered samples of aquatic macroinvertebrates and zooplankton, and conducted surveys to estimate relative abundance of submerged aquatic plants. We also assessed presence and composition of fish communities in each lake. Sediment samples were gathered from selected lakes within boundaries of Itasca State Park, from lakes within our Chippewa core, and from Alexandria, Twin Cities, and Windom areas. Samples for determination of major nutrient and chlorophyll *a* concentrations are being processed using facilities available at the University of St. Thomas (St. Paul, MN). Similarly, samples of zooplankton (column samples) and macroinvertebrates are being enumerated in the lab during January - May 2010. Watershed delineations have been mostly completed for the 128 Extensive lakes and we are in final stages of selecting 8 sites for intensive aspects described above. Presently, we are awaiting a final decision on a proposal submitted to Legislative Citizens Commission on Minnesota Resources (funding has been approved). During the past year, Dr. Kyle Zimmer (our study collaborator) received National Science Foundation support (funding started in July 2009) for studies of carbon burial in a subset of our study lakes and we expect to partner on some aspects of data gathering to facilitate transfer of study results and interpretation between projects (Zimmer et al. 2009b). Also, during 2010-2011, we plan more detailed measurements of groundwater contributions on a subset of lake sites to better assess extent to which geomorphic setting influences ecological characteristics of these sites (details in Bischof et al. 2010).

Although results of formal analyses are not yet available, several preliminary observations are noteworthy. Climate/precipitation gradients were strong in Minnesota in 2009 with a number of Twin Cities lakes in severe drawdown during July. In contrast, we observed normal to high water levels in Windom, and normal water levels in our other study areas. These patterns should have important influences on shallow lake characteristics, and will likely vary yearly. Contrary to our expectations, preliminary data indicate that turbid regimes (sensu Zimmer et al. 2009a) are possible for lakes within the NLF ecoregion in north-central Minnesota, at least one site was turbid in both the Itasca and Chippewa study areas. Shallow lakes in our Windom study area were not all turbid.

Trends in fish communities across study regions were also surprising. For example, our Itasca study area, especially lakes within Itasca State Park, had the highest prevalence of fishless sites of the 6 landscapes we studied. The Chippewa study area had the highest richness of planktivorous and piscivorous fish, and benthivorous fish richness was higher than Itasca, but similar to other study areas. The higher number of planktivorous and piscivorous fish species probably reflects the more widespread distribution of certain minnows (northern redbelly dace, central mudminnow, golden shiner), yellow perch, and northern pike within the Chippewa. Average biomass of planktivores and piscivores was also higher in the Chippewa than other study areas (Figure 2). Although carp are widely distributed and are often associated with poor water quality in shallow lakes, some lakes in the Windom area did not have carp. Fish communities and fish densities changed sharply in several of Elbow Lake sites compared to 2006 (when we last worked on these sites), and many of these lakes switched from turbid to clear regimes, presumably due to lower fish densities, perhaps following winterkill.

During 2010, we expect to continue comprehensive food web, sediment, and water chemistry sampling of 128 shallow lakes (Extensive sites). At the same time, we plan to begin sampling the 8 case study lakes (Intensive sites) and to start developing datasets that will allow us to assess shallow lake responses to various lake rehabilitation efforts. We also plan to begin developing conceptual models to assess cost-effectiveness of lake rehabilitation activities We expect that final review of watershed delineations for our shallow lake study sites, and development of land cover summaries will be completed within the next year.

LITERATURE CITED

Anderson, D.R., Burnham, K.P., and Thompson, W.L. 2000. Null hypothesis testing: problems, prevalence, and an alternative. Journal of Wildlife Management 64: 912-923.

Bayley, S.E., and Prather, C.M. 2003. Do wetland lakes exhibit alternative stable states? Submersed aquatic vegetation and chlorophyll in western boreal shallow lakes. Limnology and Oceanography 48: 2335-2345.

Bendell, B.E., and McNicol, D.K. 1987. Fish predation, lake acidity and the composition of aquatic insect assemblages. Hydrobiologia 150: 193-202.

Bischoff, S., M. A. Hanson, B. R. Herwig, and T. J. Kroeger. 2010. Identifying proxy variables for assessing influences of lake geomorphic setting: A research prospectus. Submitted in April 2010 to Section of Wildlife, Minnesota DNR, St. Paul, MN.

Borchard, D., Legendre, P., and Drapeau, P. 1992. Partialling out the spatial component of ecological variation. Ecology 73: 1045-1055.

Braig, E.C., and Johnson, D.L. 2003. Impact of black bullhead (*Ameiurus melas*) on turbidity in a diked wetland. Hydrobiologia 490: 11-21.

Breiman, L., Friedman, J.H., Olshen, R.A., and Stone, C.J. 1984. Classification and Regression Trees. Chapman & Hall (Wadsworth, Inc.), New York, NY.

Burnham, K.P., and Anderson, D.R. 1998. Model selection and inference: a practical information-theoretic approach. Springer-Verlag, New York, NY.

Carpenter, S.R., Benson, B.J., Biggs, R., Chipman, J.W., Foley, J.A., Golding, S.A., Hammer, R.B., Hanson, P.C., Johnson, P.T.J., Kamarainen, A.M., Kratz, T.K., Lathrop, R.C., McMahon, K.D., Povencher, B., Rusak, J.A., Solomon, C.T., Stanley, E.H., Turner, M.G., Vander Zanden, M.J., Wu, C.-H., and Yuan, H. 2007. Understanding regional change: a comparison of two lake districts. BioScience 57: 323-335.

De'ath, G., and Fabricius, K.E. 2000. Classification and regression tree: a powerful yet simple technique for ecological data analysis. Ecology 81(11): 3178-3192.

Deppe, E.R., and Lathrop, R.C. 1992. A comparison of two rake sampling techniques from sampling aquatic macrophytes, Wisconsin Bureau of Research Findings 32, Madison.

Detenbeck, N.E., Johnston, C.A., and Niemi, G.J. 1993. Wetland effects on lake water quality in the Minneapolis/St. Paul metropolitan area. Landscape Ecology 8: 39-61.

Euliss, N.H., Jr., LaBaugh, J.W., Fredrickson, L.H., Mushet, D.M., Laubhan, M.K., Swanson, G.A., Winter, T.C., Rosenberry, D.O., and Nelson, R.D. 2004. The wetland continuum: a conceptual framework for interpreting biological studies in the prairie pothole region. Wetlands 24: 448-458.

Friederichs, S., K.D. Zimmer, B.R. Herwig, and M.A. Hanson. In Review. Top-down and bottom-up influences on shallow lake food webs. Submitted to Oikos.

Gorman, M.W., Zimmer, K.D., Herwig, B.R., Hanson, M.A., Wright, R.G., Vaughn, S.R., and Younk, J.A. In prep. Ambient nutrient levels, fish communities, and watershed characteristics as drivers of phytoplankton abundance in shallow lakes. (Target journal: Ecological Applications)

Hale, C. M., L. E. Frelich and P.B. Reich. 2005. Exotic European Earthworm Invasion Dynamics in Northern Hardwood Forests of Minnesota, USA. Ecological Applications 15: 848-860.

Hanson, M.A., and Butler, M.G. 1994. Responses of plankton, turbidity, and macrophytes to biomanipulation in a shallow prairie lake. Canadian Journal of Fisheries and Aquatic Sciences 51: 1180-1188.

Hanson, M.A., Zimmer, K.D., Butler, M.G., Tangen, B.A., Herwig, B.R., and Euliss, N.H., Jr. 2005. Biotic interactions as determinants of ecosystem structure in prairie wetlands: an example using fish. Wetlands 25: 764-775.

Heiskary, S.A., Wilson, C.B., and Larsen, D.P. 1987. Analysis of regional patterns in lake water quality: using ecoregions for lake management in Minnesota. Lake and Reservoir Management 3: 337-344.

Herwig, B.R., Hanson, M.A., Reed, J.R., Parsons, B.G., Potthoff, A.J., Ward, M.C., Zimmer, K.D., Butler, M.G., Willis, D.W., and Snook, V.A. 2004. Walleye stocking as a tool to suppress fathead minnows and improve habitat quality in semipermanent and permanent wetlands in the prairie pothole region of Minnesota. Minnesota Department of Natural Resources, Division of Fisheries and Wildlife, Special Publication 159, St. Paul, MN.

Herwig, B.R., L.W. Schroeder, K.D. Zimmer, M.A. Hanson, D.F. Staples, R.W. Wright, and J.A. Younk. In Prep. Factors influencing amphibian site occupancy and abundance patterns in permanent wetlands in prairie and parkland landscapes of Minnesota, U.S.A.: roles of fish, basin characteristics, and adjacent landscape configuration. (Target journal: Biological Conservation)

Herwig, B.R., Zimmer, K.D., Hanson, M.A., Konsti, M.L., Younk, J.A., Wright, R.W., Vaughn, S.R., and Haustein, M.D. 2010. Factors influencing fish distributions and community structure in shallow lakes within prairie and prairie-parkland regions of Minnesota, USA. Wetlands 30:609-619.

Hobaek, A., Manca, M., and Andersen, T. 2002. Factors influencing species richness in lacustrine zooplankton. Acta Oecologia 23: 155-163.

Jackson, D.A., and Harvey, H.H. 1989. Biogeographic associations in fish assemblages: local vs. regional processes. Ecology 70: 1472-1484.

Jackson, L.J. 2003. Macrophyte-dominated and turbid states of shallow lakes: evidence from Alberta lakes. Ecosystems 6: 213-223.

Jessen, R., and Lound, R. 1962. An evaluation of a survey technique for submerged aquatic plants, Minnesota Department of Conservation Game Investigational Report 6, St. Paul.

Johnson, D.H. 1999. The insignificance of significance testing. Journal of Wildlife Management 63: 763-772.

Kratz, T.K., Webster, K.E., Bowser, C.J., Magnuson, J.J., and Benson, B.J. 1997. The influence of landscape position on lakes in northern Wisconsin. Freshwater Biology 37: 209-217.

Laurence, A. P. and M. A. Bowers. 2002. A test of the 'hot' mustard extraction method of sampling earthworms. Soil Biology and Biochemisty, 34: 549-552.

Littell, R.C., Milliken, G.A., Stroup, W.W., and Wolfinger, R.D. 1996. SAS Systems for Mixed Models. SAS Institute Inc., Cary, North Carolina, USA.

McCune, B., and Grace, J.B. 2002. Analysis of Ecological Communities. MJM Software Design, Gleneden Beach, OR.

Murkin, H.R., Abbott, P.G., and Kadlec, J.A. 1983. A comparison of activity traps and sweep nets for sampling nektonic invertebrates in wetlands. Freshwater Invertebrate Biology 2: 99-106.

Omernik, J.M. 1987. Ecoregions of the conterminous United States. Map (scale 1:7,500,000). Annals of the Association of American Geographers 77: 118-125.

Parkos III, J.J., Santucci, V.J., Jr., and Wahl, D.H. 2003. Effects of adult common carp (*Cyprinus carpio*) on multiple trophic levels in shallow mesocosms. Canadian Journal of Fisheries and Aquatic Sciences 60: 182-192.

Paukert, C.P., and Willis, D.W. 2003. Aquatic invertebrate assemblages in shallow prairie lakes: fish and environmental influences. Journal of Freshwater Ecology 18: 523-536.

Potthoff, A.J., Herwig, B.R., Hanson, M.A., Zimmer, K.D., Butler, M.G., Reed, J.R., Parsons, B.G., and Ward, M.C. 2008. Cascading food-web effects of piscivore introductions in shallow lakes. Journal of Applied Ecology 45:1170-1179.

Prepas, E.E., Planas, D., Gibson, J.J., Vitt, D.H., Prowse, T.D., Dinsmore, W.P., Halsey, L.A., McEachern, P.M., Paquet, S., Scrimgeour, G.J., Tonn, W.M., Paszkowski, C.A., and Wolfstein, K. 2001. Landscape variables influencing nutrients and phytoplankton communities in Boreal Plain lakes of northern Alberta: a comparison of wetland- and upland-dominated catchments. Canadian Journal of Fisheries and Aquatic Sciences 58: 1286-1299.

Rahel, F.J. 1984. Factors structuring fish assemblages along a bog lake successional gradient. Ecology 65: 1276-1289.

Robinson, C.L.K., and Tonn, W.M. 1989. Influence of environmental factors and piscivory in structuring fish assemblages of small Alberta lakes. Canadian Journal of Fisheries and Aquatic Sciences 46: 81-89.

Scheffer, M. 1998. Ecology of Shallow Lakes. Champman and Hall, London, UK.

Scheffer, M., Hosper, S.H., Meijer, M.-L., Moss, B., and Jeppesen, E. 1993. Alternative equilibria in shallow lakes. Trends in Ecology and Evolution 8: 275-279.

Scheffer, M., VanGeest, G.J., Zimmer, K., Jeppesen, E., Søndergaard, M., Butler, M.G., Hanson, M.A., Declerck, S., and De Meester, L. 2006. Small habitat size and isolation can promote species richness: second-order effects on biodiversity in shallow lakes and ponds. Oikos 112: 227-231.

Shaw, S.P., and Fredine, C.G. 1956. Wetlands of the United States, U.S. Fish Wildlife Service Circular 39, Washington, D.C.

Spencer, C.N., and King, D.L. 1984. Role of fish in regulation of plant and animal communities in eutrophic ponds. Canadian Journal of Fisheries and Aquatic Sciences 41: 1851-1855.

Stewart, R.E., and Kantrud, H.A. 1971. Classification of natural ponds and lakes in the glaciated prairie region, Bureau of Sport Fisheries and Wildlife Resources Publication No. 92, Washington, D. C.

Strickland, J., and Parsons, T.R. 1972. A practical handbook of seawater analysis. Bull. Fish. Res. Bd. Can. 167.

Swanson, G.H. 1978. A plankton sampling device for shallow wetlands. Journal of Wildlife Management 42: 670-672.

ter Braak, C.J. 1995. Ordination. *In* Data Analysis in Community and Landscape Ecology. *Edited by* R.H. Jongman, C.J. ter Braak and O.F. van Tongeren. Cambridge University Press, Cambridge, U.K.

ter Braak, C.J., and Smilauer, P. 1998. Reference Manual and User's Guide to Canoco for Windows: Software for Canonical Community Ordination (version 4). Microcomputer Power, Ithaca, NY, USA.

ter Braak, C.J., and Wiertz, J. 1994. On the statistical analysis of vegetation change: a wetland affected by water extraction and soil acidification. Journal of Vegetation Science 5: 361-372.

Toms, J.D., and Lesperance, M.L. 2003. Piecewise regression: a tool for identifying ecological thresholds. Ecology 84: 2034-2041.

Tonn, W.M., and Magnuson, J.J. 1982. Patterns in the species composition and richness of fish assemblages in northern Wisconsin lakes. Ecology 63: 1149-1166.

Vaughn, S.R., 2009. DNR Watershed Delineation: *Project History, Methodology, Terminology & Data Attribution*. Minnesota Department of Natural Resources - Waters.

Verant, M.L., Konsti, M.L., Zimmer, K.D., and Deans, C.A. 2007. Factors influencing nitrogen and phosphorous excretion rates of a fish community in a shallow lake. Freshwater Biology 52: 1968-1981.

Walker, R.E., and Applegate, R.L. 1976. Growth, food, and possible ecological effects of young-of-the-year walleyes in a South Dakota prairie pothole. The Progressive Fish-Culturist 38: 217-220.

Zar, J.H. 1999. Biostatistical Analyses. Fourth Edition. Prentice Hall, Upper Saddle River, New Jersey, USA.

Zimmer, K.D., Hanson, M.A., and Butler, M.G. 2000. Factors influencing invertebrate communities in prairie wetlands: a multivariate approach. Canadian Journal of Fisheries and Aquatic Sciences 57: 76-85.

Zimmer, K.D., Hanson, M.A., and Butler, M.G. 2001. Effects of fathead minnow colonization and removal on a prairie wetland ecosystem. Ecosystems 4: 346-357.

Zimmer, K.D., Hanson, M.A., and Butler, M.G. 2002. Effects of fathead minnows and restoration on prairie wetland ecosystems. Freshwater Biology 47: 2071-2086.

Zimmer, K.D., Hanson, M.A., and Butler, M.G. 2003. Relationships among nutrients, phytoplankton, macrophytes, and fish in prairie wetlands. Canadian Journal of Fisheries and Aquatic Sciences 60: 721-730.

Zimmer, K.D., M.A. Hanson, B.R. Herwig, and M.L. Konsti. 2009a. Thresholds and stability of alternative regimes in shallow prairie-parkland lakes of central North America. Ecosystems 12:843-852.

Zimmer, K.D., J. Cotner, M. Edlund, and K. Theissen. 2009b. Organic carbon burial in shallow lakes. Submitted to NSF Biological Sciences Program, Division of Environmental Biology, Ecosystem Science, Washington, DC. (Awarded 1 July 2009; $1,227,272)

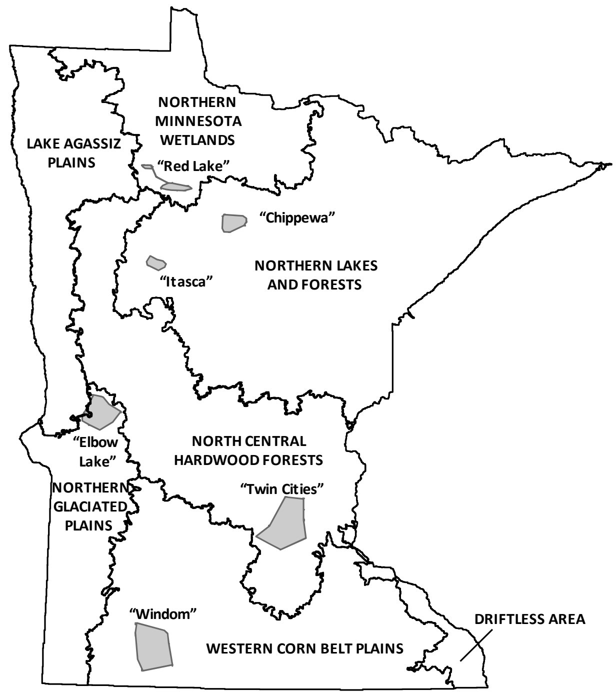


Figure 1. Map showing locations of study landscapes (shaded gray) in relationship to Minnesota's aquatic ecoregions (thick black lines).

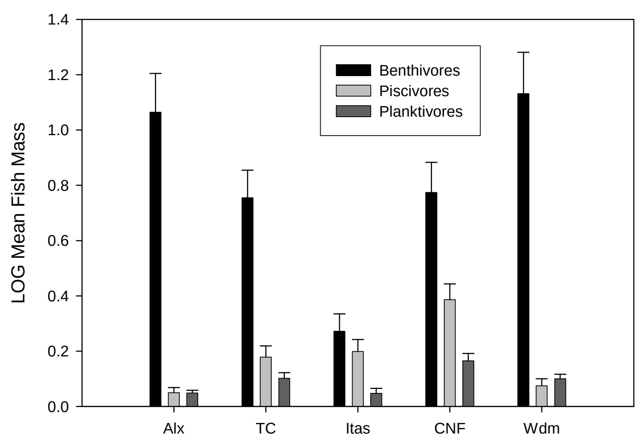


Figure 2. Log_{10}- transformed mean fish mass of benthivores, piscivores, and planktivores collected from shallow lakes in 5 study regions in Minnesota during 2009. X-axis labels depict each of 5 study areas, Alx = Alexandria, TC = Twin Cities/Metro, Itas = Itasca State Park and surrounding, CNF = Chippewa National forest, and Wdm = Windom. Vertical bars indicate 1 SE.

NESTING ECOLOGY OF RING-NECKED DUCKS IN NORTHERN MINNESOTA

Charlotte Roy, Christine Sousa, Jody Kennedy[1], and Elizabeth Rave[1]

SUMMARY OF FINDINGS

We have completed 2 years of field work on this research project. Thus far, we have searched 75 wetlands, located 38 nests, marked 22 hens, and followed 15 broods. We have searched lakes with (16%) and without (84%) boat accesses, 48-1583 m from roads, near both dirt (48%) and paved roads (52%), with houses (49%) and without houses (51%). Nest success is within the range of previous reports for north-central Minnesota. Hen and brood survival estimates require additional data collection to enable interpretation.

INTRODUCTION

The ring-necked duck (*Aythya collaris*) is a characteristic and important species for the Laurentian Mixed Forest province of Minnesota (MN DNR 2006). Recent surveys of 14 lakes important for ring-necked duck breeding near Bemidji have indicated declines in ring-necked duck numbers, despite increases elsewhere in their breeding range (Zicus et al. 2005). Unfortunately, basic information on nest success, hen survival, and brood survival in north-central Minnesota are unavailable to enable a more informed interpretation of these local survey data and to understand how vital rates affect population growth of ring-necked ducks in the forest. These data are pertinent given the increasing development and recreational use in the forest (MN DNR 2006) and predictions that the spruce-fir forest will shift north of Minnesota as a result of global climate change (Iverson and Prasad 2001).

OBJECTIVES

1. To obtain baseline information on ring-necked duck nest success, hen survival, and brood survival before fledging in the Laurentian forest; and
2. To examine how these vital rates vary along a gradient of human development and recreational use (e.g., number of dwellings, boat access, proximity to roads).

STUDY AREA

The study area is approximately 65 km x 65 km and lies in the heart of the Laurentian mixed forest province of Minnesota. This area is characterized by mixed coniferous and hardwood forest interspersed with lakes. Wetlands in the area commonly have wild rice (*Zizania palustris*) or other emergent vegetation, sedges (*Carex* spp.), and floating bog mats along the margins.

METHODS

We searched for ring-necked duck nests in the spring and summer of 2008 and 2009. We used multiple methods and data sources to identify lakes to search, including locations of pairs and lone males from a ring-necked duck helicopter survey conducted 2004-2009 and from ground surveys conducted on 10-14 lakes in the Bemidji area beginning in 1969. The survey data were used to identify land cover attributes of wetlands that ring-necked ducks used (GAP types 12 and 13 surrounded by GAP types 10, 14, and 15). We identified 103 lakes within a 25 mile radius of Bemidji with similar land cover attributes to those used in the 2 surveys and also searched the 6 lakes within our study area which had pairs or lone males in the ring-necked

[1] Department of Biology, Bemidji State University, Bemidji, MN

duck survey. In 2008, we searched 39 lakes. In 2009, we scouted 101 wetlands in early spring and focused nest searching efforts on the 36 wetlands where ring-necked ducks were seen. Scouting wetlands for ring-necked ducks before nest searching improved the efficiency of searches. We searched lakes with and without boat accesses (16% and 84%, respectively), 48-1583 m from roads, near both dirt (48%) and paved roads (52%), both with and without houses (49 and 51%, respectively).

To locate ring-necked duck nests, we searched emergent vegetation on floating bog mats and along wetland margins using bamboo poles and nest drags. When a nest was located, we determined the stage of incubation by candling eggs (Weller 1956) and from the appearance of new eggs in the nest. At each nest and at one random point located 25 m from the nest, we determined water depth, concealment using a Daubenmire frame and Robel pole (Daubenmire 1959, Robel et al. 1970), predominant vegetation (e.g., cattail (*Typha* spp.), sedge), and distance to open water. Wetland size, distance to roads and dwellings, and wetland class were determined in GIS for use in models of nest survival.

Late in incubation, we trapped hens on nests with Weller traps (Weller 1957) to attach radiotransmitters. Because a surgical transmitter attachment method might be disruptive to incubating hens, we tried a bib-type transmitter attachment method which had been used with previous success in wood ducks (*Aix sponsa*; Montgomery 1985). This attachment method was faster and less invasive than surgical methods. Hens received a transmitter fastened to a Herculite® fabric bib with dental floss and superglue (total weight of approximately 11 g). We modified the method used unsuccessfully in redheads (*Aythya americana*) by Sorenson (1989) by securing the bib more tightly and by preening the bib into the breast feathers as in Montgomery (1985). After the transmitter was in place, we trimmed any excess fabric so that feathers concealed the transmitter. We released birds at the edge of the wetland. Nests were monitored every 4-7 days to determine fate (abandoned, depredated, or successful).

After nests hatched, we monitored broods every 3-4 days. At each observation, we counted the ducklings present and aged them based on plumage characteristics (Gollop and Marshall 1954). Broods were monitored for 60 days or until total brood loss occurred. We considered hens to have lost their entire brood if hens were observed without any ducklings for 5 observations or if the hen was found >10 miles from the nesting lake. We continued to monitor hens after the brood-rearing period to examine hen survival until migration using the Kaplan-Meier method (Kaplan and Meier 1958).

RESULTS

Nest Survival

We searched for nests on 39 wetlands a total of 73 times between 22 May-22 July 2008 and 36 wetlands 54 times between 29 May-22 July 2009. We located 18 and 20 nests on 10 wetlands each year. The return per unit effort (i.e., #nests found/search) in 2009 was greater than the first year. In 2008, 8 nests hatched, 4 were depredated when found, 3 were depredated after they were found, and 3 nests were flooded by rising water levels following rain events. Average clutch size was 9.1 ± 0.6 (range: 7-15, n = 12) for nests known to be complete and 86.6 ± 0.1% of eggs (n = 109 eggs) hatched in successful nests.   In 2009, 7 nests hatched, 9 were depredated after they were found, and 4 were abandoned, with at least 2 cases of abandonment likely due to trapping. The average clutch size was 8.3 ± 0.3 (range: 7-11, n = 19 nests, 158 eggs) and 89.5 ± 0.6% of the eggs hatched in nests that were successful. Ring-necked ducks nests were found predominately in leatherleaf (*Chamaedaphne calyculata*; 35%) and sedge (65%). Mayfield nest success for 35-days of laying and incubation (Mendall 1958, Mayfield 1975) was 29.6% in 2008 and 26.5% in 2009. Three females that lost their nest early in the season were later seen with males but no evidence of re-nesting was detected.

Hen Survival

We put transmitters on 8 hens in 2008 and 14 hens in 2009. In 2008, 2 hens died due to predation during the tracking season; 1 lost her nest late in incubation and the other had a brood. Both of these birds had been documented preening more than other birds with transmitters, although this behavior occurred during the first 2 weeks after marking and then subsided. Both deaths occurred after this period, one 28 days post-marking and the other 33 days post-marking. All birds in 2008 continued to nest and rear broods after transmitter attachment, with the exception of birds that lost their nests to flooding. In 2009, 6 hens died during the monitoring period (16, 18, 29, 31, 52, 80 days post-marking). Evidence obtained at the recovery site indicated that radioed birds were either depredated or scavenged by avian predators (3) or by mammalian predators (1). Additionally, there were 2 cases in which a probable cause of death could not be determined, because the transmitter was underwater and no carcass was found. All of the hens that died in 2009 did not have broods at the time of death; 3 lost their nest late in incubation, 1 abandoned her nest due to trapping, and 2 lost broods early after hatching. Of the 8 hens that did not die during the monitoring period: 2 abandoned their nest (1 likely due to trapping), 1 nest was depredated, and 4 hatched nests, with 1 hen fledging young. Hen survival through September was 0.73 for 2008 and 0.54 for 2009.

Brood Survival

In 2008, 7 radiomarked hens had broods and 1 additional hen, which we did not trap in time to give a transmitter, also had a brood (*n* = 8 broods, 57 ducklings). One brood survived to fledge 5 ducklings. Other broods dwindled slowly, with total brood loss at the IA (1), IB (1), IC (1), and IIA (2) stages (Gollop and Marshall 1954). The fate of 1 brood could not be determined, because the hen died when the brood was at the IIA stage, and we could no longer relocate the ducklings without the marked hen. Another brood made it to the IC stage, but we did not trap the hen in time to give her a transmitter, so their fate was uncertain.

Seven broods were monitored in 2009 (*n* = 56 ducklings). Total brood losses occurred at IA (3), IB (1), and IC (1) age classes. One brood fledged 2 young. Another brood matured to IIA before the hen left the wetland, after which time 1 duckling was seen on the wetland and no hens were present. Brood movements were also observed in 2009. For example, a hen moved her 3 (IC) young from the nesting wetland to another wetland (~1207 m) from which they fledged. In another instance, a hen and her brood of 6 (IB) were seen walking to another wetland ~364 m from their nesting wetland.

We also observed duckling adoption. In 2008, 1 hen lost her nest, but then was observed to be unambiguously associating with a brood of 4 IA ducklings. We saw 2 cases of creching. One brood of 4 at hatch was later seen as 8 at the IB stage. The other instance involved a female that hatched 7 young and was later seen with 9 young at the IB stage. Two females left their nesting wetland after they lost their broods and were seen with 1 or 2 other hens and a brood on another wetland. In these situations, the hens without the broods were alert and protective of the other hens' brood as if it was their own. Young and some females were not marked, so uncertainty existed with regard to the relationships among young and females.

DISCUSSION

Our success finding nests has been comparable to that in other studies of ring-necked ducks (45 nests in 3 years, Maxson and Riggs 1996; 35 nests in 2 years, Koons and Rotella 2003, 188 nests in 6 years by R. T. Eberhardt). Thus far, our results have been similar to findings by R. T. Eberhardt in northern Minnesota during 1978-1984 (Hohman and Eberhardt 1998). Our nest survival rates were within the range of Eberhardt's estimates of 44% (range 17-88%) based on 188 nests. The causes of nest failure in our study (24% flooding and 76%

depredation) were also similar to those of other studies (flooding 16-24%, depredation 67-80%, and desertion 5%, Mendall 1958, McAuley and Longcore 1989). Estimates of hatching success appear to be slightly lower than those of Eberhardt's previous study in north central Minnesota (94%, Hohman and Eberhardt 1998), but the springs and summers of 2008 and 2009 were very cool and rainy, which may have chilled eggs and flooded nests.

Our hen survival rates for the period June-September 2008 were lower than those reported for hen mallards during April-September (0.60, Blohm et al. 1987, and 0.67, Brasher et al. 2006, 0.80, Cowardin et al. 1985). Our brood survival rates also appear low. Brood survival in ring-necked ducks has only been examined previously in Maine (77% to 45 days, n = 64; McAuley and Longcore 1988). Duckling survival in the same study was 37%. Reliable estimation of brood survival was difficult in our study due to brood amalgamation and adoption. The degree to which these phenomena occur in ring-necked ducks is unknown. Creching has only been reported in ring-necked ducks once before by Toft et al. (1984) in the subartic taiga and was considerably lower (0.8% of broods) than in our study (20% of broods creched).

In 2010, we hope to collect additional data on nest success, hen survival, and brood survival. Results will be reported in future Summaries of Wildlife Research Findings.

ACKNOWLEDGMENTS

This research is primarily funded by the Upper Mississippi River and Great Lakes Joint Venture. The Minnesota Department of Natural Resources provides additional support for the project in the form of salary to C. Sousa and C. Roy, as well as fleet and transmitter expenses. Bemidji State University also contributes match in the form of salary to E. Rave. This work could not have been accomplished without the help of our interns S. Bischof, P. Christensen, R. Hanauer, T. Peterson, N. Besasie, E. Zlonis, J. Marty, D. Borhart, E. Smith, E. Sievers, and K. Kerfeld. We also thank all the landowners who provided us access to their properties to search for nests.

LITERATURE CITED

Blohm, R. J., R. E. Reynolds, J. P. Bladen, J. D. Nichols, J. E. Hines, K. H. Pollock, and R. T. Eberhardt. 1987. Mallard mortality rates on key breeding and wintering areas. Transactions of the North American Wildlife and Natural Resources Conference. 52:246-257.

Brasher, M. G., T. W. Arnold, J. H. Devries, and R. M. Kaminski. 2006. Breeding-season survival of male and female mallards in Canada's prairie-parklands. Journal of Wildlife Management 70:805-811.

Cowardin, L. M., D. S. Gilmer, and C. W. Shaiffer. 1985. Mallard recruitment in the agricultural environment of North Dakota. Wildlife Monographs 92:1-37.

Daubenmire, R. F. 1959. A canopy-coverage method of vegetational analysis. Northwest Science 33:43-64.

Gollop, J. B. and W. H. Marshall. 1954. A guide to aging duck broods in the field. Unpublished report. Mississippi Flyway Council.

Hohman, W. L., and R. T. Eberhardt. 1998. Ring-necked Duck (*Aythya collaris*). In The Birds of North America, No. 329 (A. Poole and F. Gill, eds.). The Birds of North America, Inc., Philadelphia, PA.

Iverson, L. R., and A. M. Prasad. 2001. Potential changes in tree species richness and forest community types following climate change. Ecosystems 4:186-199.

Kaplan, E. L., and P. Meier. 1958. Nonparametric estimation from incomplete observations. Journal of the American Statistical Association 53:457-481.

Mayfield, H. F. 1975. Suggestions for calculating nest success. The Wilson Bulletin 87(4):456-466.McAuley, D. G., and J. R. Longcore. 1989. Nesting phenology and success of ring-necked ducks in east-central Maine. Journal of Field Ornithology 60:112-119.

McAuley, D. G., and J. R. Longcore. 1988. Survival of juvenile ring-necked ducks on wetlands of different pH. Journal of Wildlife Management 52:169-176.

Mendall, H. L. 1958. Ring-necked duck in the Northeast. University of Maine, Orono.

Minnesota Department of Natural Resources. 2006. A Vision for Wildlife and its Use – Goals and Outcomes 2006 – 2012, Minnesota Department of Natural Resources, St. Paul, Minnesota, USA, http://files.dnr.state.mn.us/aboutdnr/reports/strategic-documents/wildlife-stategic-plan05-12.pdf Last accessed 17 February 2010.

Montgomery, J. 1985. A collar radio-transmitter attachment for wood ducks and other avian species. Proceedings of the International Conference on Wildlife Biotelemetry 5:19-27.

Robel, R. J., J. N. Briggs, A. D. Dayton, and L. C. Hulbert. 1970. Relationships between visual obstruction measurements and weight of grassland vegetation. Journal of Range Management 23:295-297.

Sorenson, M. D. 1989. Effects of neck collars on female redheads. Journal of Field Ornithology 60(4):523-528.

Toft, C. A., D. L. Trigger, and H.W. Murdy. 1984. Seasonal decline in brood sizes of sympatric waterfowl (*Anas* and *Aythya*, Anatidae) and proposed evolutionary explanation. Journal of Animal Ecology 53:75-92.

Weller, M. W. 1956. A simple field candler for waterfowl eggs. Journal of Wildlife Management 20:111-113.

Weller, M. W. 1957. An automatic nest-trap for waterfowl. Journal of Wildlife Management 21:456-458.

Zicus, M. C., D. P. Rave, J. Fieberg, J.Giudice, and R. Wright. 2005. Minnesota's ring-necked ducks: a pilot breeding pair survey. Pages 137 – 158 *in* P. J. Wingate, R. O. Kimmel, J. S. Lawrence, and M. S. Lenarz, editors. Summaries of wildlife research findings 2004. Minnesota Department of Natural Resources, St. Paul, Minnesota, USA.

2009 RING-NECKED DUCK BREEDING PAIR SURVEY

Christine M. Sousa, David P. Rave, and Michael C. Zicus

SUMMARY OF FINDINGS

A pilot study was conducted in 2004 – 2006 to develop a survey for Minnesota's ring-necked duck (*Aythya collaris*) resident breeding population because little was known about the distribution and abundance of breeding ring-neck ducks in the state. We employed the survey design and methods developed during the pilot study (Zicus et al. 2008) to estimate the breeding population in 2007. In 2008 and 2009, we surveyed only 3 of 6 geographic strata and 2 of 4 habitat classes due to budget limitations. Helicopter-based counts in 2009 entailed 6 survey-crew days from 5 – 12 June totaling ~39 hrs of flight time. In 2009, the resident breeding population for the 3 geographic strata was estimated to be 11,000 indicated breeding pairs and 23,000 birds, which are similar to the 2008 estimate and the recalculated estimates for 2006 and 2007.

INTRODUCTION

Growing concern among biologists about the status of ring-necked ducks in Minnesota prompted the initiation of a pilot study to develop a breeding pair survey (Zicus et al. 2008). At the time, little was known about the breeding distribution and abundance of resident ring-necked ducks in Minnesota. Concerns were raised, in part, due to counts from 10 wetlands in the Bemidji area, which have shown a ~70% decline in ring-necked duck breeding pairs since 1969 (Zicus et al. 2004). Counts from this geographically limited survey suggest that the Minnesota population may be declining despite continental increases (U.S. Fish and Wildlife Service 2008). Additionally, the species was identified as a forest indicator because of its unique habitat associations (Minnesota Department of Natural Resources 2006). The importance of this species to Minnesota is also reflected in the number of ring-necked ducks harvested annually, often the 3rd most common duck taken by hunters (U.S. Fish and Wildlife Service, unpublished reports). The primary objectives of this survey have been to estimate breeding pair numbers and monitor population trends in northern Minnesota.

METHODS

Number of breeding pairs and population size within a stratified random sample of survey plots have been estimated using 2 stratification variables: (1) Ecological Classification System (ECS) sections; and (2) presumed nesting-cover availability (i.e., a surrogate for predicted breeding ring-necked duck density, Zicus et al. 2008). The pilot study and the first year of the operational survey (2007) were restricted to an area believed to be primary breeding range of ring-necked ducks for logistical efficiency (Zicus et al. 2008) and included 6 ECS sections (Figure 1). In 2008 and 2009, 3 of the ECS sections were dropped from the survey (Figure 1). Public Land Survey (PLS) sections (~2.6-km^2 plots, range = 1.2 – 3.0 km^2) were used as primary sampling units. The PLS sections at the periphery of the survey area that were <121 ha in size were removed from the sampling frame to reduce the probability of selecting these small plots.

We used the same habitat class definitions that were used for stratification in 2006 (Table 1; Zicus et al. 2008). Similar to 2006, in 2008 and 2009, a stratified sampling design was used to estimate breeding ducks in the best ring-necked duck habitat (habitat class 1 and 2 plots). The sampling frame consisted of 6 strata (i.e., 3 ECS sections x 2 habitat classes, Figure 1A), and we proportionally allocated 175 plots to the 6 strata. In previous surveys we also used a 2-stage simple random sampling design to estimate population size in the remainder of the survey area (habitat class 3 and 4 plots). Although habitat class 3 and 4 plots provided information on use by ring-necked ducks of what we consider to be poorer quality habitat, only 8.6% of the breeding pairs were found there in 2006 and 2007 (Rave et al. 2008). When survey funds were reduced, a decision was made to not survey these plots where few ducks were expected to be found. For each plot, location, date, and time were recorded as were all ring-necked ducks observed on study plots and their sex and social status (Zicus et al. 2008). We considered pairs, lone males, and males in flocks of 2–5 to indicate breeding pairs (IBP; J. Lawrence, MNDNR, personal communication). The resident breeding population in the survey area was considered to be twice the IBP plus the number of lone females, flocked females, mixed sex groups, and single-sex groups >5 birds. We used the R library survey (Lumley 2009, R Development Core Team 2009) to estimate IBP and resident breeding population totals for habitat class 1 and 2 plots in each ECS section and the entire survey area.

RESULTS

In 2009, plots were well distributed throughout the study area (Figure 1B). Most plots (104) were located in the Northern Minnesota Drift and Lake Plains section, while the fewest plots (20) were located in the Lake Agassiz, Aspen Parklands section (Table 2). The sampling rate was higher in the Lake Agassiz, Aspen Parklands section than the other 2 ECS sections (5.9% versus 1.4% and 1.5%; Table 2). We were unable to survey 1 of the 175 plots in the Northern Minnesota Drift and Lake Plains section due to access limitations for 1 plot at the National Guard's Camp Ripley in Little Falls, Minnesota.

The survey was conducted 5–12 June and entailed 6 survey-crew days totaling ~39 hrs of flight time. A total of 273 ring-necked ducks were observed in 57 (33%) of 174 plots (Table 3). By habitat type, birds were detected on 31 (34%) of habitat class 1 plots and 26 (31%) of habitat class 2 plots. Overall, counts on occupied plots ranged from 1 to 19 birds (median = 4 birds/plot). Numbers of IBP on occupied plots ranged from 0 to 8 (median = 3 IBP/plot). Numbers of birds on occupied plots ranged from 1 to 23 ducks (median = 6 breeding birds/plot). Of the birds observed, 61% were classified as pairs, 17% lone males, 16% flocked males, 5% mixed groups, and <1% lone females. Of IBP, 47% were classified as pairs, 27% lone males, and 25% flocked males (Figure 2). These IBP ratios suggest that survey timing was reasonably good for estimating the resident breeding population.

Estimated IBP in the survey area was 10,947 pairs (SE = 1,563; Table 4, Figure 3A). The estimated resident breeding population of ring-necked ducks in the survey area was 19,488 birds (SE = 3,240; Table 4, Figure 3B). Because of sampling frame changes in 2008 and 2009, estimates from 2006 and 2007 were re-calculated with a 3 ECS sampling frame. Data from 2004 and 2005 were not re-calculated, because habitat classifications have also changed since those surveys were conducted. Estimates (IBP and breeding population) from 2009 were similar to 2006 and slightly higher than 2007 and 2008 but were within the error of prior surveys. The resident breeding population ranged from a high of 7,064 pairs and 14,948 breeding birds in the Northern Minnesota Drift and Lake Plains section to a low of 436 pairs and 871 breeding birds and in the Lake Agassiz, Aspen Parklands section (Table 5).

DISCUSSION

A number of trade-offs were involved in reducing the survey scope. By limiting the survey to 3 ECS, we are no longer monitoring populations in northeastern Minnesota. Birds in the area dropped from the survey are at relatively low densities; however, this area is quite large and represented approximately 30% of the resident breeding population in Minnesota based on surveys conducted in 2004 – 2007 (Table 5). Although we lost information on distribution and abundance by dropping 3 ECS, we have gained precision for the area that was sampled with reduced standard errors for the estimates. We also dropped the habitat class 3 and 4 plots. Dropping these plots allows us to focus where greater numbers of birds should be located based on the presence of suitable nesting habitat. These plots represented 12% of the population estimates from 2006 and 2007 (Rave et al. 2008).

The resident breeding population appears to be relatively stable in the few years that this survey has been conducted, remaining between 18,000 and 23,000 breeding birds based on the estimates for the 3 ECS; however, many additional years are needed to detect population trends. Further, the survey was designed to estimate numbers of breeding ring-necked ducks and monitor population trends and as such is not optimized for detecting changes in the size of the resident population. Additionally, the survey is now focused on some of the best nesting habitat in the state. We do not know how this will affect our ability to monitor this resident breeding population.

The survey was also not designed explicitly to describe the distribution of resident breeding ring-necked ducks, but observations accumulated thus far have improved our knowledge of ring-necked duck distribution in the survey area (Figure 4). Most of the IBP and breeding population to date have been located along the north and northwest margin of the Northern Minnesota Drift and Lake Plains section. Another concentration of breeding ring-necked ducks is found at Agassiz National Wildlife Refuge in the center of the Lake Agassiz, Aspen Parklands section. From 2005 through 2008, very few ring-necked ducks have been observed along the southern margin of the study area, although there have been a number of survey plots in this area. In 2009, we did find a number of ring-necked ducks in the southern portion of the Minnesota and Northeast Iowa Morainal ECS (Figure 4). This survey is planned to continue in 2010.

ACKNOWLEDGMENTS

Brian Hargrave and Nancy Dietz provided the initial Minnesota Gap Analysis Program (MNGAP) data, and Dan Hertel supplied the Habitat and Population Evaluation Team (HAPET) data used to define the primary breeding range. We thank pilots John Heineman and Mike Trenholm for help with survey planning and for flying the survey. Shelly Sentyrz and Chris Scharenbroich created the navigation maps used during the survey. Frank Swendsen has served as observer for a portion of the plots. We also acknowledge the Red Lake, Nett Lake, and Bois Forte bands of the Ojibwe, National Guard personnel at Camp Ripley, and Steve Windels at Voyageurs National Park for allowing plots under their purview to be surveyed.

LITERATURE CITED

Giudice, J. H., J. R. Fieberg, M. C. Zicus, D. P. Rave, and R. G. Wright. 2010. Cost and precision functions for aerial quadrat surveys: A case study of ring-necked ducks in Minnesota. Journal of Wildlife Management 74(2):342 – 349.

Lumley, T. 2009. Survey analysis in R. <http://faculty.washington.edu/tlumley/survey/>. Accessed 13 April 2010.

Minnesota Department of Natural Resources. 2006. *A vision for wildlife and its use – goals and outcomes 2006 – 2012*. Minnesota Department of Natural Resources, St. Paul.

R Development Core Team. 2009. R: a language and environment for statistical computing. R Foundation for Statistical Computing, Vienna, Austria. http://www.R-project.org.

Rave, D. P., M. C. Zicus, J. R. Fieberg, J. H. Giudice, and R. G. Wright. 2008. Minnesota's ring-necked duck breeding pair survey. Pages 469 – 482 in M. W. DonCarlos, R. O. Kimmel, J. S. Lawrence, and M. S. Lenarz, editors. Summaries of wildlife research findings 2007. Minnesota Department of Natural Resources, St. Paul.

Smith, G. W. 1995. A critical review of the aerial and ground surveys of breeding waterfowl in North America. National Biological Service, Biological Science Report 5, Washington, D.C.

U. S. Fish and Wildlife Service. 2008. Waterfowl breeding population and habitat survey. U. S. Fish and Wildlife Service Division of Migratory Bird Management, Laurel, Maryland. http://mbdcapps.fws.gov/.

Zicus, M. C., R. T. Eberhardt, J. Dimatteo, and L. L. Johnson. 2004. Bemidji area ring-necked duck survey. Pages 169 – 183 *in* M. W. DonCarlos, R. O. Kimmel, J. S. Lawrence, and M. S. Lenarz, editors. Summaries of wildlife research findings 2003. Minnesota Department of Natural Resources, St. Paul.

Zicus, M. C., D. P. Rave, J. R. Fieberg, J. H. Giudice, and R. G. Wright. 2008. Distribution and abundance of Minnesota breeding ring-necked ducks Aythya collaris. Wildfowl 58:31 – 45.

Table 1. Habitat classes assigned to Public Land Survey section plots in the Minnesota ring-necked duck breeding pair survey area, June 2004 – 2009.

Habitat class	Definition[a] 2004	Definition[a] 2005 - 2008[b]	Percent of survey area 2004	2005	2006-2007	2008-2009
1	Plots with $\geq$ the median amount of MNGAP class 14 and/or 15 cover within 250 m of and adjacent to MNGAP class 12 cover (i.e., high pair potential).	Plots with $\geq$ the median amount of MNGAP class 10, 14, and/or 15 cover within 250 m of and adjacent to MNGAP class 12 and/or 13 cover (i.e., high pair potential).	15.3	24.5	21.5	70.7
2	Plots with < the median amount of MNGAP class 14 and/or 15 cover within 250 m of and adjacent to MNGAP class 12 cover (i.e., moderate pair potential).	Plots with < the median amount of MNGAP class 10, 14, and/or 15 cover within 250 m of and adjacent to class 12 and/or 13 cover (i.e., moderate pair potential).	15.3	24.5	21.5	29.3
3	Plots with no MNGAP class 14 and/or 15 cover that include MNGAP class 12 cover that is within 250 m of a shoreline (i.e., low pair potential).	Plots with no MNGAP class 10, 14, and/or 15 cover that include class 12 and/or 13 cover that is within 100 m of a shoreline (i.e., low pair potential).	25.2	7.7	13.5	0.0
4	Plots with no MNGAP class 14 and/or 15 cover and no MNGAP class 12 cover within 250 m of a shoreline (i.e., no pair potential).	Plots with no MNGAP class 10, 14, and/or 15 cover and no class 12 and/or 13 cover within 100 m of a shoreline (i.e., no pair potential).	44.2	43.3	43.5	0.0

[a]Plots are Public Land Survey sections. MNGAP = Minnesota GAP level 4 land cover data. Class 10 = lowlands with <10% tree crown cover and >33% cover of low-growing deciduous woody plants such as alders and willows. Class 12 = lakes, streams, and open-water wetlands. Class 13 = water bodies whose surface is covered by floating vegetation. Class 14 = wetlands with <10% tree crown cover that is dominated by emergent herbaceous vegetation such as fine-leaf sedges. Class 15 = wetlands with <10% tree crown cover that is dominated by emergent herbaceous vegetation such as broad-leaf sedges and/or cattails.
[b]Habitat class definitions in 2005 – 2009 were the same, but MNGAP class 10, 14, and 15 cover associated with lakes having a General or Recreational Development classification under the Minnesota Shoreland Zoning ordinance was not considered nesting cover in 2006 – 2009.

Table 2. Sampling rates in the habitat class 1 and 2 strata by Ecological Classification System (ECS) section for Minnesota's ring-necked duck breeding-pair survey, June 2004 – 2009.

ECS section	No. of plots[a]				No. of plots surveyed (Sampling rate [%])				
	2004	2005	2006-2007	2008-2009	2004	2005	2006-2007	2008	2009
W & S Superior Uplands[b]	1,638	2,461	2,218	-	18 (1.1)	22 (0.9)	20 (0.9)	-	-
Northern Superior Uplands	1,810	4,648	4,209	-	13 (0.7)	36 (0.8)	33 (0.8)	-	-
N Minnesota & Ontario Peatlands	1,817	2,737	2,389	-	26 (1.4)	35 (1.3)	30 (1.3)	-	-
N Minnesota Drift & Lake Plains	5,048	8,383	7,145	7,145	78 (1.5)	94 (1.1)	77 (1.1)	108 (1.5)	104 (1.5)
Minnesota & NE Iowa Morainal	3,510	4,033	3,561	3,561	50 (1.4)	35 (0.9)	32 (0.9)	53 (1.5)	51 (1.4)
Lake Agassiz, Aspen Parklands	316	363	340	340	15 (4.7)	8 (2.2)	8 (2.4)	13 (3.8)	20 (5.9)

[a]Number of Public Land Survey sections in the ECS section(s).
[b]Western and Southern Superior Uplands sections combined due to the small area of the Southern Superior Uplands occurring in the survey area.

Table 3. Survey results for habitat class 1 and 2 strata in the Minnesota ring-necked duck breeding pair survey area, June 2004 – 2009.

Year	No. of plots surveyed	No. plots with birds (%)	Birds[a]			IBP[b]			Resident breeding birds[c]		
			Total	Per plot	Per occupied plot	Total	Per plot	Per occupied plot	Total	Per plot	Per occupied plot
2004	200	50 (25)	278	1.39	5.56	160	0.81	3.20	353	1.77	7.06
2005	230	37 (16)	147	0.64	3.97	92	0.43	2.49	218	0.95	5.89
2006	200	50 (25)	279	1.40	5.58	167	0.85	3.34	375	1.88	7.50
2007	200	52 (26)	152	0.76	2.92	137	0.72	2.63	296	1.48	5.69
2008	174	58 (33)	296	1.70	5.10	173	0.99	2.98	364	2.09	6.28
2009	174	57 (33)	273	1.57	4.79	173	0.99	3.04	362	2.08	6.35

[a]Total number of ring-necked ducks counted during the survey.
[b]The number of indicated breeding pairs (IBP) is the sum of the pairs, lone males, and males in flocks of 2–5.
[c]The total resident breeding population in the survey area was considered to be twice the IBP plus the number of lone females, flocked females, mixed sex groups, and single-sex groups >5 birds.

Table 4. Estimated indicated breeding pairs (IBP) and resident breeding population size in the habitat class 1 and 2 strata in the Minnesota ring-necked duck breeding pair survey area, June 2004 – 2009.

	IBP (CV[%])		Resident breeding population (CV[%])	
Year	6 ECS[a]	3 ECS[b]	6 ECS[a]	3 ECS[b]
2004	9,443 (17.8[c])	-	20,321 (18.1[c])	-
2005	7,496 (20.0[c])	-	17,279 (21.5[c])	-
2006	14,770 (17.6[c])	9,851 (23.8)	32,621 (17.4[c])	21,849 (23.1)
2007	12,787 (17.7)	8,705 (19.9)	26,026 (17.5)	17,863 (19.5)
2008	-	9,439 (16.8)	-	19,488 (16.6)
2009	-	10,947 (14.3)	-	22,987 (15.0)

[a]Population estimates were based on a stratified random sample of habitat class 1 and 2 Public Land Survey (PLS) sections in 12 strata (2 habitat classes and 6 Ecological Classification System [ECS] sections).
[b] Population estimates were based on a stratified random sample of habitat class 1 and 2 Public Land Survey (PLS) sections in 6 strata (2 habitat classes and 3 Ecological Classification System [ECS] sections). Population estimates were not adjusted for 2004 and 2005, because the habitat classifications have also changed since those surveys were conducted.
[c]Variance estimate is biased low because no birds were observed in one or more strata. As a result, the confidence interval is too narrow and the CV is optimistic.

Table 5. Estimated indicated breeding pairs (IBP) and resident breeding population by Ecological Classification System (ECS) section in the habitat class 1 and 2 strata in the Minnesota ring-necked duck breeding pair survey area, June 2005 – 2009.

	IBP (CV [%])				
ECS section	2005	2006	2007	2008	2009
W & S Superior Uplands[b]	444 (99.5[c])	669 (59.1)	671 (99.6)	-	-
Northern Superior Uplands	1,169 (46.8)	2,679 (33.7)	2,694 (46.5)	-	-
N Minnesota & Ontario Peatlands	239 (54.1[c])	1,572 (34.7)	717 (46.5)	-	-
N Minnesota Drift & Lake Plains	3,490 (33.0)	6,334 (31.5)	5,686 (26.0)	4,948 (24.6)	7,064 (17.1)
Minnesota & NE Iowa Morainal	918 (43.6)	2,102 (53.9)	2,118 (38.8)	3,689 (26.0)	3,449 (28.4)
Lake Agassiz, Aspen Parklands	1,235 (40.1[c])	1,414 (35.2)	902 (40.9)	803 (38.4)	436 (35.5)

[a]Western and Southern Superior Uplands sections combined due to the small area of the Southern Superior Uplands occurring in the survey area.

Table 5. Continued.

ECS section	Resident breeding population (CV [%]) 2005	2006	2007	2008	2009
W & S Superior Uplands[b]	889 (99.5[c])	1,338 (59.1)	1,342 (99.6)	-	-
Northern Superior Uplands	2,339 (46.8)	5,357 (33.7)	5,388 (46.5)	-	-
N Minnesota & Ontario Peatlands	477 (54.1[c])	4,076 (42.3)	1,434 (46.5)	-	-
N Minnesota Drift & Lake Plains	6,981 (33.0)	14,816 (29.6)	11,651 (25.4)	10,264 (24.3)	14,948 (18.2)
Minnesota & NE Iowa Morainal	4,122 (56.4)	4,204 (53.9)	4,236 (38.8)	7,377 (26.0)	7,170 (29.2)
Lake Agassiz, Aspen Parklands	2,471 (40.1[c])	2,829 (35.2)	1,976 (42.3)	1,846 (41.4)	871 (35.4)

[a]Western and Southern Superior Uplands sections combined due to the small area of the Southern Superior Uplands occurring in the survey area.

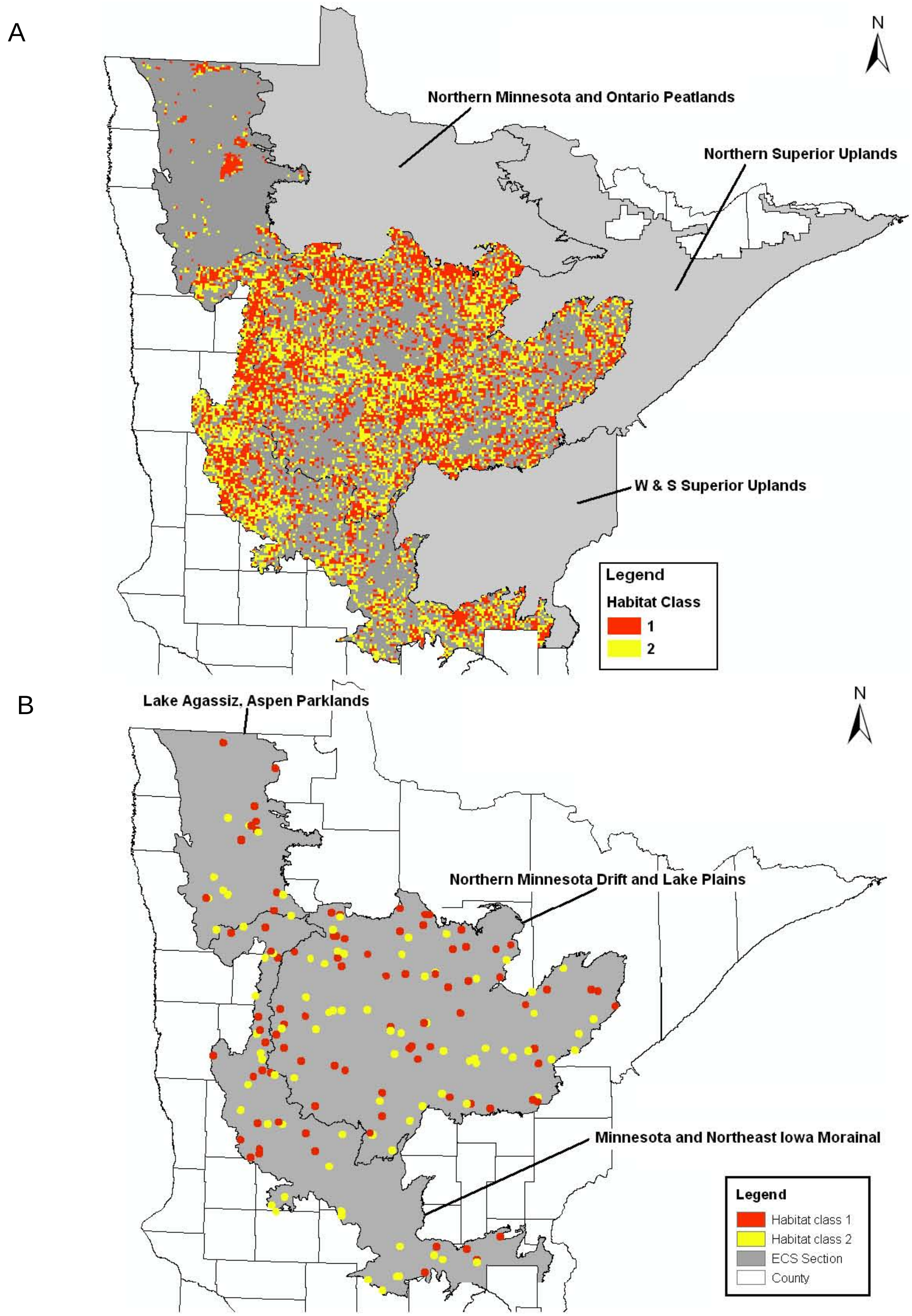


Figure 1. In the 3 Ecological Classification Section (ECS) sampling frame (A) all Public Land Survey (PLS) plots and (B) 2009 survey plots (enlarged for visibility) indicated by habitat class for Minnesota's ring-necked duck breeding pair survey.

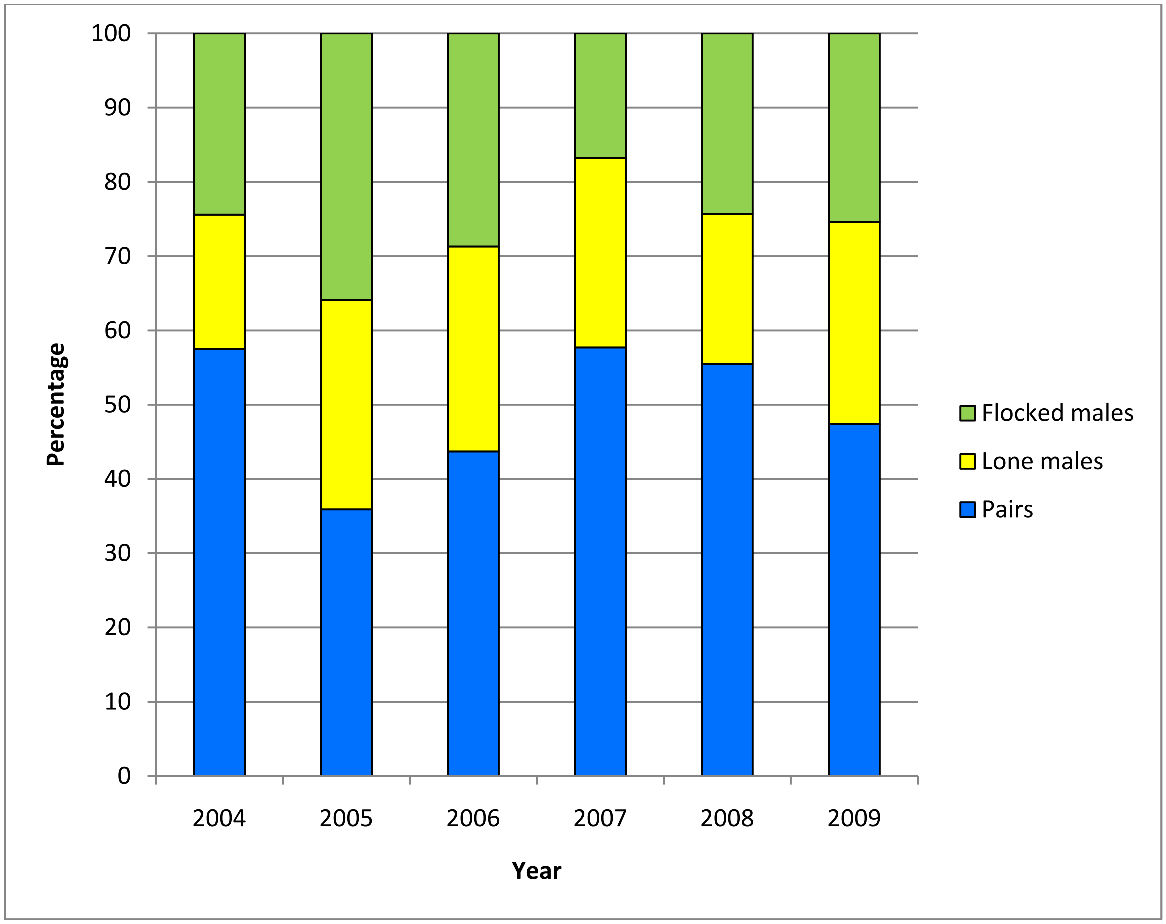
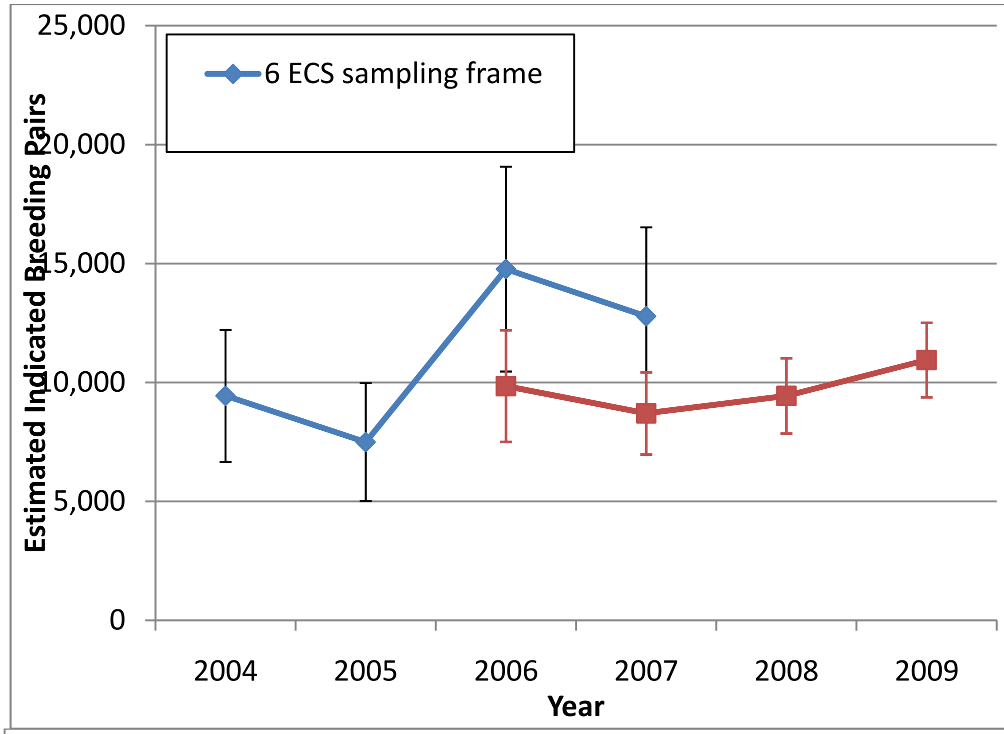


Figure 2. Social status of the indicated breeding pairs observed in the Minnesota ring-necked duck breeding pair survey area, June 2004 – 2009. Surveys were conducted 6 – 17 June 2004, 12 – 24 June 2005, 6 – 16 June 2006, 5 – 13 June 2006, 9 – 17 June 2008, and 5 – 12 June 2009.

A



B

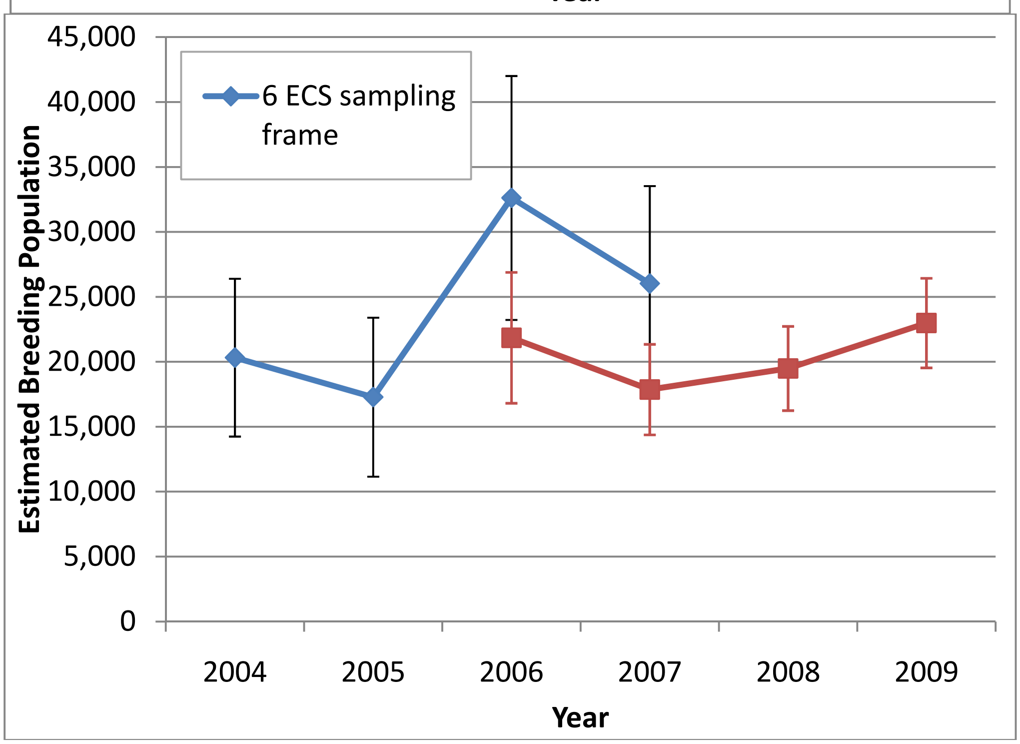


Figure 3. For the habitat class 1 and 2 strata (A) estimated indicated breeding pairs with SE bars and (B) estimated ring-necked duck resident breeding population with SE bars in the Minnesota ring-necked duck breeding pair survey area, June 2004 – 2009. Estimates were based on a stratified random sample of Public Land Survey (PLS) sections in habitat classes 1 and 2 for 6 Ecological Classification System (ECS) sections in 2004 – 2007 and for 3 ECS sections in 2008 and 2009. Estimates from 2006 and 2007 were recalculated using the same sampling frame as 2008 and 2009 (3 ECS instead of 6 ECS) for comparison; population estimates were not adjusted for 2004 and 2005, because the habitat classifications have also changed since those surveys were conducted.

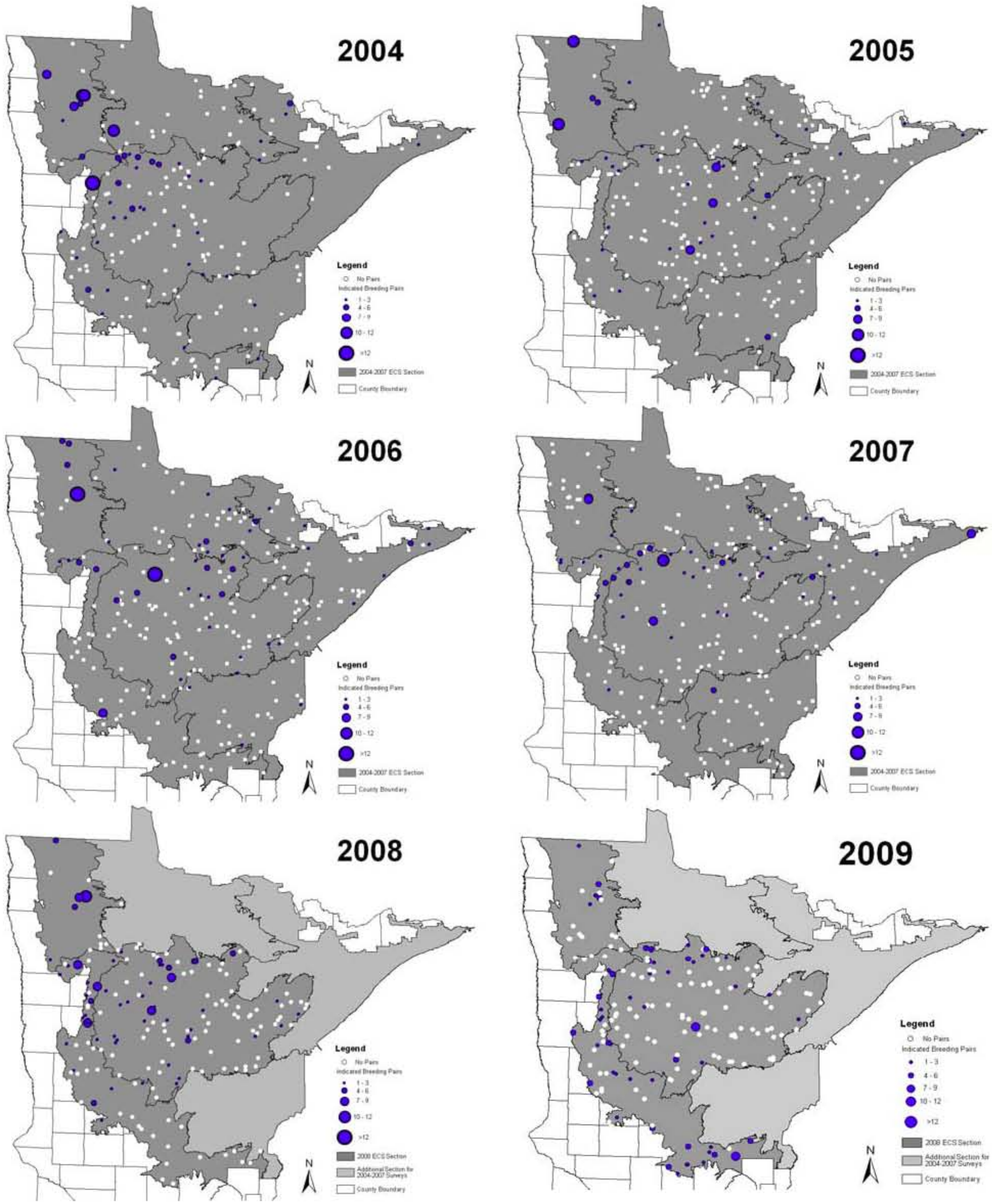


Figure 4. Plot locations and numbers of indicated breeding pairs (IBP) observed on survey plots in the Minnesota ring-necked duck breeding pair survey area in June 2009 (bottom right). White circles indicate plots where no indicated pairs were seen. Maximum number of indicated breeding pairs per plot was 8 pairs in 2009 (13 in 2004; 11 in 2005; 16 in 2006; 11 in 2007; 10 in 2008). The Ecological Classification System (ECS) sections are also shown.

Wildlife Disease Program
Carlos Avery Wildlife Management Area
5463-C West Broadway
Forest Lake, MN 55025
(651) 296-5200

SURVEILLANCE FOR HIGHLY PATHOGENIC AVIAN INFLUENZA IN MINNESOTA'S MIGRATORY BIRDS

Erik Hildebrand[1], Michelle Carstensen, and Erika Butler

SUMMARY OF FINDINGS

As part of a national strategy for early detection of highly pathogenic avian influenza (HPAI) in North America, the Minnesota Department of Natural Resources (MNDNR) and the United States Department of Agriculture (USDA) conducted surveillance for the virus in waterfowl in the state. A combined total of 1,409 birds were sampled for HPAI in Minnesota during 2009. Testing did not result in any positive cases of HPAI, however nearly 200 did test positive for a low pathogenic strain of avian influenza. Approximately 44,374 wild birds were sampled throughout the United States in 2009, and no positive cases of HPAI were detected. Minnesota will continue surveillance for the virus in the state's waterfowl in 2010, in cooperation with the Mississippi Flyway Council of the U.S. Fish and Wildlife Service and the USDA.

INTRODUCTION

Recent worldwide attention on the spread of a highly pathogenic strain of avian influenza, subtype H5N1, from Asia to Europe and Africa in 2006 has led to the development of a coordinated National Strategic Plan for early detection of HPAI-H5N1 introduction into North America by wild birds. Although movements of domestic poultry or contaminated poultry products, both legally and illegally, are believed to be the major driving force in the spread of HPAI-H5N1, migratory birds are thought to be a contributing factor.

Avian Influenza is a viral infection that occurs naturally in wild birds, especially waterfowl, gulls, and shorebirds. It is caused by type A influenza viruses that have 2 important surface antigens, hemagglutinin (H) and nuraminidase (N), that give rise to 144 possible virus subtypes. Influenza viruses vary widely in pathogenicity and ability to spread among birds. The emergence of an Asian strain HPAI-H5N1 virus in 1996 and subsequent spread of the virus in Asia, Africa, and Europe has killed thousands of wild birds and millions of domestic poultry. In 1997, HPAI-H5N1 became zoonotic in Hong Kong and to-date has infected at least 496 humans in Eurasia and Africa, resulting in over 293 deaths.

The National Plan outlined a surveillance strategy that focused on sampling of wild bird species in North America that have the highest risk of being exposed to or infected with HPAI-H5N1 because of their migratory movement patterns. Currently, these include birds that migrate directly between Asia and North America, birds that may be in contact with species from areas in Asia with reported outbreaks, or birds that are known to be reservoirs of AI. A step-down plan was developed by the Mississippi Flyway Council in 2006 identifying Minnesota as a key flyway state needed to participate in regional sampling for early detection of HPAI-H5N1 in migratory ducks, geese, and shorebirds.

In July 2009, the MNDNR entered into a $70,000 cooperative agreement with the United States Department of Agriculture's Wildlife Services (USDA-WS) to sample 600 wild birds (either live-caught or hunter-harvested) in Minnesota for HPAI-H5N1 during 2009. In addition to the 600 samples to be collected by MNDNR, USDA-WS was also planning to collect a similar number of samples in the state during the same period. Bird species that were targeted include those listed as priority species in the National Strategic Plan or approved for sampling in Minnesota by the Mississippi Flyway Council. There have been surveillance efforts for the past 4 years with nearly 6,600 samples from MN, submitted for HPAI-H5N1 testing.

[1] Corresponding author e-mail: erik.hildebrand@state.mn.us

METHODS

The MNDNR planned to sample 50 common goldeneye (*Bucephala clangula),* 50 ring-neck ducks (*Aythya collaris),* 50 mallards (*Anas platyrhynchos),* and 30 blue-winged teal (*Anas discors)* during the summer months, primarily in conjunction with planned banding activities. In the fall, through hunter-harvested surveillance, sampling targets were as follows: 80 Northern pintails (*Anas acuta)*, 80 mallards, 80 American green-winged teal (*anas crecca*), 80 American blue-winged teal (*Anas discors*), 50 Northern shovelers (*Anas clypeata*), and 50 American wigeon (*Anas Americana*). USDA-WS planned to sample a similar number of the duck species mentioned above or others from their functional group (e.g., dabblers, divers, shorebirds), as well as 50 Canada geese. If sampling goals per species could not be met, other targeted waterfowl species within the same functional group could be sampled and counted toward the state's total. Sampling strategies were coordinated between the MNDNR and USDA-WS to maximize access to targeted birds species through existing banding operations and fall hunter-harvested surveillance.

Cloacal and oral-pharyngeal swabs were used to collect samples and they were submitted to the Veterinary Diagnostic Laboratory in St. Paul, MN for initial screening for the virus. If positive for avian influenza virus, samples were forwarded to the National Veterinary Services Laboratories in Ames, IA for strain-typing.

RESULTS AND DISCUSSION

From April 1, 2009 through March 31, 2010 MNDNR and USDA collected a total of 1,409 samples from wild-caught live birds (*n*=310), hunter-harvested birds (*n*=1,016), agency (USDA-WS) harvested (*n*=73), and mortality/morbidity events (*n*=10) (Figure 1).

Testing did not result in any positive cases of HPAI-H5N1; however 8 different duck species tested positive for a low pathogenic strain of avian influenza with the subtype H5, and only 1 tested positive for a N1 subtype. The testing protocol was limited to the screening for H5, H7, and N1 subtypes only; however in some cases other subtypes were identified and reported elsewhere (Table 1, Figure 2).

According to the latest numbers of the United States Geologic Survey's website (http://wildlifedisease.nbii.gov/ai/), approximately 44,374 birds have been sampled for HPAI-H5N1 in the U.S. in 2009. No positive cases of HPAI-H5N1 have been found anywhere in North America to date. Since the majority of H5 positives (low pathogenic forms only) detected by USDA-WS in the United States since 2006 have been found in dabbling ducks, the primary focus of future sampling will be on these species (Genus *Anas, Aix, Cairina,* and *Dendrocygna*). Surveillance for HPAI-H5N1 will likely continue in Minnesota and other parts of the U.S. next year. The USDA has banked all samples taken from 2006 to 2009, and is currently accepting proposals from state agencies and universities for further avian influenza research. Minnesota remains prepared to assist with future surveillance objectives if needed. In addition, the MNDNR has developed a surveillance and response plan for HPAI in wild birds, which includes increased vigilance of mortality and morbidity events within the state.

ACKNOWLEDGEMENTS

This project would not have been possible without the valuable contribution of the waterfowl research group, including Jeff Lawrence, Steve Cordts, Jim Berdeen, and Jim's group of banding interns. Other MNDNR staff that provided valuable assistance to this project included Joel Huener, Dawn Torrison, Randy Pracher, Perry Loegering, Jeff Hines, Joel Anderson, Dave Trauba, Kevin Kotts, Martha Minchak, Dan Rhode, Bob Welsh, Bryan Lueth, David Pauly, Judy Markl, Jon Cole, Shelly Gorham, and Blane Klemek. I would also like to recognize our USDA-WS partner on the project, Paul Wolf, for his efforts to ensure that we met our overall sampling goals. Lastly, much of the hunter-harvested sampling was accomplished

through assistance from Pat Reddig, University of Minnesota, and numerous students from both the Natural Resources program and the veterinary college.

REFERENCES

Halvorson, D. A., C. J. Kelleher, and D.A. Senne. 1985. Epizootiology of avian influenza: effect of season on incidence in sentinel ducks and domestic turkeys in Minnesota. Applied and Environmental Microbiology 49: 914-919.

Hanson, B. A., D. E. Stallknecht, D.E. Swayne, L. A. Lewis, and D. A. Senne. 2003. Avian influenza viruses in Minnesota ducks during 1998-2000. Avian Diseases 47: 867-871.

Interagency Asian H5N1 Early Detection Working Group. 2006. An early detection system for Asian H5N1 highly pathogenic avian influenza in wild migratory birds: U.S. Interagency Strategic Plan. Unpubl. Rept. Report to the Department of Homeland Security, Policy Coordinating Committee for Pandemic Influenza Preparedness.

Michigan Department of Natural Resources, Wildlife Division. 2006. Michigan surveillance and response plan for highly pathogenic avian influenza in free-ranging wildlife. Unpubl. Rept.

Mississippi Flyway Council. 2006. Surveillance for early detection of highly pathogenic avian influenza H5N1 in wild migratory birds: a strategy for the Mississippi Flyway. Unpubl. Rept.

Table 1. Bird species sampled for highly pathogenic avian influenza H5N1 by Minnesota Department of Natural Resources and United States Department of Agriculture-Wildlife Services in 2009. Table includes live wild birds, hunter harvested, agency harvested, and morbidity/mortality.

SPECIES SAMPLED		***n***
Ducks		
	American Coot	3
	American Green-Winged Teal	106
	American Widgeon	56
	American Blue-Winged Teal	180
	Bufflehead	23
	Canvasback	8
	Common Goldeneye	53
	Common Merganser	1
	Gadwall	32
	Greater Scaup	2
	Hooded Merganser	16
	Lesser Scaup	45
	Mallard	231
	Northern Pintail	64
	Northern Shoveler	40
	Redhead	51
	Ring-Necked Duck	200
	Ruddy Duck	2
	Woodduck	166
	Canada Geese	95
Other		
	American Golden-Plover	1
	American White Pelican	20
	Double Crested Cormorant	3
	Greater Yellowlegs	1
	Ring-Billed Gull	10
Total		1,409

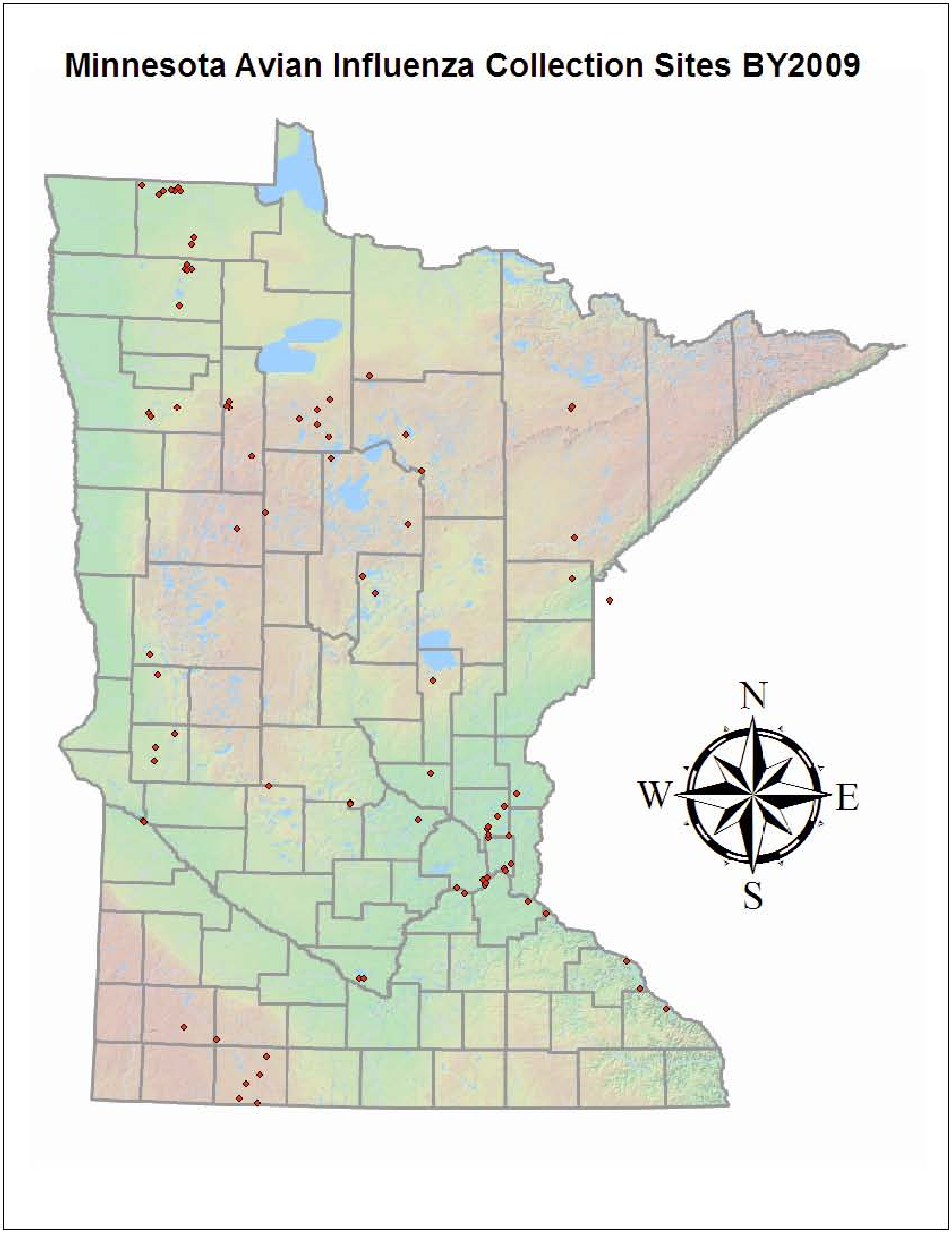


Figure 1. Collection sites from which live bird samples (*n*-1,409) were tested for highly pathogenic avian influenza in Minnesota during 2009.

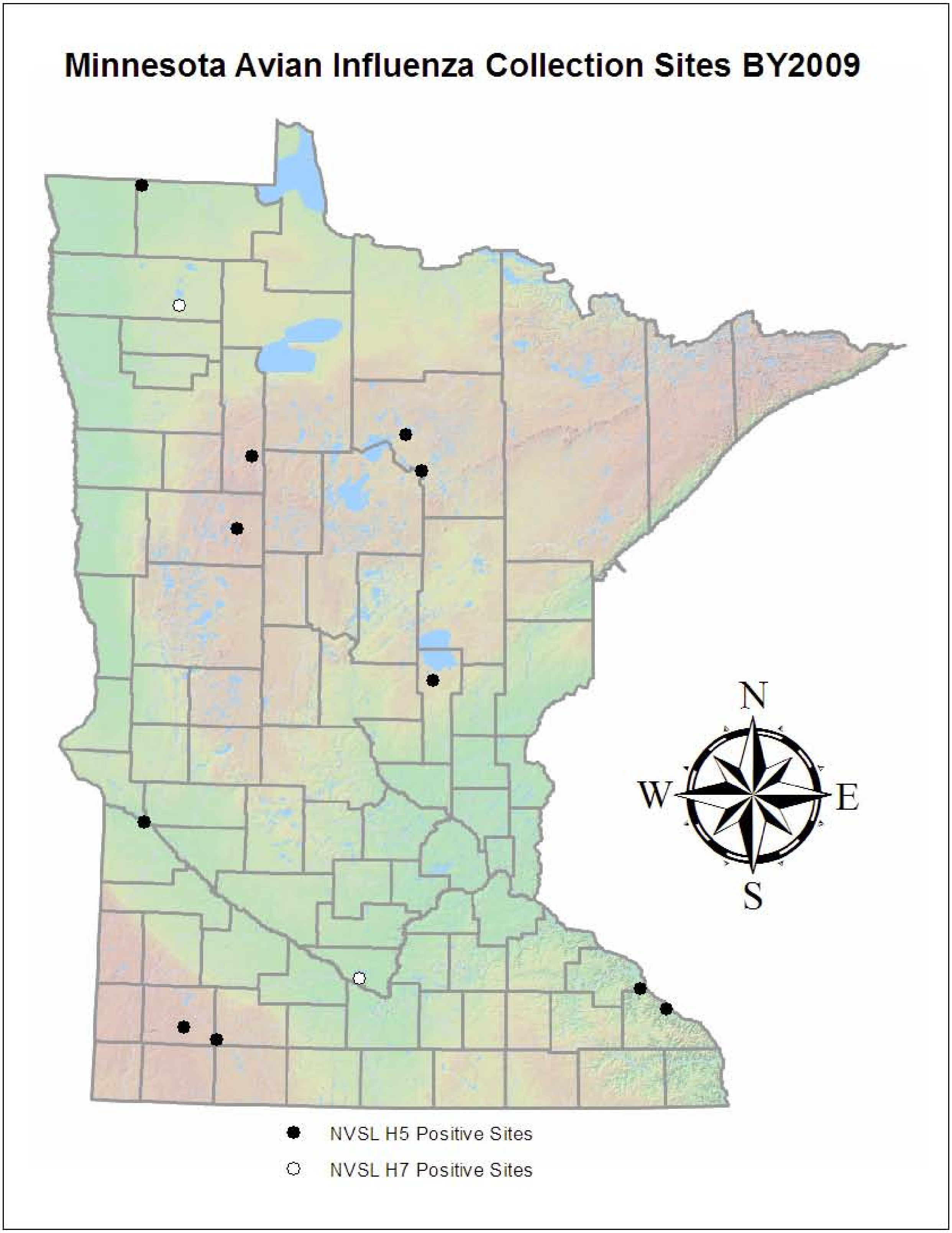


Figure 2. Collection sites where a low pathogenic H5 strain (black dots) and H7 strain (white dots) were detected among the waterfowl (*n*-199) sampled in Minnesota during 2009.

MINNESOTA DEPARTMENT OF NATURAL RESOURCES CWD SURVEILLANCE PROGRAM 2009

Michelle Carstensen[1], David Pauly, Erika Butler, Erik Hildebrand, and Lou Cornicelli

SUMMARY OF FINDINGS

In fall 2009, the Minnesota Department of Natural Resources (MNDNR) sampled 2,685 hunter-harvested white-tailed deer (*Odocoileus virginianus*) for chronic wasting disease (CWD) in southeastern Minnesota. The surveillance effort was initiated primarily on the discovery of a CWD-positive captive elk facility in Olmsted county, and secondarily to monitor the ongoing risk of disease spread from CWD-infected wild deer from Wisconsin. All of the samples were negative for CWD. In addition, MNDNR submitted samples from 28 deer through statewide targeted surveillance, which included sick animals, escaped captive cervids, and roadkills; these samples were also negative for the disease. MNDNR plans to conduct hunter-harvested surveillance in southeastern MN in fall 2010, with efforts limited to a 15-mile radius around the CWD-infected captive elk facility in Olmsted county.

INTRODUCTION

Chronic wasting disease is a transmissible spongiform encephalopathy (TSE) that affects elk (*Cervus elaphus*), mule deer (*Odocoileus hemionus*), white-tailed deer, and moose (*Alces alces*). TSEs are infectious diseases that alter the morphology of the central nervous system, resulting in a “sponge-like” appearance of this tissue. The etiological agent of CWD is an infectious protein, called a prion. Precise mechanisms and rates of CWD transmission remain unclear, although recent studies support animal-to-animal contact and environmental contamination as mechanisms that promote the spread of the disease. For example, one recent study has proven prions are shed in feces of infected deer 7-9 months before the onset of clinical signs, further supporting the high rate of horizontal transmission in infected populations. Incubation time of the disease, from infection to clinical signs, averages 16 months but can range from a few months to nearly 3 years. There is a limited distribution of infection in the body (primarily brain, spinal column, spleen, and lymph nodes) although a recent study demonstrated that prions can also be found in muscle. Clinical signs may include a loss of body condition and weight, excessive salivation, ataxia, and behavioral changes. Currently, there is no known treatment for the disease and it is always fatal. There is also no documented evidence of transmission of CWD to other species, including humans.

To date, CWD has been diagnosed in 3 captive elk herds and 1 captive white-tailed deer herd within the state of Minnesota. Two of the elk herds (Stearns and Aitkin counties) were discovered in 2002 and depopulated; no additional CWD positive animals were found. In spring 2006, a captive white-tailed deer was found infected with CWD from a mixed deer/elk herd in Lac Qui Parle county. That herd was also depopulated without additional infection being detected. In all of these cases, the original source of the CWD has not been identified. In early 2009, a third captive elk herd (Olmsted county) was found infected with CWD. An 8-year old female was found CWD-positive at slaughter and records indicated she was born on the farm in 2001, thus suggesting that she was exposed to another CWD positive animal(s) on the farm. This herd was indemnified by the United States Department of Agriculture (USDA) and 558 adult elk were depopulated in September. An additional 3 adult elk were found infected with CWD (2 females, 1 male). Further, a management plan as enacted following the elk herd depopulation to further prevent a spillover of CWD to wild deer outside this facility. This plan included the cleaning and disinfecting of all livestock barns and equipment, maintenance of perimeter fencing, and a ban of captive cervids being restocked for 5 years. Since the property of the former elk facility has been sold and plans for development of a

[1]Corresponding author e-mail: michelle.carstensen@state.mn.us

biomedical research park are in place, MNDNR has been concerned about environmental contamination of prions should fencing been removed. Thus, the management plan included the requirement of the top 2-inches of topsoil to be removed and stored behind 96-inch fencing prior to the initiation of any land development projects. MNDNR and BAH are working cooperatively to address the impact of CWD in these captive facilities, as well as management options to control its spread.

Over the past 8 years, MNDNR has tested in excess of 33,000 deer across the state for CWD, all of which have been negative. Consequently, in recent years, sampling has been scaled back to address 3 main components:

1. Sampling of animals exhibiting symptoms of CWD (targeted surveillance);
2. Sampling of animals in response to elevated risk factors (e.g., detection of positive animals in captive cervid farms, or proximity of Minnesota to positive CWD cases in other states); and
3. Sampling of hunter-killed deer for CWD in conjunction with surveillance for bovine tuberculosis.

METHODS

Hunter-harvested surveillance occurs at deer registration stations during the regular firearm hunting season. Stations are staffed with MNDNR personnel and students (veterinary medicine and natural resources) that were trained in lymph node extraction. Hunters were asked to voluntarily submit retropharyngeal lymph node samples for CWD testing. Samples were submitted to the Veterinary Diagnostic Laboratory at the University of Minnesota for disease screening. Any presumptive positive samples would be submitted to the National Veterinary Services Laboratories (Ames, IA) for official confirmation of the disease. Hunter information was recorded, including the hunter's name, address, telephone number, MNDNR number, and location of kill. Maps were provided to assist the hunters in identifying the location (Township, Range, and Section) of the kill. Cooperating hunters were given a cooperator's patch and entered into a raffle to win a firearm donated by the Minnesota Deer Hunter's Association.

During fall 2009, registration stations were selected based on deer volume and distribution through the surveillance zone to meet a sampling goal of 300 deer per sampling block (n = 10), or an overall sampling goal of 3,000 samples (Figure 1). Registration stations were also selected based on their proximity to the CWD-positive captive elk facility and along the MN-WI border, to maximize our sampling of deer from those high-risk areas.

MNDNR continues to sample deer exhibiting clinical symptoms consistent with CWD (targeted surveillance) statewide. Information has been disseminated to wildlife staff regarding what to look for regarding symptomatic deer. Staff were provided the necessary equipment and training for lymph node removal and data recording. The number of samples expected through targeted surveillance is estimated to be less than 100 animals annually, as few reports of sick deer are taken.

RESULTS AND DISCUSSION

From June 2009 to April 2010, MNDNR collected a total of 28 samples from targeted surveillance efforts. This includes samples from 11 escaped captive cervids, 14 free-ranging sick deer and 3 wild deer removed from within the perimeter fence of the CWD-positive elk facility; all samples were negative for CWD.

MNDNR collected a total of 2,685 samples from hunter-harvested deer for CWD screening during fall 2009 (Figure 2). All samples were also negative for CWD. The sampling distribution of 300 samples collected per block was met in 50% of the sampling units (Figure 2, Tables 1,2). Even though the sampling goal fell short in half of the sampling units, we achieved 88% of our overall surveillance goal of 3,000 samples. Further, a high proportion of samples were obtained within a 15-mile radius of the CWD-positive captive elk facility (Figure 3), as well as along the WI-MN border, where the risks of CWD in existing in wild deer are the highest.

Since the agency has now collected in excess of 33,000 negative samples in statewide surveillance efforts, we feel that future resources for CWD surveillance, in addition to targeted surveillance, are better spent addressing changing risk factors. Specifically, it is important to monitor the CWD surveillance activities occurring in our bordering states, and conduct periodic surveillance in Minnesota in response to CWD status changes in these states. Additionally, periodic surveillance in the vicinity of previous cases of CWD in captive cervids in Minnesota may be prudent. Given the most recent case of a CWD-infected cervid farm in Olmsted county, MNDNR plans to repeat surveillance efforts within a 15-mile radius of that farm during the fall 2010 firearm hunting season. Targeted surveillance of suspect deer is expected to continue throughout the state.

SURVEILLANCE COSTS

Conducting a disease surveillance effort that spans a large area and encompasses numerous deer registration stations requires a large, trained work force and a significant amount of expenditures to support the effort. The CWD surveillance effort in fall 2009 spanned the entire regular firearms season, requiring 4 consecutive weekends of staffed registration stations to obtain samples. The number of stations staffed each weekend ranged from 21 to 29, and summed to 102 stations over the duration of the project. In total, 116 trained MNDNR staff and 113 student workers were needed in the effort. Costs associated to the surveillance effort are listed in Table 3.

ACKNOWLEDGEMENTS

We would especially like to recognize the tremendous amount of work and commitment by Dave Pauly, whom stepped into the role of CWD Coordinator to assist the Wildlife Health Program in this massive surveillance effort. We would like to thank the all the MNDNR staff, students, and faculty from the University of Minnesota, Colleges of Veterinary Medicine and Natural Resources, for assisting in our sampling efforts. Also, a special thanks to Julie Adams and Bob Wright for making our surveillance maps.

REFERENCES

Angers, R. C., Browing, S. R., Seward, T. S., Sigurdson, C. J., Miller, M. W., Hoover, E. A., and G.C.Telling. 2006. Prions in skeletal muscle of deer with chronic wasting disease. Science 311 (5764) :1117.

Baeten, L. A, Powers, B. E., Jewell, J. E., Spraker, T. R., and M. W. Miller. 2007. A natural case of chronic wasting disease in free-ranging moose (*Alces alces shirasi*). Journal of Wildlife Diseases 43(2): 309–318.

Diefenbach, D. R., C. S. Rosenberry, and R. C. Boyd. 2004. From the Field: Efficacy of detecting Chronic Wasting Disease via sampling hunter-killed white-tailed deer. Wildlife Society Bulletin 32: 267–272.

Miller, M. W., H. M. Swanson, L. L. Wolfe, F. G. Quartarone, S. L. Huwer, C. H. Southwick, and P. M. Lukacs. 2008. Lions and prions and deer demise. PLoS One 3(12): e4019.

Miller, M.W., E.S. Williams, N.T. Hobbs, and L.L. Wolfe. 2004. Environmental sources of prion transmission in mule deer. Emerg. Infec. Dis. 10(6): 1003–1006.

Miller, M.W., E. S. Williams, C. W. McCarty, T. R. Spraker, T. J. Kreeger, C. T. Larsen, and E. T. Thorne. 2000. Epizootiology of chronic wasting disease in free-ranging cervids in Colorado and Wyoming. Journal of Wildlife Diseases 36: 676–690.

Spraker, T. R., M. W. Miller, E. S. Williams, D. M. Getzy, W. J. Adrian, G. G. Schoonveld, R. A. Spowart, K. I. O'Rourke, J. M. Miller, and P. A. Merz. 1997. Spongiform encephalopathy in free-ranging mule deer (*Odocoileus hemionus*), white-tailed deer

(*Odocoileus virginianus*) and Rocky Mountain elk (*Cervus elaphus nelsoni*) in northcentral Colorado. Journal of Wildlife Diseases 33: 1–6.

Tamguney, Gultekin, M.W. Miller, L. L. Wolfe, T. M. Sirochmann, D. V. Glidden, C. Palmer. A. Lemus, S. J. DeArmond, and S. B. Prusiner. 2009. Asymptomatic deer excrete infectious prions in feces. Nature 461: 529–532.

Williams, E. S., and S. Young. 1980. Chronic Wasting Disease of captive mule deer: a spongiform encephalopathy. Journal of Wildlife Diseases 16: 89–98.

Williams, E.S., and S. Young. 1993. Neuropathology of chronic wasting disease of mule deer (*Odocoileus hemionus*) and elk (*Cervus elaphus nelsoni*). Veterinary Pathology 30: 36–45.

Williams, E.S., M. W. Miller, and E.T. Thorne. 2002. Chronic wasting disease: implications and challenges for wildlife managers. Presented at the North American Wildlife and Natural Resources Conference, April 3–7, 2002, Dallas, Texas.

Table 1. Breakdown of hunter-harvested deer samples collected for chronic wasting disease surveillance in southeastern Minnesota by registration station and sampling block, fall 2009.

Samples Collected By Check Station and Surveillance Block																
	Surveillance Block											Non-Surveillance Blocks				Grand
Check Station	233_293	341	342	343	344	345	346	347	348	349	Totals	255	339	601	Totals	Totals
Brownsville - Bissen's Tavern	0	0	0	0	0	0	2	0	0	84	86	0	0	0	0	86
Caledonia - Caledonia True Value	0	0	0	0	0	0	2	0	0	30	32	0	0	0	0	32
Cannon Falls - Curt's Cannonball	41	38	1	1	0	0	0	0	0	0	81	0	15	0	15	96
Chatfield - Magnum Sports	0	0	0	77	2	11	0	89	99	5	283	1	0	0	1	284
Elba - Elba Valley Express	0	0	2	0	20	1	0	0	0	0	23	0	0	0	0	23
Elba - Mauer Brother's Tavern	0	1	1	4	76	2	0	0	0	0	84	0	0	0	0	84
Frontenac - B-Wells	0	11	0	0	0	0	0	0	0	0	11	0	0	0	0	11
Frontenac - Frontenac SP	0	29	0	0	0	0	0	0	0	0	29	0	0	0	0	29
Houston - Houston Amoco	0	0	0	0	0	1	87	0	0	149	237	0	0	0	0	237
Kellogg - Prairie Bait Shop	0	0	98	0	15	2	0	0	0	0	115	0	0	0	0	115
Kenyon - Kenyon Meats	48	6	2	3	0	0	0	0	1	0	60	0	1	0	1	61
LaCrescent - Pump for Less	0	0	0	0	0	0	15	0	0	10	25	0	0	0	0	25
LaCrescent - Tri State Bait	0	0	0	0	0	0	23	0	0	3	26	0	0	0	0	26
Lake City - Big Bear Get-N-Go	0	28	68	0	3	0	0	0	0	6	105	0	0	0	0	105
Lanesboro - S & A Petroleum	0	0	0	0	0	0	1	3	94	3	101	0	0	0	0	101
Mantorville - Mantor Mart	17	4	0	44	2	0	1	0	0	0	68	0	0	0	0	68
Millville - Becklund A R	0	11	39	3	0	0	0	0	0	0	53	0	0	0	0	53
Nerstrand - Big Woods SP	15	0	0	0	0	0	0	0	0	0	15	0	0	0	0	15
Pine Island - Greenway Cooperative	1	27	2	51	0	0	0	0	1	0	82	0	0	0	0	82
Plainview - Kreofsky Building Supply	0	0	8	10	9	0	0	0	0	0	27	0	0	0	0	27
Preston - Skeets Deer Processing*	0	0	0	0	0	0	0	5	2	0	7	0	0	0	0	7
Red Wing - 4 Season's Sports Shop	0	118	0	0	1	1	0	0	0	0	120	0	11	0	11	131
Rochester - Archery Headquarters	0	2	2	36	0	0	0	0	0	0	40	0	0	0	0	40
Rochester - Gander Mountain	0	21	10	127	4	1	1	2	1	0	167	3	0	1	4	171
Rushford - Kwik Trip	0	0	0	0	0	1	0	0	5	7	13	0	0	0	0	13
Rushford - Pam's Corner Convenience	0	0	0	0	0	49	35	6	92	72	254	0	0	0	0	254
Spring Grove - Solie Services	0	0	0	0	0	0	0	0	0	53	53	0	0	0	0	53
Spring Valley - S & S Bait	0	0	0	0	0	0	0	28	0	1	29	0	0	0	0	29
St. Charles - Good Sport Liquor	0	2	1	18	10	43	2	0	5	1	82	1	0	0	1	83
Winona - Ledebuhr Meat Processing	0	0	4	1	10	39	32	1	2	7	96	0	0	1	1	97
Winona - Witoka Tavern	0	0	0	0	0	3	97	0	0	0	100	0	0	0	0	100
Wykoff - Wykoff Short Stop	0	0	0	0	0	0	0	61	6	1	68	2	0	0	2	70
Zumbro Falls - Falls Standard	0	49	20	4	0	0	0	0	0	0	73	0	0	0	0	73
Totals	122	347	258	379	152	154	298	195	308	432	2645	7	27	2	36	2681
Percent Towards Goal**	40.7%	115.7%	86.0%	126.3%	50.7%	51.3%	99.3%	65.0%	102.7%	144.0%	88.2%	Not Applicable				NA
	Does not include 4 samples with no location information.															
	**Goal: 3,000 samples (300/surveillance block).															

Table 2. Breakdown of hunter-harvested deer samples collected for chronic wasting disease surveillance in southeastern Minnesota by location and sampling weekend, fall 2009.

CWD Surveillance of Deer in Southeast Minnesota - November 2009

Collection of Retropharyngeal Lymph Nodes at Deer Check Stations

Samples Collected by Station Location

Location	Weekend 1	Weekend 2	Weekend 3	Weekend 4	Totals	Non-Surveillance	Grand Totals
Brownsville	46	15	12	13	86	0	86
Caledonia	18	2	12	0	32	0	32
Cannon Falls	57	18	6	0	81	15	96
Chatfield	147	26	80	30	283	1	284
Elba	37	6	44	20	107	0	107
Frontenac	0	11	29	0	40	0	40
Houston	83	40	85	29	237	0	237
Kellogg	60	12	29	14	115	0	115
Kenyon	48	10	2	0	60	1	61
LaCrescent	26	13	10	2	51	0	51
Lake City	42	22	29	12	105	0	105
Lanesboro	43	13	29	16	101	0	101
Mantorville	43	6	14	5	68	0	68
Millville	0	4	30	19	53	0	53
Nerstrand	0	0	0	15	15	0	15
Pine Island	38	8	20	16	82	0	82
Plainview	18	9	0	0	27	0	27
Preston*	0	5	6	0	11	0	11
Red Wing	68	24	29	0	121	10	131
Rochester	103	38	38	27	206	5	211
Rushford	104	38	83	42	267	0	267
Spring Grove	28	10	15	0	53	0	53
Spring Valley	0	0	18	11	29	0	29
St. Charles	40	7	18	17	82	1	83
Winona	94	22	62	18	196	1	197
Wykoff	32	10	16	10	68	2	70
Zumbro Falls	51	0	22	0	73	0	73
Totals	**1226**	**369**	**738**	**316**	**2649**	**36**	**2685**

**Includes 4 samples with no location information.*

Samples Collected by Surveillance Block*

Block	Weekend 1	Weekend 2	Weekend 3	Weekend 4	Totals	Goal	Percent of Goal**
233_293	85	22	0	15	122	300	40.7%
341	147	60	120	20	347	300	115.7%
342	110	43	71	34	258	300	86.0%
343	207	46	79	47	379	300	126.3%
344	66	8	51	27	152	300	50.7%
345	79	19	37	19	154	300	51.3%
346	120	50	100	28	298	300	99.3%
347	72	23	64	36	195	300	65.0%
348	151	33	79	45	308	300	102.7%
349	189	65	133	45	432	300	144.0%
Totals	**1226**	**369**	**734**	**316**	**2645**	**3000**	**88.2%**
Non-Surveillance	24	4	8	0	36	NA	NA
Grand Totals	**1250**	**373**	**742**	**316**	**2681**	—	—

**Does not include 4 samples with no location information.*

**Goal: 3,000 samples (300/surveillance block).

December 8th, 2009

robert.wright@state.mn.us

Table 3. Expenditure details for fall CWD surveillance program.

Expenditure		Total cost
MNDNR Staff Salary		$174,050
MNDNR Staff Travel Expenses		$18,000
Veterinary Student Labor & Travel Expenses		$42,815
Other Student Labor & Travel Expenses		$52,000
Supplies		$6,000
Fleet		$28,025
Diagnostic Fees		$67,825
	Total	**$388,715**

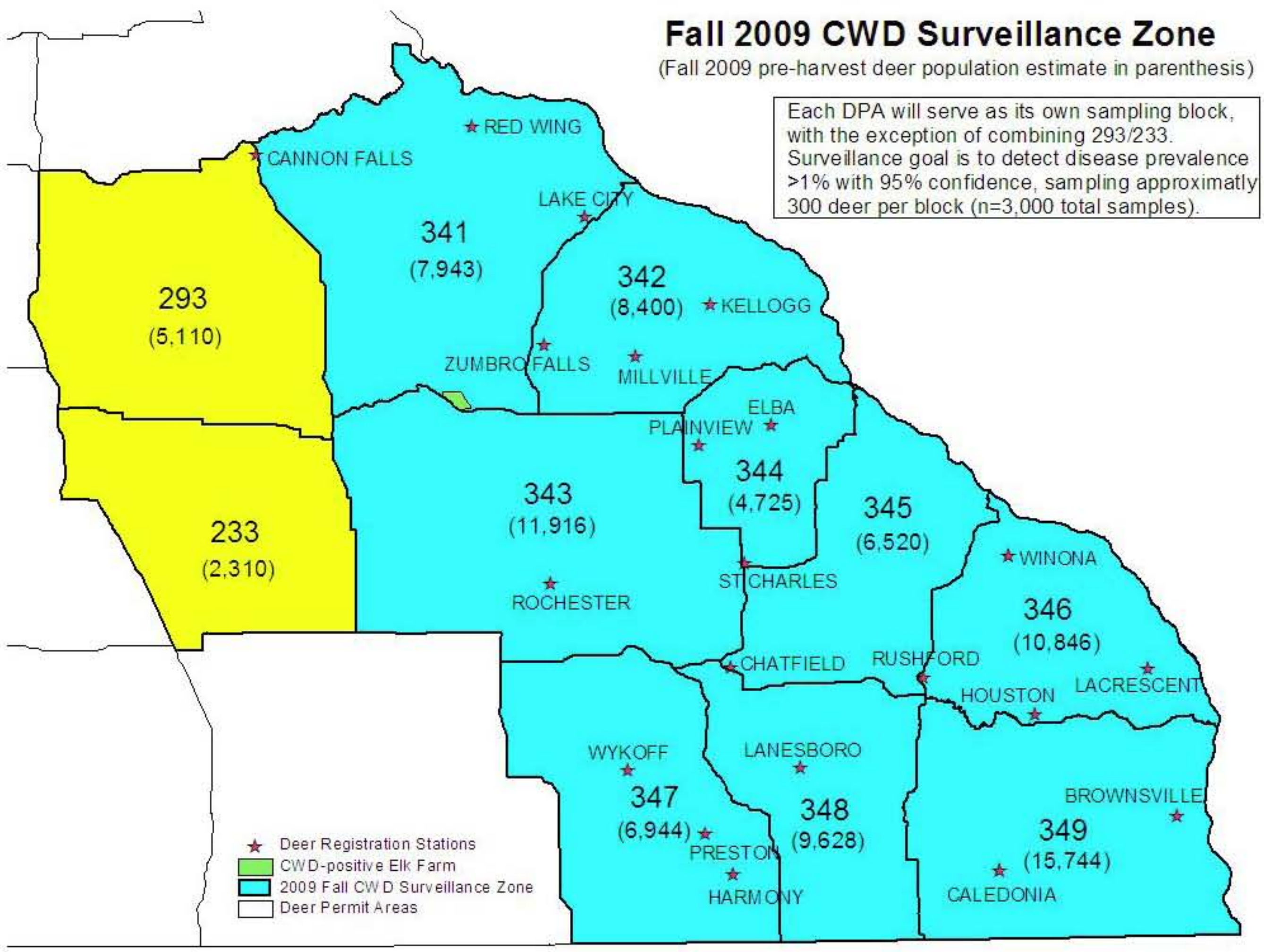


Figure 1. The Minnesota fall 2009 hunter-harvested surveillance program included 11 deer permit areas divided into 10 sampling blocks, with a total sampling goal of 3,000 samples.

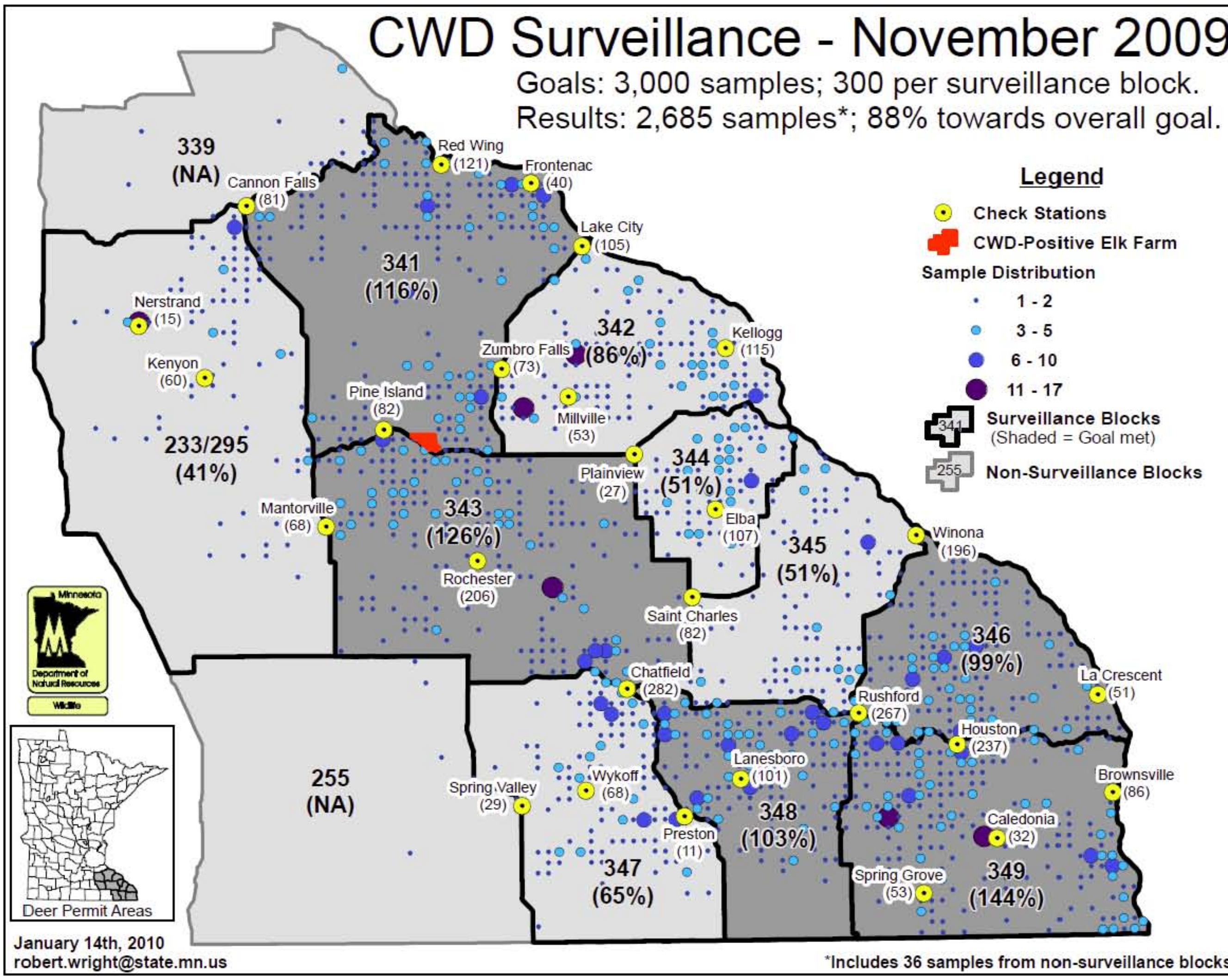


Figure 2. Sampling distribution for hunter-harvested deer (n = 2,685) tested for chronic wasting disease in southeastern Minnesota, fall 2009.

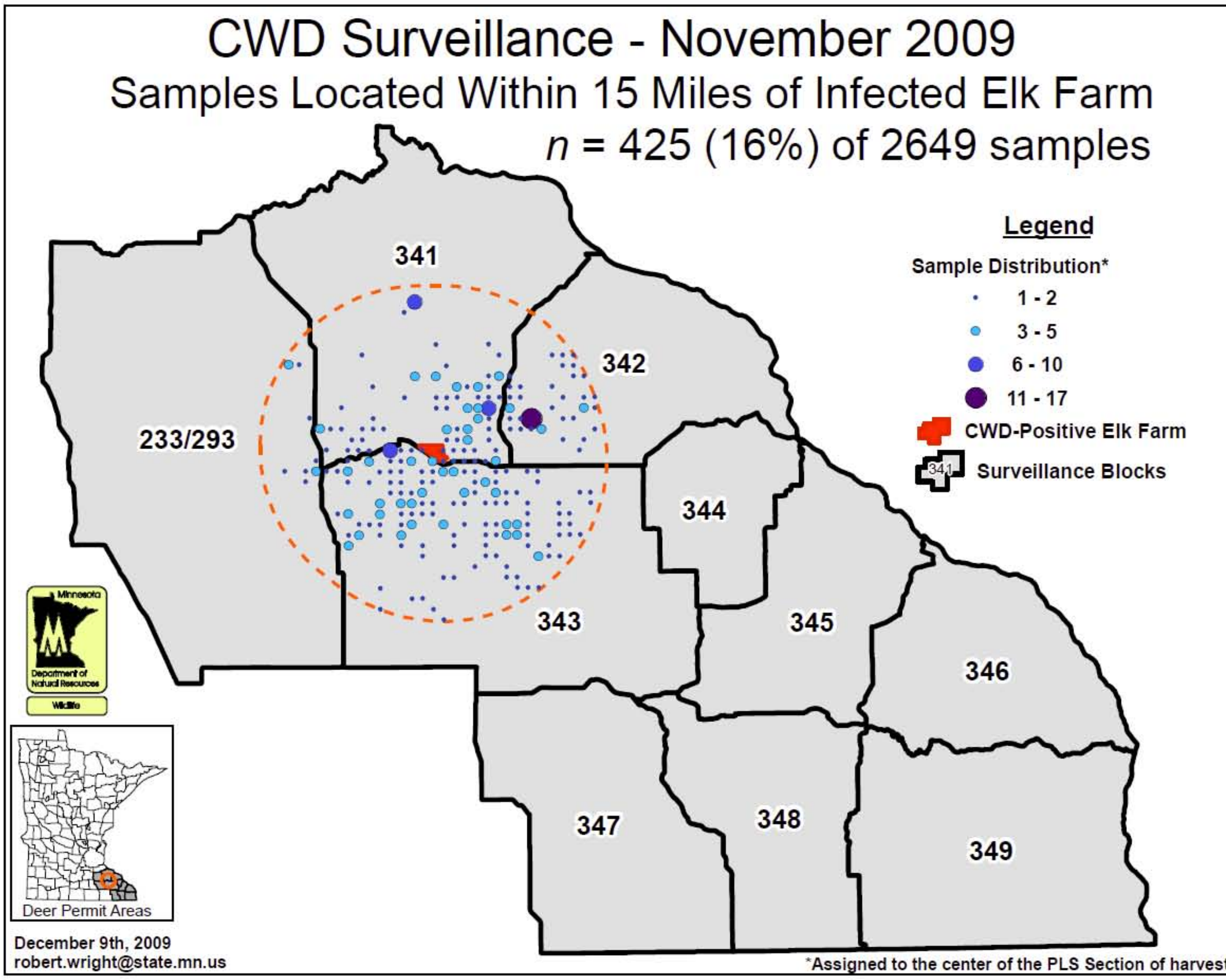


Figure 3. Sampling distribution of hunter-harvested deer (n = 425) tested for chronic wasting disease within a 15-mile radius of a CWD-positive captive elk facility in Olmsted county Minnesota, fall 2009.

PRELIMINARY RESULTS OF HERD HEALTH ASSESSMENT FOR NORTHWESTERN FREE-RANGING ELK FROM 2004-2009

Erik Hildebrand[1], Michelle Carstensen, Erika Butler, and Lou Cornicelli

SUMMARY OF FINDINGS

The goal of this project was to assess the health of free-ranging elk (*Cervus elaphus)* from northwestern Minnesota (NW MN) by screening animals for a variety of diseases and parasites. Results indicate exposure to these pathogens, and not necessarily clinical illness. From the elk included in this study (*n*=86), we identified exposure to eastern equine encephalitis, West Nile Virus, malignant catarrhal fever, *Neospora*, anaplasmosis, borreliosis, bovine viral diarrhea virus 1 and 2, bovine herpes virus 1, *Leptospira sp*., and parainfluenza virus 3. A variety of fecal parasites were also identified (*Coccidia, Strongyle-type ova,* and *Moniezia*) on fecal examination. Lung and liver tissue were cultured for bacterial infection; *Streptococcus sp.* was isolated from the lung of 1 individual and no isolations were found in liver samples. All elk were negative for *Mycobacterium paratuberculosis,* blue tongue virus, epizootic hemorrhagic disease, brucellosis, chronic wasting disease, and bovine tuberculosis. Hepatic mineral levels were also evaluated.

INTRODUCTION

Elk are native to Minnesota and were formally protected from hunting in 1893. By the early 1900s, elk became scarce and the last native Minnesota elk was reportedly seen in the Northwest Angle in 1932. Reintroduction efforts were initiated in 1914 and 1915 which brought elk from Yellowstone National Park (WY) and Jackson (WY) to Minnesota's Itasca State Park. The herd expanded to 25 animals by 1925 (MNDNR 2008). In 1935, 27 elk from Itasca State Park were moved to the Red Lake Game Preserve, which then expanded to nearly 100 animals by the 1940s. This herd, referred to as the Grygla herd, primarily occupies a 45 mi^2 area north of Grygla, MN (Figure 1). In 1987, as complaints of elk causing crop damage increased, the Legislature created a compensation program for crop depredation and imposed limits on the elk herd size to pre-calving numbers of 20–30 animals. To accomplish the required reduction in elk numbers, the Minnesota Department of Natural Resources (MNDNR) instituted elk hunts in 1987, 1996, 1997, and 1998; yet, very few animals were taken each year (MNDNR 2008). The decision to hold a hunting season is based on herd size, and current policy requires a hunt if there are more than 30 in the herd before calving. Hunts have occurred since 2004 and the most recent aerial survey indicated the pre-calved Grygla herd is currently at approximately 40 animals, with the goal remaining at 30-38 elk.

A second herd of elk occurs in Kittson and Roseau Counties (Figure 1), and is termed the Kittson County herd. First noted along the Manitoba border in the early 1980s, these animals winter in Manitoba, while calving and spending the summers in MN. They were originally divided into 3 subgroups based on distinctive areas of use (Figure 2). These 3 subgroups were the Water Tower subgroup (north of Lancaster), the Lancaster subgroup (east of Lancaster) and the Caribou/Vita subgroup (located between Caribou, MN and Vita, Manitoba). The Caribou/Vita subgroup is known to occupy either side of the international border at any given time of year. The extent to which the other 2 subgroups cross into Canada is unknown. Little is also known regarding the extent of animal interchange between the Caribou/Vita subgroup and the other 2 subgroups (MNDNR 2009). Due to crop depredation issues, a hunting season was first held in 2008. The most recent elk survey estimated 27 (pre-calving) animals in the Kittson County herd excluding what might be in the Caribou-Vita subgroup. The current Elk Management Plan set a pre-calving population goal for the

Corresponding author e-mail: erik.hildebrand@state.mn.us

Watertower and Lancaster subgroups at 20-30 each. The population goal for the Caribou-Vita subgroup is still under discussion with MNDNR and Manitoba Conservation. In 2010, the Water Tower and Lancaster subgroups were combined and are described as the Kittson Central Elk subgroup, thus we now begin to recognize both the Kittson Central Elk subgroup and Caribou-Vita subgroup.

Expansion of elk in MN is limited by both habitat succession and reproduction rates within the herds, but the social factor of mandating the herd to a specified level is the main limiting factor. The purpose of this project was to screen NW MN elk for a variety of disease agents to determine which diseases they were being exposed to. Positive results are not diagnostic of clinical disease. While some of the test results may be all negative, this does not necessarily imply that the disease is not present or impacting the population. Some diseases cause death quickly without an immune response; thus, finding a positive in a seemingly healthy animal would be extremely rare.

Discovery of bovine tuberculosis (TB) in cattle and free-ranging white-tailed deer (*Odocoileus virginianus*) has brought increased scrutiny as to the health status of the NW MN elk, particularly the Grygla Herd. While overlap in range between elk and known TB-infected deer or cattle farms is known to occur, there has been no evidence of TB-infection in MN's elk herd. TB-infected cattle and deer in MN share the same strain, which is considered of Mexican or southwest US origin, and is not related to the strain of bovine TB found in elk in Manitoba's Riding Mountain National Park.

METHODS

For this report, all elk sampled from NW MN were grouped as either *harvested* animals (including hunter-harvested, removed under depredation permits, agency sharpshooting, and illegally poached) or *other* (including road kills, sick, and found dead elk). All elk within the harvested category were assumed to be representative of healthy individuals within the population.

For hunter-harvested elk, hunters were asked to collect samples of lung, liver, feces, blood, hair, and an incisor for aging. MNDNR provided a project overview, instructions for sample collection, and sampling kits at the mandatory elk hunter orientation sessions. Elk removed under depredation permits or other methods were sampled by trained MNDNR staff.

All equipment needed for sample collection/preservation was included in the sampling kit: soft-sided cooler; 1-60cc syringe for blood collection; 6-15cc serum tubes for blood storage; 3 whirlpaks for a sample of liver, lung, and feces; 2 specimen jars with formalin for liver and lung samples; 2 coin envelopes for hair and tooth; datasheet; protocol; Sharpie marker; 1 pair of large vinyl gloves; and 1 ice pack. Successful hunters dropped off their sampling kits when they registered their animal and also provided information on the location of their kill.

Hunters collected blood from the chest cavity as soon after death as possible, using a 60 cc syringe. The blood was placed in serum tubes and kept cool until they were delivered to official MNDNR registration station. Liver and lung samples were collected and split, with half placed in a formalin jar, while the other half was frozen in whirlpak bags. If the hunter found anything unusual, such as a large abscess or tumor, those samples were also collected and split between the preservation methods (formalin fixation and freezing). Blood was centrifuged at the registration stations and serum was extracted and frozen. Cranial lymph nodes and obexes were removed by trained MNDNR staff at the registration stations to allow for chronic wasting and bovine tuberculosis testing. Where appropriate, MNDNR made arrangements with taxidermists to collect samples from trophy animals. All samples were submitted to the University of Minnesota Veterinary Diagnostic Laboratory (VDL), where the majority of the testing occurred; some tests were outsourced to the National Veterinary Services Laboratories (NVSL) in Ames, IA.

RESULTS AND DISCUSSION

A total of 86 elk were included in this health assessment project (Figure 3). Harvested elk accounted for 82 of the animals (61 hunter-harvested, 17 depredation permits, 1 sharpshooting, and 3 poached). In addition, 4 other animals were sampled (1 roadkill, 1 found dead, 1 shot by law enforcement due to possible injury/sickness, and 1 clinically ill elk (observed with neurological symptoms). The sick animal was dispatched by a local conservation officer and necropsy results indicate the observed clinical illness was likely due to *P.tenuis* infection, although it was also positive for *L. interrogans serovar icterohaemorrhagicae*.

Serologic results from harvested elk indicate exposure to eastern equine encephalitis, West Nile Virus, malignant catarrhal fever, *Neospora*, anaplasmosis, borreliosis, bovine viral diarrhea virus 1 and 2, bovine herpes virus 1, *Leptospira sp*., and parainfluenza virus 3 (Table 1). Liver samples from 65 harvested elk were evaluated for heavy metal and mineral status (Table 2). Though not included in Table 2, the sick elk's hepatic mineral values fell within the means of all other harvested elk. Exact age was determined for 68 harvested elk (μ= 4.2 years; *sd* = 3.7 years; range 0.5 to 16 years old) (Figure 4). There were nearly twice as many females (n = 43) than males (n = 26) of known sex in the harvested elk category.

Complete sets of samples were not collected from all elk included in this project, as field conditions and sample quality varied; however, there were very few errors in tissue identification or insufficient sample quantities in those submitted. The following discussion provides an overview of the major findings from 86 elk included in this study (2004-2009). Samples from an additional 11 elk removed by sharpshooters from the Kittson County herd in spring 2010 are not included in this report, as results are pending.

Mosquito-Borne Viruses

Positive results were reported for 6 of 44 elk (13.6%) tested for eastern equine encephalitis (EEE) (Table 1, Figure 5). The positive results indicate that these animals were likely exposed to the EEE virus as the virus neutralization (VN) test prevents cross-reactivity with other viruses. Two harvested animals had titers ≥ 100. A titer that is greater than 100 is considered a VERY strong positive and indicates that the serum was able to neutralize nearly 100% of the virus. EEE is spread by mosquitoes and causes neurologic signs and often death. It poses a greater mortality threat for most species than West Nile Virus (WNV) does. Horses, deer, and other mammals are incidental, dead-end hosts of EEE virus. Under natural transmission conditions, they are only infected by bridge vectors, mosquito species that feed both on birds and large mammals (Schmitt et al. 2007).

Positive results were reported for 32 of 45 elk (71.1%) tested for West Nile Virus (Table 1, Figure 5). Four elk had titers ≥ 100. Little is known about the effects of WNV in elk. In white-tailed deer it has been found that they often have a low titer and no clinical signs. However, the United States Department of Agriculture (USDA) has reported that reindeer (*Rangifer tarandus*) infected with WNV have high mortality rates and high titers, indicating that the virus may be more serious for some species than others.

Malignant Catarrhal Fever

Samples from 46 elk were submitted to NVSL for peroxidase-linked assay (PLA) testing for Malignant Catarrhal Fever (MCF) from 2004-2009. If the PLA test came back positive, the samples were further screened with a VN test. A total of 13 samples tested positive on the PLA test (28.9%) (Table 1, Figure 5); 11 with titers at 1:20, and 2 at 1:100. However, all elk were negative on VN. The PLA test is more sensitive than the virus isolation, meaning it is much better at identifying true positives. VN is more specific, which means it is better at identifying true negatives. There are a couple of issues with this testing. First, the PLA reacts with multiple Gammaherpes Viruses (such as the wildebeest strain, the sheep strain, the deer strain, etc). A

PLA positive does not indicate which strain has been found, it only indicates that one has. The higher the positive value with the PLA test, the stronger the positive in the sample. Second, the VN test only screens for the wildebeest strain (which is exotic to the U.S.) and would be negative if other strains are present. This means a sample that was positive on PLA and negative on VN was likely exposed to a gammaherpes virus, but not the wildebeest strain.

Gammaherpes viruses have been documented to cause serious illness and death in elk and other ruminants. The clinical symptoms can mimic *P. tenuis* infection as the animals often exhibit neurological deficits, go blind, and thrash on the ground prior to death. While infection with MCF frequently results in death, carrier status can occur and is identified with serology. Li et al. (1996) found small numbers of United States free-ranging elk were seropositive; these animals were once exposed to MCF viruses but whether they had recovered from a non-lethal disease is unknown.

Fecal Examination for Parasites

Fecal samples from 58 elk were screened for evidence of parasites from 2004-2009. Parasites were identified in 5 samples (8.6%), including *Fascioloides magna, Coccidia sp., Strongyle-type ova,* and *Moniezia sp.* Negative results do not necessarily mean the animal was parasite-free, only that it was not actively shedding at the time the feces were collected.

Pulmonary *Mycoplasma* and Hepatic Salmonella Culture

From 2004-2009, a total of 18 lung samples were cultured for *Mycoplasma* and 19 liver samples were cultured for Salmonella. None was isolated.

Mycobacterium Paratuberculosis (Johne's Disease)

During this study, a total of 43 fecal samples were cultured for *M. paratuberculosis* and 57 fecal samples were genetically screened (polymerase chain reaction, PCR) from the bacterium. Additionally, a serological test (Biocor) was run on 52 samples. All culture, PCR, and Biocor results were negative for Johne's disease. The negative fecal cultures and PCR results indicate that those elk were not actively shedding the bacterium. The negative Biocor results indicate that these animals had not been exposed to the bacterium.

All species of ruminants are believed to be susceptible to Johne's and it is frequently diagnosed in cattle and sheep (Manning and Collins 2001). Elk infected with Johne's may show non-specific clinical signs including poor weight gain and poor shedding of hair coat, and rapid weight loss and diarrhea may occur just prior to death (Barber-Meyer et al. 2007). Elk, mule deer (*Odocoileus hemionus)*, white-tailed deer, bighorn x hybrid, and domestic sheep were susceptible to infection with *M.paratuberculosis* derived from paratuberculous bighorn sheep (*Ovis Canadensis)* (Williams et al. 1983). During the first year of exposure, only deer developed clinical paratuberculosis, characterized by poor body condition and diarrhea.

Anaplasmosis

A total of 46 samples were screened for Anaplasmosis (*Anaplasma phagocytopila*, formerly *Ehrlichia phagocytophila*) with the card test from 2004-2009. All animals were negative. In sheep, this disease produces significant effects on the immunological defense system, increasing their susceptibility to disease and secondary infections (Larson et al. 1994). Experimental studies have shown that elk can harbor asymptomatic infections with *A. marginale* and *A. ovis,* the causes of anaplasmosis in cattle and sheep, respectively. However, efforts to recover *Anaplasma* spp. from free-ranging elk populations have been unsuccessful, suggesting that even though these species are susceptible, they are probably not responsible for maintaining infections or acting as a source of infection for cattle (Corn and Nettles 2001).

Borreliosis (Lymes Disease)

A total of 45 elk were screened for Lyme disease with an immunofluorescence assay (IFA) from 2004-2009. Positive results were reported for 30 elk (66.7%) (Table 1, Figure 5). Borreliosis is a tick-borne bacterial disease that is maintained through a wildlife/tick cycle involving a variety of species, including mammals and birds. While evidence of natural infection in wildlife exists, there has been no documentation of clinical disease or lesions reported in wildlife species.

Brucellosis

A total of 49 elk were screened for *Brucella* with the card test. All results were negative, indicating that these animals were not likely exposed to the bacterium. Brucellosis has been a major disease issue among elk, bison and cattle in western states. The disease causes spontaneous abortions and is most likely spread through oral contact (e.g., licking or ingestion of contaminated materials) (Thorne et al. 1978).

Bovine Viral Diarrhea Virus (BVD) 1 and 2

A total of 56 elk were tested by serum neutralization (SN) for BVD 1 and 2 from 2004-2009. Seven animals tested positive (12.5%) (Table 1, Figure 5). These results indicate that the elk population from NW MN was being exposed to BVD. Two animals had positive titer levels at 32/16, 3 were positive at 32/negative, and 2 had a titer level at 8/negative.

BVD is considered a major disease of cattle and is thought to be the most common infectious cause of reproductive failure in beef herds in the western U.S. BVD also causes enteritis, mucosal disease, infections, and respiratory disorders in cattle though experimentally inoculated non-pregnant elk showed no clinical signs and remained healthy for >50 days post inoculation (Barber-Meyer et al. 2007).

Bovine Herpes Virus 1 (BHV)

A total of 57 elk were screened for BHV using a SN test from 2004-2009. Five animals were positive (7.1%) (Table 1, Figure 5). BHV is a disease of the respiratory tract. It is believed to infect all ruminant species and has been isolated from a large number of wild species. It is most commonly isolated in feedlot cattle.

Epizootic Hemorrhagic Disease (EHD) and Blue Tongue Virus (BTV)

A total of 59 elk were screened for EHD using an Agar Gel Immuno Diffusion (AGID) test and BTV using a Competitive Enzyme-Linked Immunoabsorbent Assay (cELISA) from 2004-2009. All results were negative. EHD and BTV are a hemorrhagic diseases transmitted by a biting midge that is known to cause illness and death in white-tailed deer. While it is known to be infective to a variety of domestic and wild ruminants, clinical disease is quite variable.

Leptospira sp.

A total of 59 elk were screened for 6 species of *Leptospira,* using a microscopic agglutination test (MAT), from 2004-2009. Positive results are reported per *Leptospira* species below (Table 1, Figure 5):

- *L. bratislava*:
  - 0/59
- *L. canicola*:
  - 0/59

- *L. grippothyphosa*:
    - 0/59
- *L. hardjo*:
    - 1/59 (1.7%)
- *L. interrogans* serovar *icterohaemorrhagicae*:
    - 7/59 (11.9%)
- *L. pomona*:
    - 1/59 (1.7%)

The positive *L. hardjo* had a titer level of 200. Of the positives for *L. interrogans* serovar *icterohaemorrhagicae,* 5 had a titer of 100 and 1 with a titer of 200. The positive *L. Pomona* had a titer of 800.

Leptospirosis is a bacterial disease that can infect a wide variety of mammals, both domestic and wild. Exposure usually occurs through direct contact with urine from carrier animals or indirectly by contact with a urine- contaminated environment (Bender and Hall 1996). Much of the landscape of NW MN contains environments where moist alkaline soils are present to house the bacteria, and it may survive for several weeks (Bender and Hall 1996).

Neospora sp.

A total of 51 elk were screened for *Neospora* with an ELISA test from 2004-2009. All samples tested negative. While clinical disease due to neospora infection is best described in domestic animals, reports of ill effects due to *Neospora* infection in wildlife do exist. Systemic neosporosis was diagnosed in a California black-tailed deer (*Odocoileus hemionus*) that was found dead (Woods et al. 1996). *Neospora caninum* causes abortion and serious clinical disease in livestock and companion animals, although dogs and coyotes are its only known definitive hosts that can shed oocysts (Dubey and Thulliez 2005). Recent study of neospora prevelance in white-tailed deer in northwestern MN reported 71% prevalence, indicating the parasite is present in elk range (Dubey et al. 2009).

Parainfluenza Virus 3 (PI)

A total of 53 elk were screened for PI using a hemagglutination inhibition (HI) test from 2004-2009. Eighteen animals were positive (34%) (Table 1, Figure 5), with titers of 6 at 10, 4 at 20, 4 at 40, 3 at 80, and 1 at 160.

The positive results indicate that NW MN elk were exposed to PI. Domestic ruminants are considered the main source of infection for free-ranging ruminants. PI causes mild respiratory disorders in domestic cattle and sheep that serve as initiators for secondary infections of *Pasteurella* spp., which can result in bacterial pneumonia, but clinical symptoms in wild elk remain unknown (Barber-Meyer et al. 2007).

Chronic Wasting Disease (CWD)

From 2004-2009, a total of 58 elk were screened for CWD using immunohistochemistry (IHC); including 42 animals with obex samples and 53 retropharyngeal lymph node samples. All results were negative. CWD is a transmissible spongiform encephalopathy that causes neurological disease in cervids. CWD is known to occur in elk, but has never been documented in wild cervids in MN.

Bovine Tuberculosis

From 2004-2009, 77 sets of cranial lymph nodes (parotid, retropharyngeal, and submandibular) were collected and cultured for *Mycobacterium bovis*. All results were negative. Bovine tuberculosis is a chronic, progressive bacterial disease that infects a wide array of mammals. Bovine tuberculosis has been found in wild white-tailed deer in a small, localized area in NW MN which overlaps with the elk range, but has not been found in any wild elk in the state.

Distribution of Positive Results for Select Diseases

The geographic distribution of positive results for select disease agents was briefly evaluated (Figure 5). It was interesting to note that elk which tested positive for BVD all originated from the Kittson county herd while the elk which tested positive for BHV all originated from the Grygla herd. The significance of this finding is unknown. There was no clustering of positives observed for WNV, EEE, MCF, Borrelia, PI or Leptospirosis.

ACKNOWLEDGMENTS

This project would not have been possible without assistance from a number of MNDNR employees and volunteers. We would like to especially thank staff in NW MN who helped with collecting samples: Joel Huener, Donovan Pietruszewski, Christine Reisz, Randy Prachar, Dawn Torrison, Lou Cornicelli, and Marshall Deters. Finally we appreciate the help of all the volunteers including veterinary students Jonna Swanson, Macdonald Farnham, and Andrea Widdel who helped with this elk project.

REFERENCES

Barber-Meyer, S., White, P., and Mech, D. 2007. Survey of selected pathogens and blood parameters of Northern Yellowstone elk: wolf sanitation effect implications. The American Midland Naturalist 158 (2): 369-381.

Bender, L., and Hall, P. 1996. *Leptospira interrogans* exposure in free-ranging elk in Washington. Journal of Wildlife Diseases 32(1): 121-124.

Corn, L. and Nettles, V. 2001. Health protocol for translocation of free-ranging elk. Journal of Wildlife Diseases 37(3): 413-426.

Dubey, J. P, M. C. Jenkins, O.C.H. Kwok, R. L. Zink, M. L. Michalski, V. Ulrich, J. Jill, M. Carstensen, and P. Thulliez. 2009. Seroprevalence of *Neospora caninum* and *Toxoplasma gondii* antibodies in white-tailed deer (*Odocoileus virginianus*) from Iowa and Minnesota using four serologic tests. Veterinary Parasitology 161: 330–334.

Dubey, J. P., and Thulliez, P. 2005. Prevalence of antibodies to *Neospora caninum* in wild animals. The Journal of Parasitol., 91(5): 1217-1218.

Larsen, H. J. S., G. Overnes, H. Waldeland, and G.M. Johansen. 1994. Immunosuppression in sheep experimentally infected with *Ehrlichia phagocytophila*. Research in Veterinary Science 56: 216-224.

Li, H., D. Shen, D. Jessup, D. Knowles, J. Gorham, T. Thorne, D. O'Toole, and T. Crawford. 1996. Prevalence of antibody to malignant catarrhal fever virus in wild and domestic ruminants by competitive-inhibition ELISA. The Journal of Wildlife Diseases 32(3):437-443.

Manning, E. J. B., and M. T. Collins. 2001. *Mycobacterium avium* subsp. *paratuberculosis:* pathogen pathogenesis and diagnosis. *In* Mycobacterial infections in domestic and wild animals, E. J. B. Manning and M. T. Collins (eds.). Revue Scientifique et Technique Office International des Epizooties 20: 133-150

Minnesota Department of Natural Resources. 2008. Minnesota long range management plan for elk: Draft. Division of Fish and Wildlife, Minnesota Department of Natural Resources.19 pp.

Minnesota Department of Natural Resources. 2009. Strategic management plan for elk: Draft. Division of Fish and Wildlife, Minnesota Department of Natural Resources. 34pp.

Schmitt, M., S, Cooley, T., Fitzgerald, S., Bolin, S., Lim, A., Schaefer, S., Kiupel, M., Maes, R., Hogle, S., and D. O'Brien. 2007. An outbreak of eastern equine encephalitis virus in free-ranging white-tailed deer in Michigan. Journal of Wildlife Diseases 43(4): 635-644.

Thorne, T., Morton, J., Blunt, F., and Dawson, H. 1978. Brucellosis in elk. II. clinical effects and means of transmission as determined through artificial infections. Journal of Wildlife Diseases 14: 280-291.

Williams, E., Snyder, S., and Martin, K. 1983. Experimental infection of some north American wild ruminants and domestic sheep with *Mycobacterium paratuberculosis:* clinical and bacteriological findings. Journal of Wildlife Diseases 19(3): 185-191.

Woods, L. W., P. K. Swift, B. C. Barr, M. C. Horzinek, R. W. Nordhausen, M. N. Oliver, K. R. Jones, and N. J. Maclachlan. 1996. Systemic adenovirus infection associated with high mortality in mule deer (Odocoileus hemionus) in California. Veterinary Pathology 33: 125-132.

Table 1. Serological results from harvested elk in northwestern Minnesota, 2004-2009.

Disease	*n*	Apparent prevalence %
EEE	44	13.6 (*n*=6)
MCF	45	28.9 (*n*=13)
WNV	45	71.1 (*n*=32)
Anaplasmosis	46	0
Borreliosis	45	66.7 (*n*=30)
Brucellosis	49	0
BVD 1 and 2	56	12.5 (*n*=7)
BHV	56	7.1 (*n*=4)
BTV	58	0
EHD	58	0
L. *bratislava*	58	0
L. *canicola*	58	0
L. *grippothyphosa*	58	0
L. *hardjo*	58	1.7 (*n*=1)
L. *interrogans serovar icterohaemorrhagicae*	58	10.3 (*n*=6)
L. *pomona*	58	1.7 (*n*=1)
Neospora	51	0
PI	53	34 (*n*=18)
Mycobacterium paratuberculosis	52	0

Table 2. Hepatic mineral values of harvested elk in northwestern Minnesota, 2004-2009.

Element	*n*	Mean	Standard deviation	Minimum	Maximum
Arsenic	64	0.71	0.37	0	1
Boron	25	0.50	0	0	0.50
Barium	25	0.03	0	0	0.025
Calcium	24	55.99	17.98	39.30	111
Cadmium	63	0.24	0.09	0	0.66
Cobalt	63	0.18	0.1	0	0.25
Chromium	25	0.10	0	0	0.10
Copper	65	12.54	11.93	1.30	63
Iron	65	199.65	162.69	31.30	946.3
Mercury	25	1.00	0	0	1
Potassium	24	2561.92	218.53	2108	2850
Magnesium	65	159.76	21.36	86	211.1
Manganese	65	2.62	0.95	0.28	5.60
Molybdenum	65	1.11	0.33	0.10	1.77
Sodium	24	947.21	193.24	626	1490
Phosphorous	24	4150.46	752.67	1650	5354
Lead	64	0.40	0.12	0	0.50
Antimony	25	0.50	0	0	0.50
Selenium	65	0.84	0.34	0	1.90
Thallium	25	1.25	0	0	1.25
Zinc	64	25.39	6.13	15.40	53

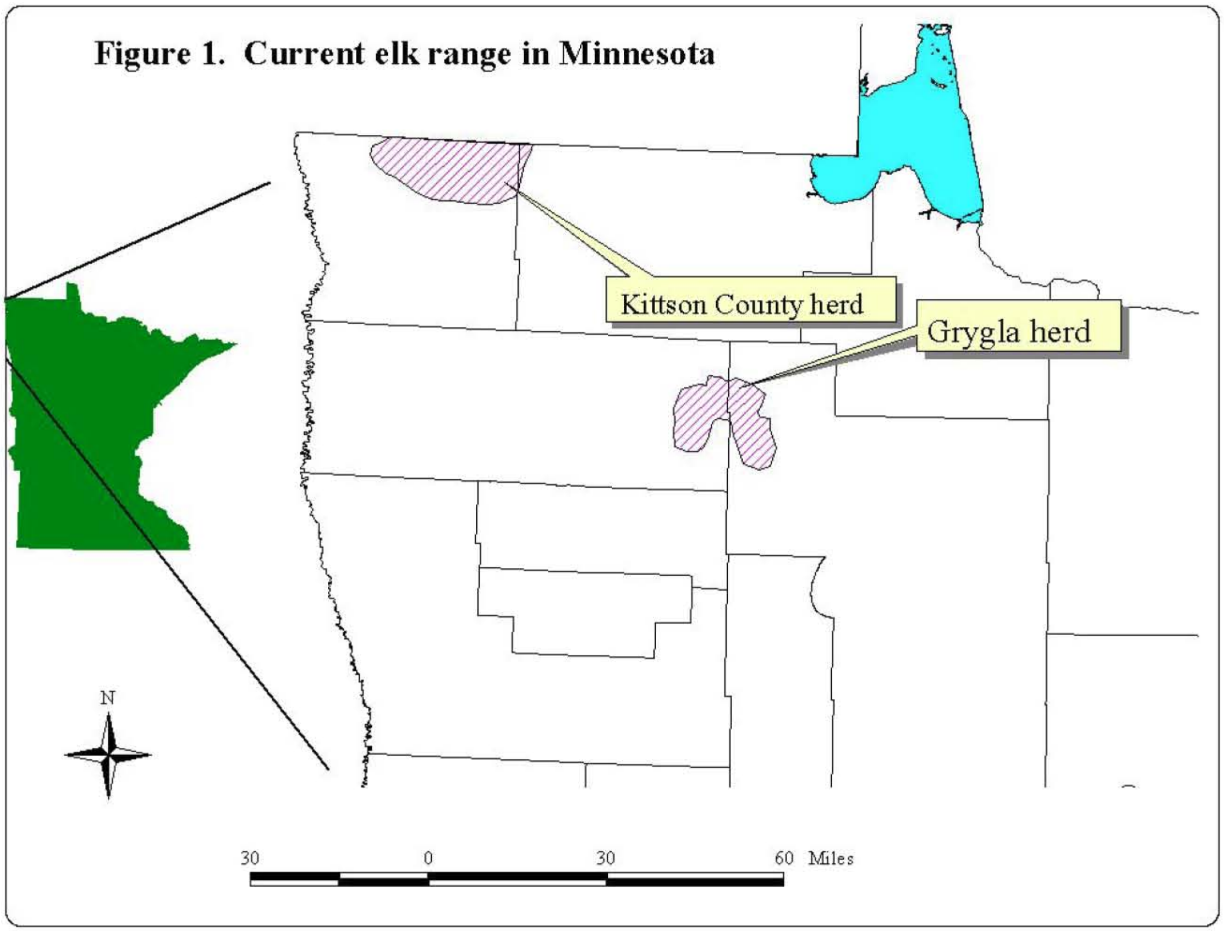


Figure 1. Current range of the 2 localized elk herds of northwest Minnesota in 2009.

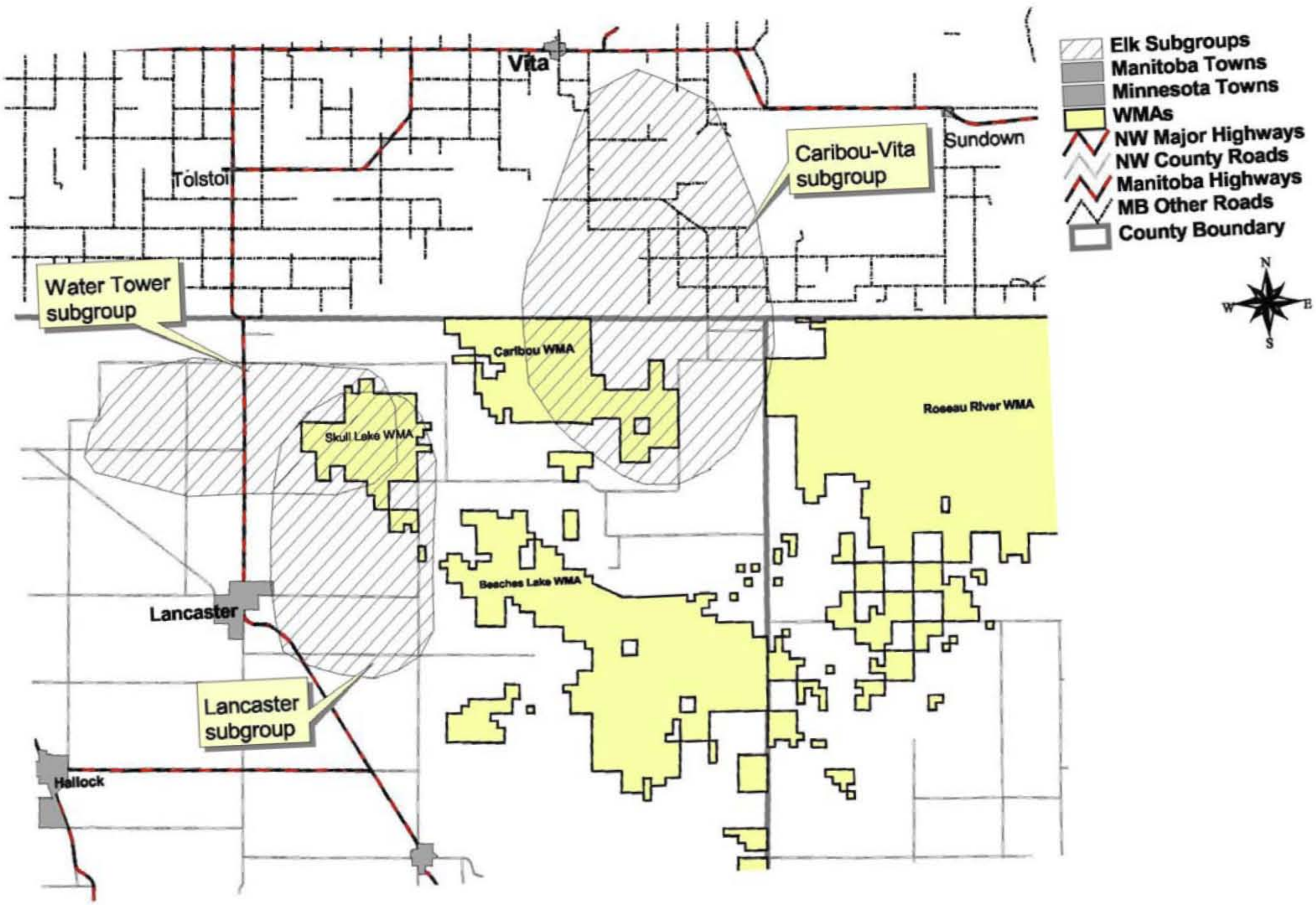


Figure 2. Three Kittson County elk herd subgroups of northwestern Minnesota in 2009.

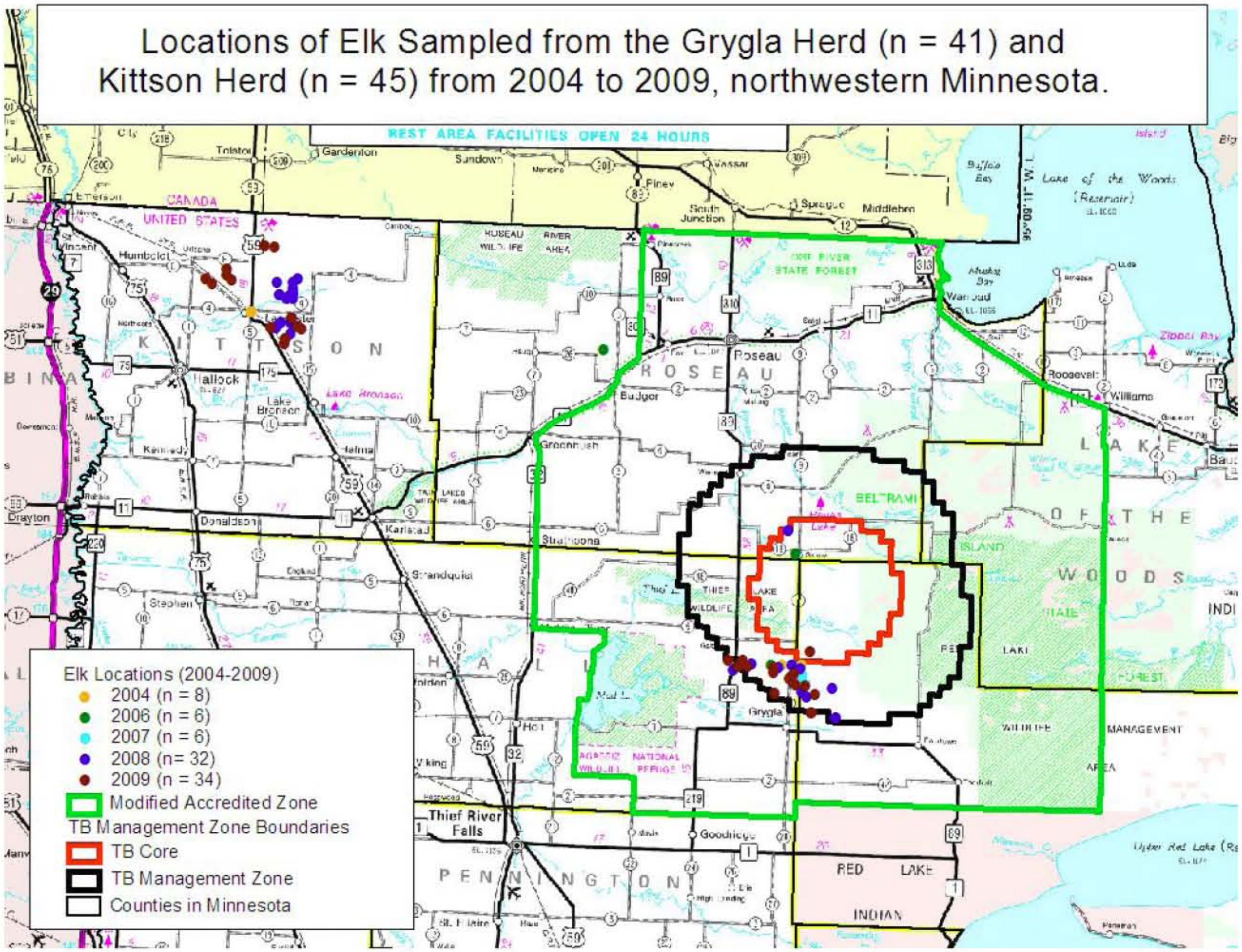


Figure 3. Locations of elk (*n*=86) included in the health assessment project, 2004-2009, northwestern Minnesota.

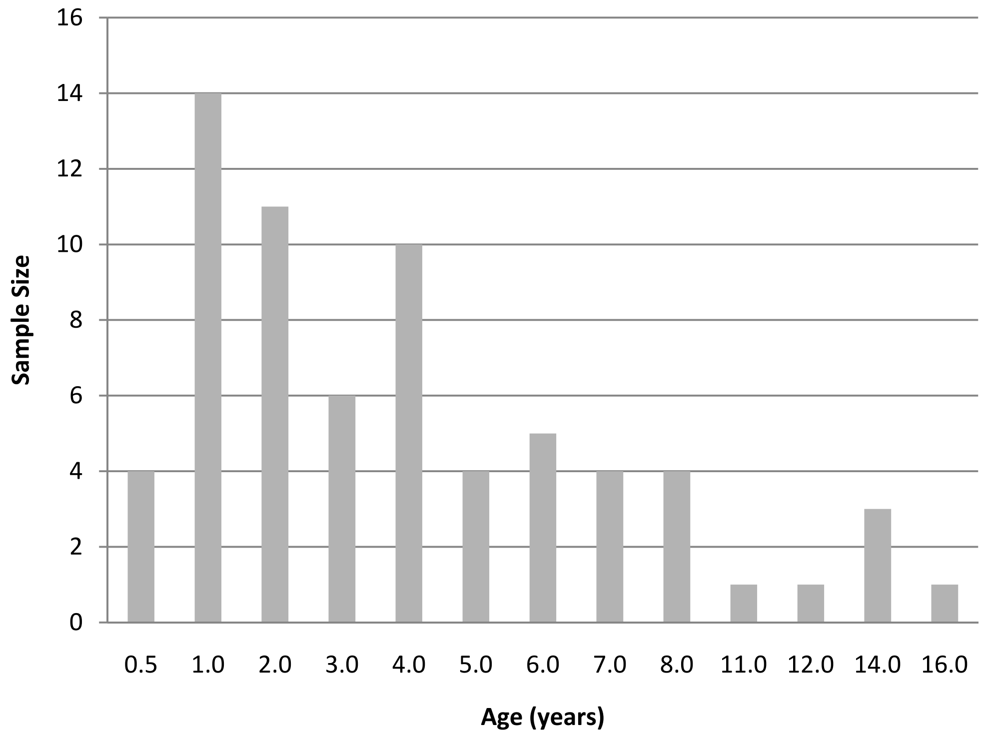


Figure 4. Age distribution of harvested elk (*n* = 68) included in the 2004-2009 health assessment project, northwestern Minnesota.

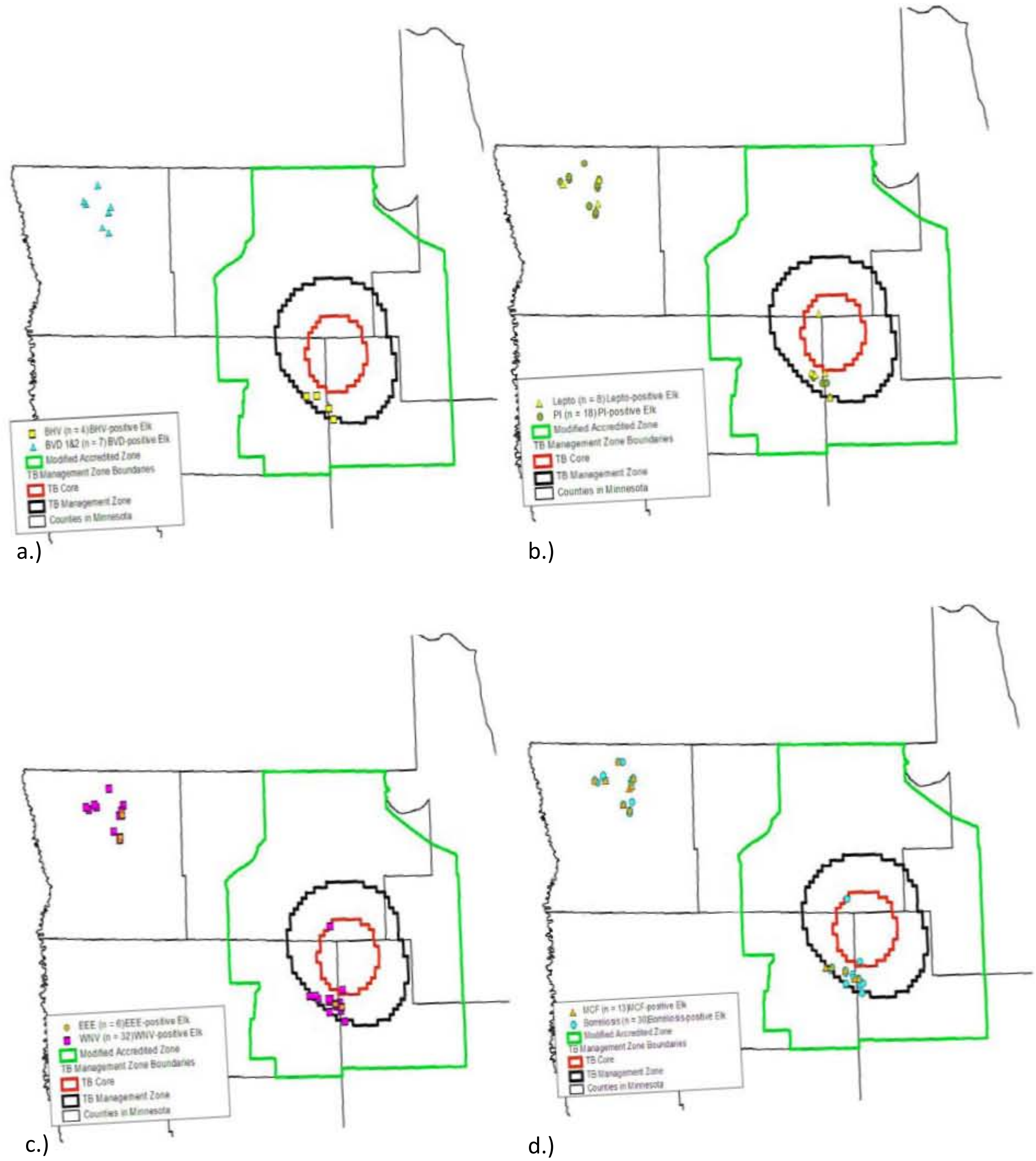


Figure 5. Locations of elk positive for a) bovine viral diarrhea (BVD, *n*=7) and bovine herpes virus (BHV, *n=4*); b) *Leptospira sp.* (Lepto, *n*=8) and parainfluenza virus 3 (PI, *n*=18); c) eastern equine encephalitis (EEE, *n=6*) and West Nile virus (WNV, *n*=32); d) malignant catarrhal fever (MCF, *n*=13) and borreliosis (*n*=30), 2004-2009, northwest Minnesota.

PRELIMINARY RESULTS FROM THE 2007-2009 MOOSE HERD HEALTH ASSESSMENT PROJECT

Erika Butler[1], Michelle Carstensen, Erik Hildebrand, John Giudice, Robert Wright, and Mike Schrage

SUMMARY OF FINDINGS

The purpose of this project was to screen 2007-2009 hunter-harvested (and presumably healthy) moose (*Alces alces*) for a variety of disease agents. Results were used to identify diseases the northeast Minnesota (NE MN) moose population have been exposed to as well served as a baseline for similar testing completed on non-hunting moose mortalities from the same population. Positive results confirmed that moose were exposed to, though not necessarily ill from, eastern equine encephalitis, West Nile Virus, malignant catarrhal fever, *Neospora*, anaplasmosis, bovine herpes virus 1, bovine viral diarrhea virus 1 and 2, borrelia, *Leptospira sp,* and parainfluenza virus 3. When possible, serological events were evaluated to determine whether there was any influence of age, location, or year of harvest. Additionally, a variety of fecal parasites were identified on fecal examination. All results were negative for *Mycobacterium paratuberculosis*, brucellosis, blue tongue virus, epizootic hemorrhagic disease, chronic wasting disease, and bovine tuberculosis. Hepatic mineral values were evaluated, whole livers were examined grossly and ranked according to the level of damage due to liver fluke infection, and histological examination of whole brains investigated how many apparently healthy moose have lesions consistent with migration tracts (presumably due to *P. tenuis*).

INTRODUCTION

Several lines of evidence suggest that the moose population in northeastern Minnesota is declining. Since 2002, annual survival and reproductive rates were substantially lower than documented elsewhere in North America (Lenarz et al. 2007) and modeling based on these vital rates indicated that the population is declining by approximately 15% per year since at least 2002 (Lenarz et al. 2010). Likewise, recruitment and twinning rates have steadily declined since 2002 (Lenarz 2009). In addition, hunter success rates have steadily decreased since 2001 (Lenarz 2009). Finally, anecdotal reports from local residents have reported a noticeable decline in moose numbers. Parasitic infection with *Parelaphostrongylus tenuis, Echinococcus granulosus, Elaeophora schneideri, Sarcocystis spp*., *Fascloides magna,* and *Dermacenter albipictus* has been documented in Minnesota's moose. Copper deficiency has been reported in some moose. Poor antler development has also been noted in some bull mortalities. Many causes of mortality remain unknown with numerous prime-age animals dying, often during low stress periods of the year.

The purpose of this project was to screen presumably healthy moose for a variety of disease agents. Results were intended to indicate which diseases the NE MN moose population were exposed to. Exposure, itself, does not imply the animal was clinically ill with the disease. They also served as a baseline, allowing for comparisons between similar testing completed on non-hunting moose mortalities from the same population. While some of the test results may be all negative, this does not necessarily mean that the disease is not present or impacting the population. Some diseases cause death quickly and without an immune response; thus finding a positive in a seemingly healthy animal would be extremely rare.

[1] Corresponding author email: erika.butler@state.mn.us

METHODS

In order to conduct this herd health assessment, hunters (both tribal and state) were asked to collect samples of lung, liver, blood, feces, hair, ticks, and an incisor for aging. The Wildlife Health Program provided a presentation and instructions relative to the moose herd health assessment project at the mandatory Minnesota Department of Natural Resources (MNDNR) Moose Hunt Orientation Sessions and tribal natural resource offices. Hunters were given a sampling kit with instructions at the sessions. Post-harvest, the sampling kits were dropped off at official registration stations by the hunters at the time of registration. Hunters were asked to locate their kill site on appropriate maps.

MNDNR provided hunters with all equipment needed for sample collection/preservation. Sampling kits included the following items: cooler; 1-60cc syringe for blood collection; 6-15cc serum tubes for blood storage; 3 whirlpaks for a sample of liver, lung and feces; 2 specimen jars with formalin for liver and lung samples; 2 coin envelopes for tooth and hair; datasheet; protocol; Sharpie marker; 1 pair of large vinyl gloves; and 1 icepack. In 2008, 1 -5-cc whole blood tube was added to the kits.

The hunters collected blood from the chest cavity as soon after death as possible, using a 60 cc syringe. The blood was placed in serum tubes and kept cool until they were delivered to official MNDNR registration stations or tribal natural resource offices. Liver and lung samples were collected and split, with half placed in a formalin jar, while the other half was frozen in whirlpak bags. In 2009, we asked hunters to begin collecting whole livers in addition to the formalin fixed sample. If the hunter found anything unusual, regardless of the location in the carcass, such as a large abscess or tumor, those samples were collected and split between the preservative methods (formalin fixation and freezing). Blood was centrifuged at the registration stations or tribal natural resource offices and serum was extracted and frozen. In 2008, we began collecting whole blood as well, from which blood smears were made and the remaining whole blood was frozen. Also, retropharyngeal lymph nodes, obexes, and whole brains (2008 and 2009 only) were removed by trained MNDNR staff, tribal staff, and volunteers at the registration stations with permission of the hunters. Portable refrigerators were located in advance at the registration stations to maintain the tissue samples. Samples were submitted to the University of Minnesota Veterinary Diagnostic Laboratory, where much of the testing occurred. A few of the tests were outsourced to the National Veterinary Services Laboratories (NVSL) in Ames, IA.

RESULTS AND DISCUSSION

Samples from 368 moose were submitted for diagnostic screening from 2007-2009 (n = 128, 2007; n = 118, 2008; n = 122, 2009). Our samples originated from hunter-harvested animals and state hunters were only allowed to harvest males (some tribal hunters were allowed to take females); thus most of our samples were from male moose (Table 1). Samples were collected throughout our moose hunting zones (Figure 1.) and from three counties (Figure 2.). Precise ages were determined from moose in which hunters provided a central incisor. Ages of animals sampled ranged from <1 year to 14 years old (Figure 3.). A summary of serological testing results can be found in Table 2.

Eastern Equine Encephalitis (EEE)

Three hundred and thirty serum samples were submitted to NVSL for Virus Neutralization (VN) (2007-2008) or plaque reduction neutralization (PRNT) (2009) screening for EEE. Positive results were reported for 20 of the moose (6.1%) (Figure 4). The positive results indicated that these animals were exposed to the EEE virus. A titer that is greater than 100 is considered a VERY strong positive and means that the serum was able to neutralize nearly 100% of the virus. Multiple animals had titers $\geq$100.

MNDNR will be continuing EEE surveillance in hunter-harvested moose for the next 3 years (2010-2013) in an attempt to determine if there is a year effect in the prevalence rate. EEE is spread by mosquitoes and causes neurologic signs and often death. It poses a greater mortality threat for most species than West Nile Virus does, though the effects of EEE infection have not been studied in moose.

West Nile Virus (WNV)

A total of 330 samples were submitted to NVSL for VN (2007 and 2008) and PRNT (2009) screening for WNV. Positive results were reported for 115 moose (34.8%) (Figure 5). Positive results indicated that animals were exposed to the WNV. A titer that is greater than 100 is considered a VERY strong positive and means that the serum was able to neutralize nearly 100% of the virus. Multiple animals had titers ≥ 100.

We found some evidence of a higher WNV exposure rate among adult males (36.4%, 95% CI: 30.8-42.5) compared to yearling males (21.4%, 95%CI: 9.0-41.0), but the estimate for yearlings was imprecise due to small sample sizes. For adult male moose, we found no evidence, or only very weak evidence, that the probability of testing positive for WNV was correlated with moose age (adult males only), year of harvest, or county.

MNDNR will be continuing WNV surveillance in hunter-harvested moose for the next 3 years (2010-2013) in an attempt to determine if there is a year affect in the prevalence rate. Little is known about the effects of WNV in moose. In white-tailed deer (*Odocoileus virginianus*) it has been found that they often have a low titer and no clinical signs. However, the USDA has found that reindeer (*Rangifer tarandus*) infected with WNV have high mortality rates and high titers, indicating that the virus is more serious for some species than others.

Malignant Catarrhal Fever (MCF)

A total of 326 samples were submitted to NVSL for peroxidase-linked assay (PLA) testing for MCF. If the PLA test came back positive, the samples were screened with a VN test. A total of 114 samples tested positive on the PLA test (35%)(Figure 6). One of the 114 tested on VN came back positive (at 1:4). The PLA test is more sensitive than the virus isolation, meaning it is much better at identifying true positives. VN is more specific, which means it is better at identifying true negatives. There are a couple of issues with this testing. First, the PLA reacts with multiple Gammaherpes Viruses (such as the wildebeest strain, the sheep strain, the deer strain, etc). Second, a PLA positive does not indicate which strain has been found, only that one strain has been identified. The higher the positive value with the PLA test, the stronger the positive in the sample. The VN test only screens for the wildebeest strain (which is exotic to the U.S.) and would be negative if other strains are present. This means a sample that was positive on PLA and negative on VN was likely exposed to a gammaherpes virus, but not the wildebeest strain. The one positive result was a weak positive, and likely, a false positive.

Adult (92/255, 36.1%) and yearling males (11/27, 40.1%) had similar rates of exposure to MCF. Probability of MCF exposure was independent of age (adult males only) and we found only weak evidence that MCF varied by county. There were, however, large differences in estimated probability of MCF exposure by year (range: 3.7% in 2007 to 74.6% in 2008).

We have been collaborating with researchers to determine which strain of MCF the NE MN moose are being exposed to. To date, all attempts at strain-typing have been unsuccessful.

Gammaherpes viruses have been documented to cause serious illness and death in moose and other ruminants. The clinical symptoms can mimic *P. tenuis* infection as the animals often exhibit neurological deficits, go blind, and thrash on the ground prior to death. While infection with MCF frequently results in death, carrier status can occur and is identified with serology. Zarnke et al. 2002 found serologic evidence of exposure in numerous species across Alaska and reported 1% prevalence in moose.

Fecal Examination for Parasites

A total of 318 fecal samples were screened for evidence of parasites on fecal floatation. While no ova, oocysts, or cysts were observed in 277 samples, 41 of the samples had evidence of parasitic infection (12.9%). Parasites identified include strongyle-type ova, *Dictyocaulus*, *Moniezia*, and *Nematodirus*. Negative results do not necessarily mean the animal was parasite free, only that it was not actively shedding at the time the feces were collected.

Pulmonary *Mycoplasma* Culture

In 2007, 119 lung samples were submitted for *Mycoplasma* culture. This bacterium was not isolated on any of the samples. Culture efforts were discontinued in 2008.

Mycobacterium paratuberculosis (Johne's Disease)

We submitted 179 fecal samples for *M. paratuberculosis* culture in 2007 and 2008. All culture results were negative. This was discontinued in 2009. PCR was performed on 316 samples, with all results negative, and Biocor was run on 335 samples, with all of the results negative.

The negative fecal cultures and PCR results indicate that those moose were not actively shedding the bacterium. The negative Biocor results indicate that these animals had not been exposed to the bacterium.

All species of ruminants are believed to be susceptible to Johne's and it is frequently diagnosed in cattle and sheep (Manning and Collins 2001). Clinical signs in wild ruminants are similar to those seen in sheep, and 1 moose with diarrhea, which resulted in death, was diagnosed with Johne's (Soltys et al. 1967). Serologic evidence of exposure to Johne's in moose has been documented, with 9/426 (2.1%) seropositive moose in Norway (Tryland et al. 2004).

Anaplasmosis

A total of 319 samples were screened for Anaplasmosis (*Anaplasma phagocytopila*, formerly *Ehrlichia phagocytophila*) with the card test. Only 1 of moose was positive (1/319, 0.3%); indicating that exposure to this bacterium is likely occurring, albeit at a low rate.

Moose are known to be susceptible to infection with *A. phagocytophilum*. In Norway, anaplasmosis was diagnosed in a moose calf, which displayed apathy and paralysis of the hind-quarters (Jenkins et al. 2001). This moose was concurrently infected with *Klebseilla* pneumonia, to which the calf's death was attributed, though the *Klebseilla* infection was most likely secondary to and facilitated by the primary infection with *A. phagocytophilum* (Jenkins et al. 2001). In sheep, this disease produces significant effects on the immunological defense system, increasing their susceptibility to disease and secondary infections (Larsen et al. 1994).

A. phagocytophilum is known to occur in MN. In fact, from 1998-2005, 790 human cases were reported in MN and in recent years the MN Department of Health has documented an expansion in the areas in which MN residents are exposed to vector-borne diseases (MN Department of Health). The NE MN population of moose overlaps with the primary area of tick-borne disease risk determined by the MN Department of Health and NE MN.

Borreliosis (Lyme disease)

A total of 319 samples were screened for lymes disease with an immunofluorescence assay (IFA). Positive results were reported for 73 of the samples (22.9%, 95% CI: 18.6-27.9) (Figure 7). We found evidence of higher Borrelia exposure among yearling males (44.4%, 95% CI: 26.4-63.9) compared to adult males (19.8%, 95% CI: 15.2-25.2), but the estimate for yearlings was imprecise due to small sample size. The probability of Borrelia exposure among adult males was substantially lower in 2008 (2.2%, 95% CI: 0-5.2) compared to 2007 (34.8%, 95% CI: 23.5-46.0) or 2009 (28.8%, 95%CI: 18.8-38.7). Conversely, probability of exposure among adult males was independent of age, and there was only weak evidence of differences among counties.

Borreliosis is a tick borne bacterial disease that is maintained in a wildlife/tick cycle involving a variety of species, including mammals and birds. While evidence of natural infection in wildlife exists, there has been no documentation of clinical disease or lesions reported in wildlife species.

Brucellosis

A total of 303 samples were submitted for *Brucella* screening with the card test. There was only 1 positive result, which was then forwarded for confirmatory testing using rivanol agglutination (RIV). The RIV result was negative, indicating that the positive result from the card test was likely a false positive. These negative results indicate that moose were not likely exposed to the bacterium. While naturally occurring fatal *Brucella* infections have been documented in free-ranging moose (Honour and Hickling 1993) and serologic evidence suggests that some moose populations are being exposed to *Brucella sp*. (Zarnke 1983), evidence suggests that the prevalence is low (Honour and Hickling 1993).

Bovine Viral Diarrhea Virus (BVD) 1 & 2

A total of 333 samples were submitted for serum neutralization (SN) testing for BVD 1 & 2. Positive results were reported for 3 of the samples (1%); including 1 strong positive at a 1024/4096 titer. These results indicate that the moose population is being exposed to BVD at a very low rate.

BVD is considered a major disease of cattle and is thought to be the most common infectious cause of reproductive failure in beef herds in the western U.S. BVD is also considered a disease of wild ruminants such as moose, caribou (*Rangifer tarandus*), and deer. Some clinical signs of BVD include diarrhea, dehydration, fever, impaired vision and hearing, depression, abortions, and weakened neonates. Serologic evidence of BVD has been documented in 4 of 22 moose sampled in Alberta (Thorsen and Henderson 1971).

Bovine Herpes Virus 1 (BHV)

A total of 333 samples were screened for BHV using a SN test. Only 1 moose was found positive (0.9%). BHV is a disease of the respiratory tract. It is believed to infect all ruminant species and has been isolated from a large number of wild species. It is most commonly isolated in feedlot cattle.

Blue Tongue Virus (BTV)

A total of 334 samples were screened using a Competitive Enzyme-Linked Immunoabsorbent Assay (cELISA) for BTV. All results were negative.

BTV is a hemorrhagic disease transmitted by a biting midge that is known to cause illness and death in white-tailed deer. While it is known to be infective to a variety of domestic and wild ruminants, clinical disease is quite variable.

Epizootic Hemorrhagic Disease (EHD)

A total of 334 samples were screened for EHD using an Agar Gel Immuno Diffusion (AGID) test. All results were negative.

EHD is a hemorrhagic disease transmitted by a biting midge that is known to cause illness and death in white-tailed deer. While it is known to be infective to a variety of domestic and wild ruminants, clinical disease is quite variable.

Leptospirosis

A total of 334 samples were screened for 6 species of *Leptospira* using a microscopic agglutination test (MAT). Positive results per species are reported below:

- *L. bratislava*:
  - 6/334 (1.8%)
- *L. canicola*:
  - 2/334 (0.6%)
- *L. grippothyphosa*:
  - 8/334 (2.4%)
- *L. hardjo*:
  - 3/334 (0.9%)
- *L. interrogans* serovar *icterohaemorrhagicae*:
  - 22/334 (6.6%)
- *L. pomona*:
  - 23/334 (6.9%)

Leptospirosis is a bacterial disease that can infect a wide variety of mammals, both domestic and wild. Moose could be at an increased risk for Leptospirosis as it is often propagated by mud and water contaminated with urine, and moose are known to frequent these habitats.

Neospora sp.

A total of 334 samples were screened for *Neospora* with an ELISA test. Nine moose were found positive for this parasite (2.7%).

While clinical disease due to infection is best described in domestic animals, reports of ill effects due to *Neospora* infection in wildlife do exist. Systemic neosporosis was diagnosed in a California black-tailed deer (*Odocoileus hemionus*) that was found dead (Woods et al., 1994) and the parasite was identified in the brain of a full-term stillborn deer from a zoo in France (Dubey et al., 1996). Antibodies to Neospora have been found in numerous species of wildlife, including 8/61 moose from NE MN (Gondim et al. 2006).

Parainfluenza Virus 3 (PI3)

A total of 232 samples were screened for PI3 using a hemagglutination inhibition (HI) test in 2007 and 2008. There was 1 positive moose (10.4%).

The positive result indicates that NE MN moose are being exposed to PI3, although at a very low rate. Domestic ruminants are considered the main source of infection for free-ranging ruminants. However, studies of white-tailed deer, which were geographically isolated from livestock, indicate that large wild ruminant populations can maintain PI3 and latency of the viruses allows them to be maintained in a restricted host population for a long period (Sadi et al. 1991).

Chronic Wasting Disease (CWD)

A total of 87 obex samples and 88 retropharyngeal lymph nodes were screened for CWD using immunohistochemistry (IHC). All results were negative. CWD is a transmissible spongiform encephalopathy (TSE) that causes neurological disease in cervids. CWD is known to occur in moose, but has never been documented in wild cervids in MN.

Bovine Tuberculosis

Cranial lymph nodes (parotid, retropharyngeal, and submandibular) from 88 moose were collected and cultured for *Mycobacterium bovis*. All results were negative.

Bovine tuberculosis is a chronic, progressive bacterial disease that infects a wide array of mammals. Bovine tuberculosis has been found in wild white tailed deer in a small, localized area in northwestern MN, but has not been found in any wild animals within the moose hunt permit areas.

Brain Histopathology

In 2008 and 2009, MNDNR collected whole brains from moose at registration stations. Brains were formalin-fixed and submitted for histological examination. A total of 47 whole brains were collected. Four complete coronary brain, cerebellum, and brain stem sections were processed for histological examination from each moose. An average of 25 histological slides per animal were examined. Areas examined included the frontal, temporal, parietal, and occipital lobes and the basal nuclei, thalamus, mesencephalon, and brain stem. This examination is meant to help identify lesions consistent with migration tracts (presumably due to *P. tenuis*) that may be present in brains of apparently healthy animals. No lesions were found in 41 of the brains, 5 had lymphocytic infiltration (unspecific chronic inflammatory lesion), and 1 had larval tracts present in the white matter (with mild to moderate meningitis, axonal degeneration, and secondary demyelination). MNDNR will continue the collection of whole brains of moose at registration stations in 2010-2013.

Whole Liver Evaluation

In 2009 only, hunters were asked to collect whole livers. A total of 57 livers were submitted for gross examination. The purpose of this is to develop a ranking system to evaluate liver fluke load and damage caused by liver flukes. The ranking system that was developed is as follows: no fluke induced lesions (no evidence of fluke migration), mild infection (approximately less than 15% of liver liver parenchyma is affected with mild prominence/fibrosis of bile ducts and few smaller nodules characterized by peripheral fibrosis and central presence of opaque brown pasty material), moderate infection (approximately 15-50% of the liver parenchyma affected by nodules and fibrosis), and marked infection (approximately 51-100% of the liver parenchyma affected with deformation of the entire liver by larger nodules with widespread fibrosis). Of the 57 livers examined, 34 had no fluke induced lesions, 15 had mild infection, 6 had moderate infection, and 2 had marked infection. Collection of whole livers will continue in 2010-2013. In addition, serum will be submitted for a serum chemistry profile in an attempt to correlate serum liver enzyme levels with the level of fluke induced damage.

Hepatic Mineral Values

Frozen liver samples were submitted for analysis of mineral values. A total of 293 samples were digested by wet ash and analyzed using inductively coupled plasma atomic emissions (ICPAES) spectroscopy. There was a change in diagnostic laboratory in 2009, thus some additional screening were performed on a subset of the sample. As a result, all 293 samples were analyzed for cadmium, arsenic, copper, iron, magnesium, manganese, molybdenum, lead, selenium, and zinc levels, while only 100 samples were analyzed for barium, calcium, boron, chromium, mercury, antimony, thallium, potassium, sodium, and phosphorus levels (Table 3). All results for arsenic, boron, chromium, mercury, antimony, selenium, and thallium were below the detectable threshold. While these results have not been fully evaluated, it is clear that some of the moose tested had deficient copper levels.

ACKNOWLEDGMENTS

This project would not have been possible without assistance from a number of MNDNR employees, tribal biologists, and volunteers. We would like to especially thank our crews that worked moose registrations stations: Tom Rusch, Jeff Hines, Dave Ingebrigsten, Bob Kirsch, Nancy Gellerman, Walt Gessler, Dan Litchfield, David Pauly, Martha Minchak, Penny Backman, Kevin Carlisle, Margaret Dexter, and Julie Adams; as well as Julie Adams for making our area maps. We'd also like to thank tribal biologists Mike Schrage, Andy Edwards, Seth Moore, and Lance Overland for their help with the project. A big thanks to John Giudice for his assistance with the preliminary data analysis. Finally, we appreciate the help of volunteers Andrea Widdel, Melissa Wolfe, and MacDonald Farnham.

REFERENCES

Dubey, J. P., J. Rigoulet, P. Lagourette, C. George, L. Longeart, J. L. LeNet. 1996. Fatal transplacental neosporosis in a deer (*Cervus eldi siamensis*). The Journal of Parasitology 82(2): 338-339.

Fischer, S., E. Weiland, and K. Froelich. 1998. Characterization of a bovine viral diarrhea virus isolated from roe deer in Germany. Journal of Wildlife Diseases 31:47-55.

Gondim, L. F. P. 2006. *Neospora caninum* in wildlife. Trends in Parasitology 22(6): 247-252.

Honour, S., and K. M. H. Hickling. 1993. Naturally occurring *Brucella suis* biovar 4 infection in a moose (*Alces alces*). Journal of Wildlife Diseases 29(4): 596-598.

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(1): 201-203.

Larsen, H. J. S., G. Overnes, H. Waldeland, and G.M. Johansen. 1994. Immunosuppression in sheep experimentally infected with *Ehrlichia phagocytophila*. Research in Veterinary Science 56: 216-224.

Lenarz, M. S. 2009. 2009 Aerial moose survey. Minnesota Department of Natural Resources, St. Paul, MN, USA <http://files.dnr.state.mn.us/recreation/hunting/moose/moose_survey_2009.pdf>

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(5):000-000.

Lenarz, M. S., M.W. Schrage, A.J. Edwards, and M.E. Nelson. 2007. Moose population dynamics in northeastern Minnesota. Pp. 346-348 in Summaries of wildlife research findings, 2005 (M.W. DonCarlos, R.O. Kimmel, J.S. Lawrence, and M.S. Lenarz, eds.). Minnesota Department of Natural Resources, St. Paul.

Manning, E. J. B., and M. T. Collins. 2001. *Mycobacterium avium* subsp. *paratuberculosis:* pathogen pathogenesis and diagnosis. *In* Mycobacterial infections in domestic and wild animals, E. J. B. Manning and M. T. Collins (eds.). Revue Scientifique et Technique Office International des Epizooties 20: 133-150.

Sadi, L., R. Loyel, M. St. George, and L. Lamontagu. 1991. Serologic survey of white-tailed deer on Anticosti Island, Quebec for bovine herpesvirus 1, bovine viral diarrhea, and parainfluenza 3. Journal of Wildlife Diseases 27:569-577.

Soltys, M. A., C. E. Andress, and A. L. Fletch. 1967. Johne's disease in a moose (*Alces alces*). Bulletin of Wildlife Disease Association 3: 183-184.

Thorsen J. and J.P. Henderson. 1971. Survey for antibody to infectious bovine rhinotracheitis (IBR), bovine virus diarrhoea (BVD) and parainfluenza 3 (PI3) in moose sera. Journal of Wildlife Diseases 7: 93–95.

Tryland, M., I. Olsen, T. Vikoren, K. Handeland, J. M. Arnemo, J. Tharaldsen, B. Djonne,T. D. Josefsen, and L. J. Reitan. 2004. Serologic survey for antibodies against *Mycobacterium avium* subsp. *paratuberculosis* in free-ranging cervids from Norway. Journal of Wildlife Diseases 40(1): 32-41.

Woods, L. W., P. K. Swift, B. C. Barr, M. C. Horzinek, R. W. Nordhausen, M. N. Oliver, K. R. Jones, and N. J. Maclachlan. 1996. Systemic adenovirus infection associated with high mortality in mule deer (Odocoileus hemionus) in California. Veterinary Pathology 33: 125-132.

Zarnke, R. L. 1983. Serological survey for selected microbial pathogens in Alaskan wildlife. Journal of Wildlife Disease 19: 324-329.

Zarnke, R.L. 2000. Alaska wildlife serologic survey, 1975-2000. Alaska Department of Fish and Game. Federal Aid in Wildlife Restoration. Research Final Report. Grants W-24-5 and W-27-1 through W-27-4. Study 19.71. Juneau, Alaska.

Zarnke, R.L., H. Li, and T.B. Crawford. 2002. Serum antibody prevalence of malignant catarrhal fever viruses in seven wildlife species from Alaska. Journal of Wildlife Diseases 38(3):500-504.

Table 1. Age/Sex distribution of samples collected by year in northeastern Minnesota as part of the moose herd health assessment (excludes animals were age/sex data were incomplete).

Sex/Age	2007	2008	2009	All
Female calf	0	0	0	0
Female yearling	1	0	2	3
Female adult	2	1	1	4
Male calf	0	0	1	1
Male yearling	10	6	12	28
Male adult	97	102	90	289
All	110	109	106	325

Table 2. Serology results from the moose herd health assessment, 2007-2009, northeastern Minnesota.

Serological Test	Number Tested	Number Positive	Percent Positive	95% Lower Confidence Limit	95% Upper Confidence Limit
EEE	330	20	6.1	3.9	9.3
WNV	330	115	34.8	29.9	40.2
L. bratislava	334	6	1.8	0.7	4.0
L. canicola	334	2	0.6	0.0	2.3
L. grippothyphosa	334	8	2.4	1.1	4.8
L. hardjo	334	3	0.9	0.2	2.8
L. interrogans* serovar *icterohaemorrhagicae	334	22	6.6	4.3	9.9
L. pomona	334	23	6.9	4.6	10.2
M. paratuberculosis	335	0	0.0	0.0	0.9
MCF	326	114	35.0	30.0	40.3
Anaplasma	319	1	0.3	0.0	2.0
Borrelia	319	73	22.9	18.6	27.9
Brucella	303	1	0.3	0.0	2.1
BTV	334	0	0.0	0.0	0.9
BVD 1&2	333	3	0.9	0.2	2.8
EHD	334	0	0.0	0.0	0.9
BHV	333	1	0.3	0.0	1.9
Neospora	334	9	2.7	1.4	5.2
PI3	335	1	0.3	0.0	1.9

Table 3. Results of hepatic mineral analysis of hunter-harvested moose, 2007-2009, northeastern Minnesota. All liver values are reported in parts per million (ppm).

Mineral	Number Tested	Mean	Medium	Minimum	10th Percentile	90th Percentile	Max	SD
Barium	100	0.23	0.16	0.02	0.08	0.41	1.70	0.25
Calcium	100	65.9	53.1	33.4	43.5	75.9	943.0	90.1
Cadmium	293	2.01	1.80	0.05	0.83	3.48	10.40	1.24
Copper	293	66.3	62.5	0.4	32.2	106.0	346.0	35.2
Iron	293	218.0	180.0	35.8	114.1	285.9	2526.0	210
Potassium	100	2617.0	2687.0	1731.0	2217.1	2963.7	3205.0	301.9
Magnesium	293	155.56	159.30	2.70	128.54	182.88	238.60	27.74
Manganese	293	2.92	2.93	0.09	1.60	4.10	7.20	1.04
Molybdenum	293	0.81	0.80	0.10	0.50	1.10	1.40	0.26
Sodium	100	858.6	805.0	482.0	581.3	1220.7	1650.0	244.5
Phosphorous	100	3921.1	4096.5	302.0	3206.4	4651.6	4979.0	833.9
Lead	293	0.63	0.50	0.25	0.25	0.50	29.30	2.26
Zinc	293	71.6	61.4	2.1	25.7	125.3	264.0	42.7

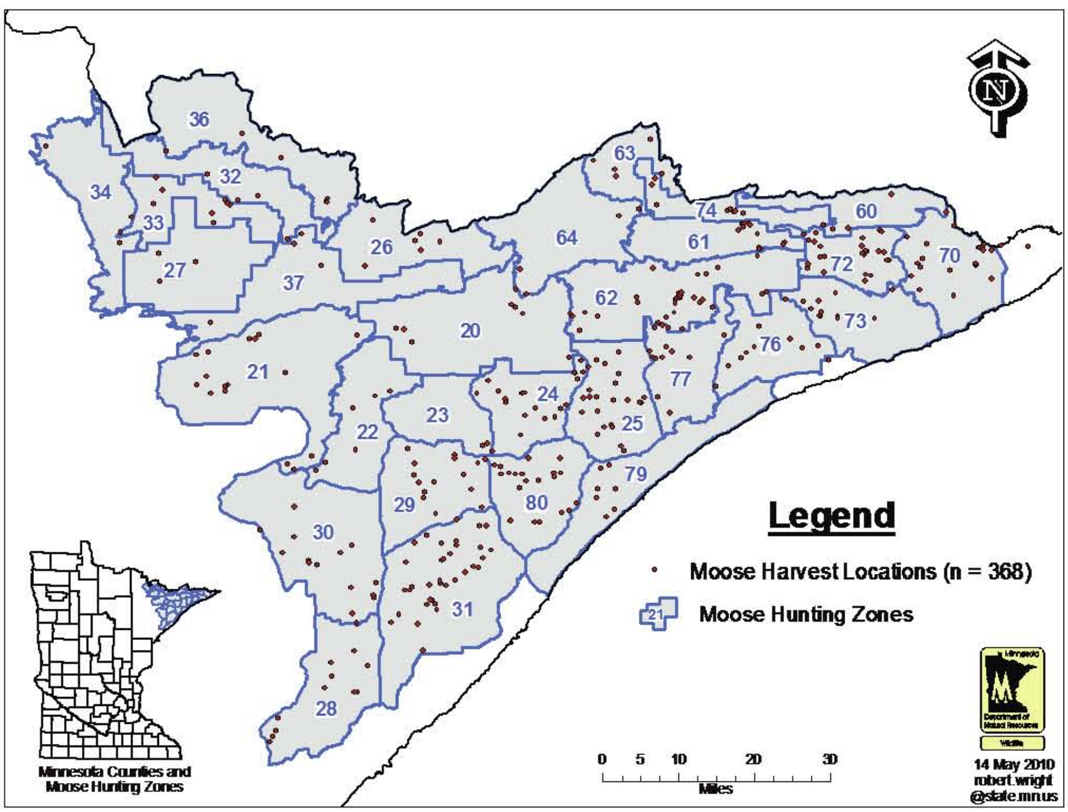


Figure 1. Harvest locations of hunter-harvested moose (n = 338) included in the 2007-2009 health assessment project, northeastern Minnesota.

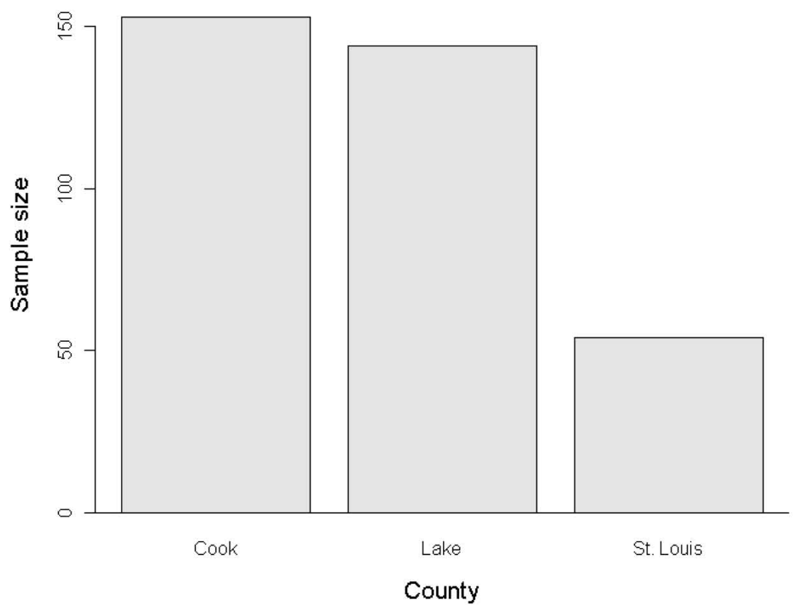


Figure 2. County of harvest location for hunter-harvested moose (n = 338) included in the 2007-2009 health assessment project, northeastern Minnesota.

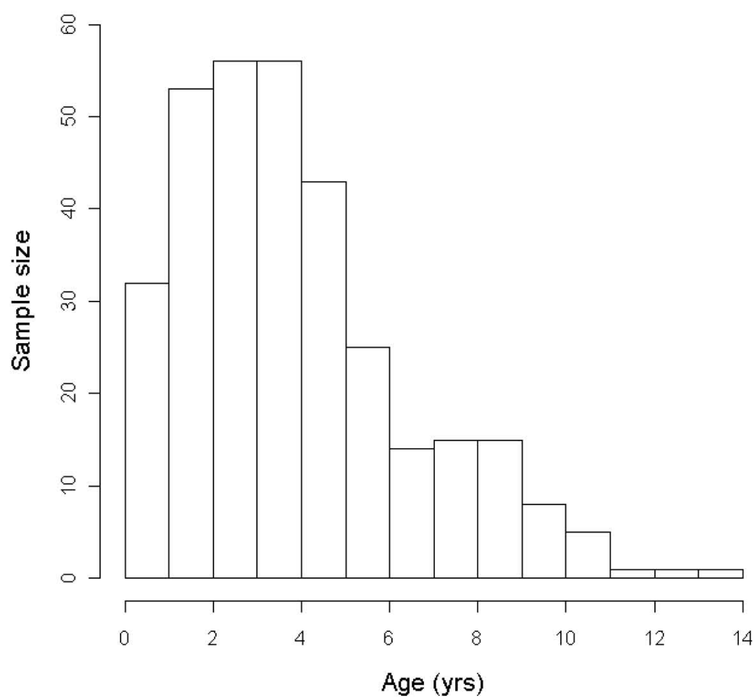


Figure 3. Age distribution of hunter-harvested moose (n = 338) included in the 2007-2009 health assessment project, northeastern Minnesota.

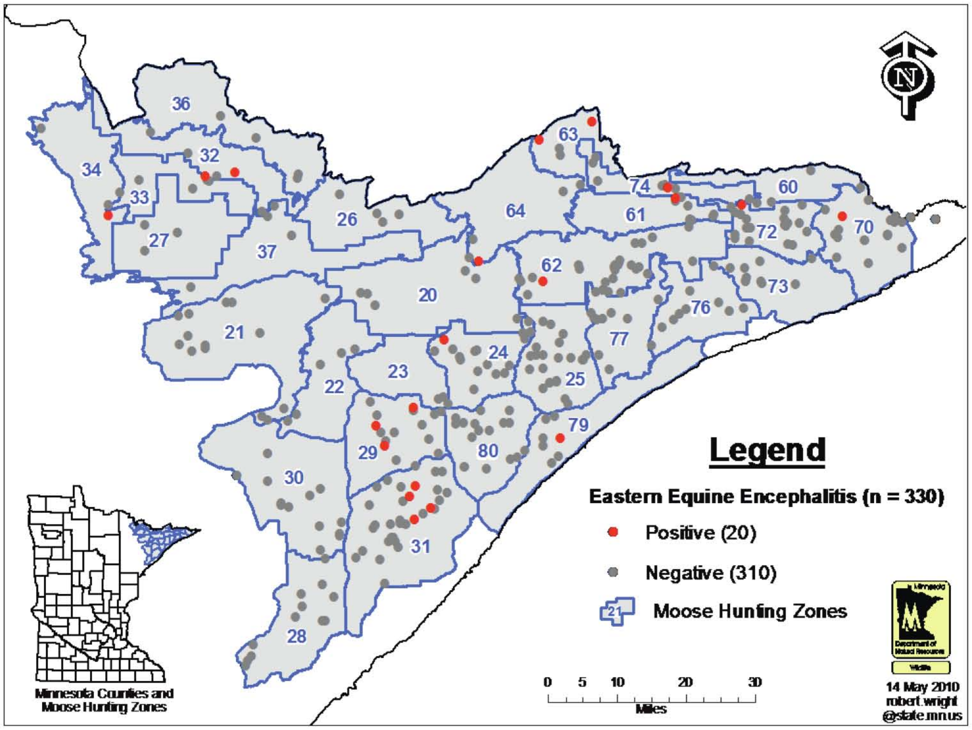


Figure 4. Harvest locations of moose (n = 20) that tested positive for Eastern Equine Encephalitis from 2007-2009, northeastern Minnesota.

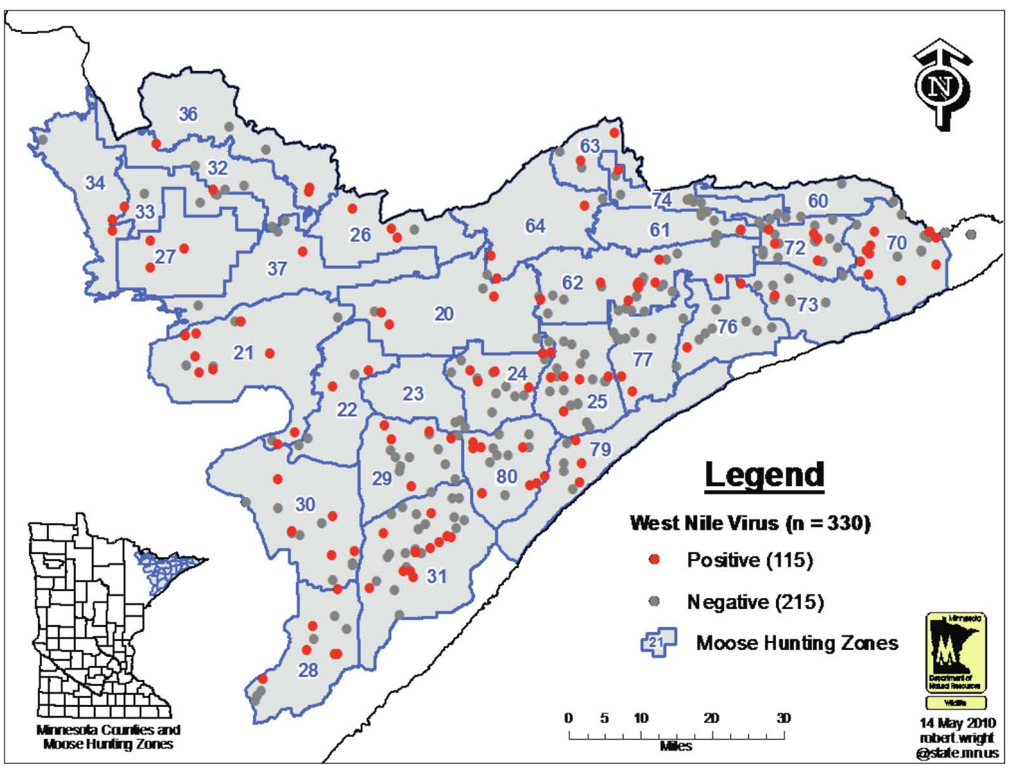


Figure 5. Harvest locations of moose (n = 115) that tested positive for West Nile Virus from 2007-2009, northeastern Minnesota.

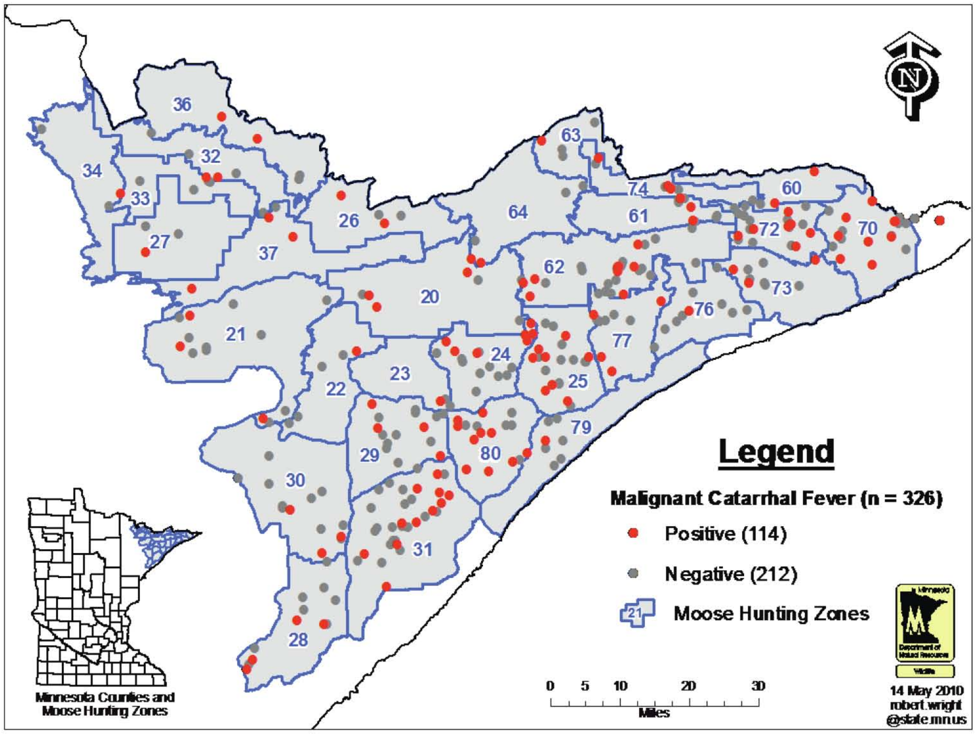


Figure 6. Harvest locations of moose (n = 114) that tested positive for Malignant Catarrhal Fever from 2007-2009, northeastern Minnesota.

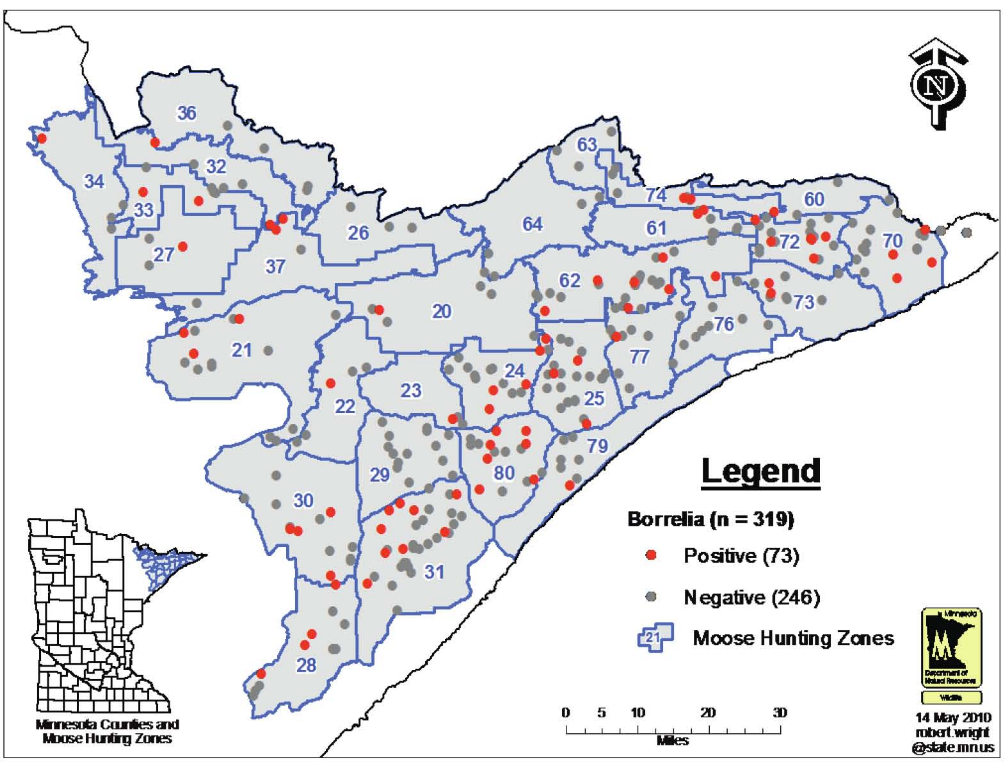


Figure 7. Harvest locations for moose that tested positive for Borrelia from 2007-2009, northeastern Minnesota.

MANAGING BOVINE TUBERCULOSIS IN WHITE-TAILED DEER IN NORTHWESTERN MINNESOTA: A 2009 PROGRESS REPORT

Michelle Carstensen[1], Erika Butler, Erik Hildebrand, and Lou Cornicelli

SUMMARY OF FINDINGS

Bovine tuberculosis (TB), first detected in Northwestern Minnesota in 2005, has since been found in 12 cattle operations and 27 free-ranging white-tailed deer (*Odocoileus virginianus*). Both deer and cattle have the same strain of bovine TB, which has been identified as one that is consistent with the disease found in cattle in the southwestern United States and Mexico. The Board of Animal Health (BAH) has been leading efforts to eradicate the disease in Minnesota's cattle, which have included the depopulation of all infected herds, a buy-out program that removed 6,200 cattle from the affected area, and mandatory fencing of stored feeds on remaining farms. In response to the disease being detected in cattle, the Minnesota Department of Natural Resources (MNDNR) began surveillance efforts in free-ranging white-tailed deer within a 15-mile radius of the infected farms in fall 2005. To date, 27 deer have been found infected with bovine TB, and nearly all (n = 26) infected deer were sampled within a 164 mi^2 area, called the Bovine TB Core, which is centered in Skime, Minnesota, and encompasses 8 of the previously infected cattle farms. The 27^{th} case of bovine TB in deer was detected in November 2009, and was located 2.2 miles west of the original Core boundary. In total, 1,488 hunter-harvested deer were tested for bovine TB in northwest Minnesota during fall 2009, with only 1 positive case detected (apparent prevalence <0.07%). An aerial survey estimated the population of the Core to be 422 ± 126 deer in January 2010, and also detected a significant concentration of deer (>360 deer) in a 23mi^2 area to the west of the Core boundary and within a 2mi radius of the most recent TB-positive deer. To further reduce deer numbers in the Core and this western expansion area, MNDNR conducted targeted removal operations using ground sharpshooting from February-April, 2010; 450 additional deer were removed. None of these deer had any clinical evidence of bovine TB infection at the time of sampling, yet final test results are pending. Further, a recreational feeding ban, instituted in November 2006 in a 4,000mi^2 region in northwestern MN to help reduce the risk of deer to deer transmission of the disease, remains in effect. Under an agreement among the United States Department of Agriculture (USDA), BAH, and MNDNR, hunter-harvested deer surveillance will continue for at least the next 5 years to monitor infection in the local deer population, and any further aggressive management actions (e.g., sharpshooting deer in key locations) will be dependent on future surveillance results.

INTRODUCTION

Bovine tuberculosis is an infectious disease that is caused by the bacterium *Mycobacterium bovis (M. bovis)*. Bovine TB primarily affects cattle; however, other mammals may become infected. Bovine TB was first discovered in 5 cattle operations in northwestern Minnesota in 2005. Since that time, 2 additional herds were found infected in 2006, 4 more in 2007, and 1 in 2008; resulting in further reduction of the state's Bovine TB accreditation to Modified Accredited in early 2008. By fall 2008, Minnesota was granted a split-state status for TB accreditation that maintained only a small area (2,670mi^2) in northwestern Minnesota as "Modified Accredited," allowing the remainder of the state to advance to "Modified Accredited Advanced." To date, 27 wild deer have been found infected with the disease in northwestern MN. Although Bovine TB was once relatively common in U.S cattle, it has historically been a very rare disease in wild deer. Prior to 1994, only 8 wild white-tailed and mule deer (*O. hemionus)* had been reported with Bovine TB in North America. In 1995, Bovine TB was detected in wild deer in Michigan. Though deer in Michigan do serve as a reservoir of Bovine

[1] Corresponding author e-mail: michelle.carstensen@state.mn.us

TB, conditions in northwestern Minnesota are different. Minnesota has no history of tuberculosis infection in deer or other wildlife, and the *M. bovis* strain isolated from the infected Minnesota herd does not match that found in Michigan. Also, there are much lower deer densities in the area of the infected herds than in the affected areas of Michigan. Further, unlike Michigan, Minnesota does not allow baiting (hunting deer over a food source), which artificially congregates deer and increases the likelihood of disease transmission.

Bovine TB is a progressive, chronic disease. It is spread primarily through the exchange of respiratory secretions between infected and uninfected animals. This transmission usually happens when animals are in close contact with each other. Animals may also become infected with Bovine TB by ingesting the bacteria from eating contaminated feed. It can take months to years from time of infection to the development of clinical signs. The lymph nodes in the animal's head usually show infection first and as the disease progresses, lesions (yellow or tan, pea-sized nodules) will begin to develop on the surface of the lungs and chest cavity. In severely infected deer, lesions can usually be found throughout the animal's entire body. Hunters do not always readily recognize small lesions in deer, as they may not be visible when field dressing deer. In fact, most infected deer appear healthy. In Michigan, only 42% of the Bovine TB positive deer had lesions in the chest cavity or lungs that would be recognized as unusual by most deer hunters. While it is possible to transmit Bovine TB from animals to people, the likelihood is extremely low. Most human tuberculosis is caused by the bacteria *M. tuberculosis*, which is spread from person to person and rarely infects animals.

METHODS

In 2009, a fall hunter-harvested surveillance strategy was developed to meet the sampling goals established in a recent Memorandum of Understanding (MOU) with USDA, signed by both MNDNR and BAH, that required 1,500 deer to be tested for Bovine TB within the newly created Modified Accredited Zone (MAZ), and 300 deer to the immediate south and west of the MAZ boundaries.

At the registration stations, hunters were asked to voluntarily submit lymph node (LN) samples for Bovine TB testing. Hunter information was recorded, including the hunter's name, address, telephone number, MNDNR number, and location of kill. Maps were provided to assist the hunters in identifying the location (Township, Range, Section, and Quarter-section) of the kill. Cooperating hunters were given a cooperator's patch and entered into a raffle for a firearm donated by the Minnesota Deer Hunter's Association.

Tissue collection procedures included a visual inspection of the chest cavity of the hunter-killed deer. Six cranial LNs (parotid, submandibular, and retropharyngeal) were visually inspected for presence of lesions and extracted for further testing. Samples were submitted to the Veterinary Diagnostic Laboratory (VDL) at the University of Minnesota for histological examination and acid-fast staining. All samples were then pooled in groups of 5 and sent to the National Veterinary Services Laboratories (NVSL) in Ames, IA for culture. Any suspect carcasses (e.g., obvious lesions in chest cavity or head) were confiscated at the registration stations and the hunter was issued a replacement deer license at no charge. Suspect carcasses were transported in their entirety to the VDL for further testing.

Additionally, MNDNR implemented efforts to further reduce deer numbers in the post-hunting season in the Bovine 164mi^2 TB Core Area, through the use of sharpshooters. During winters 2006 through 2009, sharpshooting from the ground was conducted by USDA-Wildlife Services (USDA-WS) professionals; supplemental sharpshooting was conducted by aerial operations during winters 2007 and 2008. Sharpshooter-harvested deer were transported intact to a central processing facility at Thief Lake Wildlife Management Area. Sample collection and handling was similar to that described above. Carcasses that were free of any visible lesions were salvaged for venison and made available to the public.

Prior to the start of the each winter sharpshooting effort, MNDNR conducted aerial surveys of the Bovine TB Core Area to assess deer numbers and distribution (Figure 1). This information was used to guide sharpshooting activities and estimate the percentage of deer removed from the area.

RESULTS AND DISCUSSION

In fall 2009, we collected 1,488 samples from hunter-harvested deer; 947 samples from within the Modified Accredited Zone and 541 samples outside the zone (Figure 2). An enlarged and abscessed retropharyngeal lymph node was observed from 1 deer (3.5-yr-old male) while being sampled at a registration station; bovine TB was later confirmed by NVSL. This TB-positive male was harvested 2.2 miles west of the Core boundary, and marked the first incidence of the disease being detected in a new area. However, we do not believe this signifies movement of the disease, as this buck may have had a homerange that spanned the boundaries of the Core. Further, all the past TB-positive deer (*n* = 26) would have been born on or prior to 2005, when the disease was first detected in cattle, lending itself to our "Alive in '05" theory of the spillover event. This newest case was born in 2006 and falls outside our spillover theory; however, additional cattle farms were detected TB-positive in 2006, 2007, and 2008, which could have been the exposure source for this individual deer or transmission may have occurred from another TB-infected deer.

Testing of remaining lymph node samples at NVSL has confirmed that there were no additional positive cases detected during the fall 2009 surveillance, resulting in an apparent prevalence rate of <0.07%. However, the fall sampling effort fell 553 samples (37%) short of its collection goal of 1,500 samples inside the MAZ; thus additional deer removal efforts in winter 2009–10 increased the sampling total by 450 deer, or 9% shy of the goal. Apparent prevalence of Bovine TB in the local deer population, sampled throughout a 1,730 to 2,670mi^2 Surveillance Zone, indicates a significant decreasing trend from 2006–2009 (Table 1, Figure 3).

To supplement the number of samples collected through fall hunter-harvested surveillance and to further reduce deer density in the area where TB-positive deer had been confirmed, MNDNR contracted with USDA-Wildlife Services for ground sharpshooting in the Bovine TB Core Area, and a 23mi^2 western expansion area, during February–April 2010. In total, 450 deer were removed from the TB Core Area, included approximately 150 from the western expansion area (Figure 4). No obvious clinical signs of bovine TB were observed on any of the deer; however, final test results are pending. Disease prevalence in the TB Core Area has decreased dramatically from 2007 to 2010 (Table 1, Figure 3). Although disease prevalence estimates in the TB Core Area are biased due to the limited geographic distribution of TB-positive deer and the increased probability of detecting a positive individual, the decreasing trend is consistent with the large-scale surveillance of the local deer populations in the fall.

Aerial survey results from January 2010 estimated that the deer population in the Bovine TB Core Area was a minimum of 422 ± 126 deer (Figure 1). This was significantly less than the February 2009 population estimate of 664 ± 87 (Figure 5, Table 2). It is apparent that aggressive deer removal in the TB Core Area through liberalized hunting, disease management permits, landowner shooting permits, and targeted sharpshooting has been able to reduce the deer population in this 164mi^2 area by approximately 55% since 2006 (Figure 5, Table 2). It is likely that that the TB Core Area is home to both migratory and resident deer, some of which may move out of the zone to spring-summer-fall or winter ranges during the year. It is further likely that deer from the surrounding area are immigrating into the TB Core Area as deer numbers are reduced and habitat availability increases. The lack of severe winter weather condition in recent years has also allowed for good overwinter survival, increased reproduction, and recruitment into the local deer population.

The proximity of the TB-infected deer to infected cattle herds, the strain type, and the fact that disease prevalence (<0.1%) is low, supports our theory that this disease spilled-over from cattle to wild deer in this area of the state. To date, we have sampled 8,144 deer in the northwest since 2005, and a total of 27 confirmed culture-positive deer (Figure 6). Further, the lack of infected yearlings or fawns and limited geographic distribution of infected adults further supports that this disease is not being spread efficiently in the local deer population.

In November 2006, a ban on recreational feeding of deer and elk was instituted over a 4,000mi^2 area to help reduce the risk of disease transmission among deer and between deer and livestock (Figure 7). Enforcement officers continue to enforce this rule and compliance is thought to be very high within the Bovine TB Management Zone.

Further, the Minnesota State Legislature passed an initiative in 2008 that allocated funds to buy-out cattle herds located in the Bovine TB Management Zone, spending $3 million to remove 6,200 cattle from 46 farms by January 2009; resulting in the discovery of the 12th infected cattle herd. The remaining cattle farms in the TB-endemic area (n = 27) were required to erect deer-exclusion fencing to protect stored forage and winter feeding areas, costing an additional $690,000 in state funds.

As part of the requirements to regain TB-Free accreditation, USDA has required BAH to test all cattle herds within the Modified Accredited Zone annually, with additional movement restrictions for farms located within the Bovine TB Management Zone. BAH has submitted an application for status upgrade to USDA, and a decision is expected by October 2010. The MNDNR is committed to assisting BAH in regaining Minnesota's TB-Free status as soon as possible. To accomplish this, the MNDNR will continue to conduct fall surveillance annually until 5 consecutive years with no TB-positive deer can be achieved, which would indicate that the disease was either eradicated or present in undetectable levels in the local deer population.

SURVEILLANCE COSTS

Conducting a disease surveillance effort that spans a large area and encompasses numerous deer registration stations requires a large, trained work force and a significant amount of expenditures to support the effort. The bovine TB surveillance effort in fall 2009 spanned the first 10 days of regular firearms season, and included both an early antlerless (October 2009) and special late (January 2010) hunt (special hunt surveillance relied on head drop boxes for sample submission). The number of stations staffed each weekend ranged from 10 to 23, and summed to 41 stations over the duration of the project. In total, 44 trained MNDNR staff, 15 USDA disease biologists, and 46 student workers were needed in the effort. Costs associated to the surveillance effort are listed in Table 3.

ACKNOWLEDGMENTS

There is no way to complete a project of this scale without the assistance and leadership from St. Paul and regional staff, including Ed Boggess, Dennis Simon, Dave Schad, Paul Telander, John Williams, and Mike Carroll. For all the help with field collections, we'd like to thank area staff from Thief Lake, Red Lake, Norris Camp, and Thief River Falls, Erik Hildebrand (Wildlife Health Specialist), Margaret Dexter (Fish and Wildlife Specialist), as well as students and faculty from the University of Minnesota, College of Veterinary Medicine. Randy Prachar provided excellent leadership through the sharpshooting operations, and we appreciate his efforts to support our carcass examination crews. Special thanks to Mary Reiswig for coordinating the venison donation program. Also thanks to Bob Wright, Erik Hildebrand, and the enforcement pilots (Tom Pfingsten and John Heineman) for conducting a deer survey within the Bovine TB Management Zone, as well as identifying illegal deer feeding activities. Also thanks to John Giudice for analyzing the survey data. We had an excellent team of GIS support including Steve Benson, Julie Adams, Bob Wright, and Chris Scharenbroich. We also want to recognize the tremendous amount of work conducted by USDA-Wildlife Services staff, led by John Hart (Grand Rapids) and disease biologist Paul Wolf (St. Paul). USDA-WS also loaned us disease biologists to assist with sample collections, including Darren Bruning (WA), Carl Betsill (NC), Tim White (IL), Barb Bodenstein (WI), Tom Hutton (MO), Jason Kloft (KS), and Tony Musante (NJ).

REFERENCES

Miller, R., J. B. Kaneene, S. D. Fitzgerald, and S.M. Schmitt. 2003. Evaluation of the influence of supplemental feeding of white-tailed deer (*Odocoileus virginianus*) on the prevalence of Bovine tuberculosis in the Michigan wild deer population. Journal of Wildlife Diseases 39: 84-95.

Minnesota Bovine Tuberculosis Management Plan. 2006. Minnesota Department of Agriculture, Minnesota Department of Natural Resources, Minnesota Board of Animal Health, United States Department of Agriculture, Unpubl. Rept.

O'Brien, D. J., S. D. Fitzgerald, T. J. Lyon, K. L. Butler, J. S. Fierke, K. R. Clarke, S. M. Schmitt, T. M. Cooley, and D. E. Berry. 2001. Tuberculous lesions in free-ranging white-tailed deer in Michigan. Journal of Wildlife Diseases 37: 608-613.

O'Brien, D. J., S. M. Schmitt, J. S. Fierke, S. A. Hogle, S. R. Winterstein, T. M. Cooley, W. E. Moritz, K. L. Diegel, S. D. Fitzgerald, D. E. Berry, and J. B. Kaneene. 2002. Epidemiology of Mycobacterium bovis in free-ranging whitetailed deer, Michigan, USA, 1995-2000. Preventive Veterinary Medicine 54: 47-63.

Schmitt, S. M., S. D. Fitzgerald, T. M. Cooley, C. S. Bruning-Fann, L. Sullivan, D. Berry, T. Carlson, R. B. Minnis, J. B. Payeur, and J. Sikarskie. 1997. Bovine tuberculosis in free-ranging white-tailed deer from Michigan. Journal of Wildlife Diseases 33: 749-758.

Table 1. Number of deer sampled for bovine TB and testing results listed by sampling strategy, fall 2005 to spring 2010, northwestern Minnesota.

Sampling strategy	2005	2006	2007	2008	2009	2010	Totals
Hunter-harvested (Oct-Jan)	474	942	1,166	1,246	1,488	n/a	5,316
# TB-positive	1	5	5	0	1		
Apparent Prevalence	0.21%	0.53%	0.43%	0.0%	0.07%		
Sharpshooting (Feb-May)	0	0	488	937	738	450*	2,613
# TB-positive			6	6	2		
Apparent Prevalence			1.23%	0.64%	0.27%		
Landowner/Tenant	0	90	0	125	0	0	215
# TB-positive		1		0			
Total Deer Tested	474	1,032	1,654	2,308	738	450	8,144
Total # TB-positive	1	6	11	6	3		27

*Final culture results from winter 2010 sampling are still pending at NVSL.

Table 2. Population estimates of deer within the Bovine TB Core Area, 2007–2010, northwest Minnesota.

2007-2010 population estimates[a] and asymptotic confidence intervals (95% CI)[b]

Year	Design	Var.est	n	N	Srate	Svar	SE	Xbar	SE	95% CI	**PopEst**	SE	**95% CI**	CV(%)	RP(%)
2007	StRS3	SRS	72	164	0.439			5.7	0.46	4.9-6.5	**935**	76.0	**784-1086**	8.1	16.2
2008	GRTS.SRS	Local	72	164	0.439	21.94	4.526	4.9	0.56	3.8-6.0	**807**	75.2	**659-954**	9.3	18.3
2009	GRTS.StRS2	Local	79	164	0.482	20.63	2.559	4.1	0.27	3.5-4.6	**664**	44.4	**577-751**	6.7	13.1
2010	GRTS.SRS	Local	72	164	0.439	29.30	6.701	2.6	0.39	1.8-3.3	**422**	64.4	**296-548**	15.3	30.0

[a]**Population estimate = estimated *minimum* number of deer present during the sampling interval.** Estimates are not adjusted for detectability (but intensive survey is designed to minimize visibility bias) and deer movement between sample plots is assumed to be minimal or accounted for via survey software.

[b]95% CI's are based on sampling variance only (adjusted for spatial correlation in 2008-2010); they do not include uncertainty associated with detectabilty or animal movements (temporal variation due to animals moving onto or off the study area).

Table 3. Expenditure details for 2009 Minnesota bovine tuberculosis surveillance program.

Expenditure	Total cost
MNDNR staff salary	$176,300
MNDNR staff travel expenses	30,950
Veterinary student labor & travel expenses	$42,815
Other student labor & travel expenses	$16,500
Supplies	$18,775
Fleet	$23,750
Diagnostic fees	$52,350
USDA contract for sharpshooting	$210,000
Total	$571,440

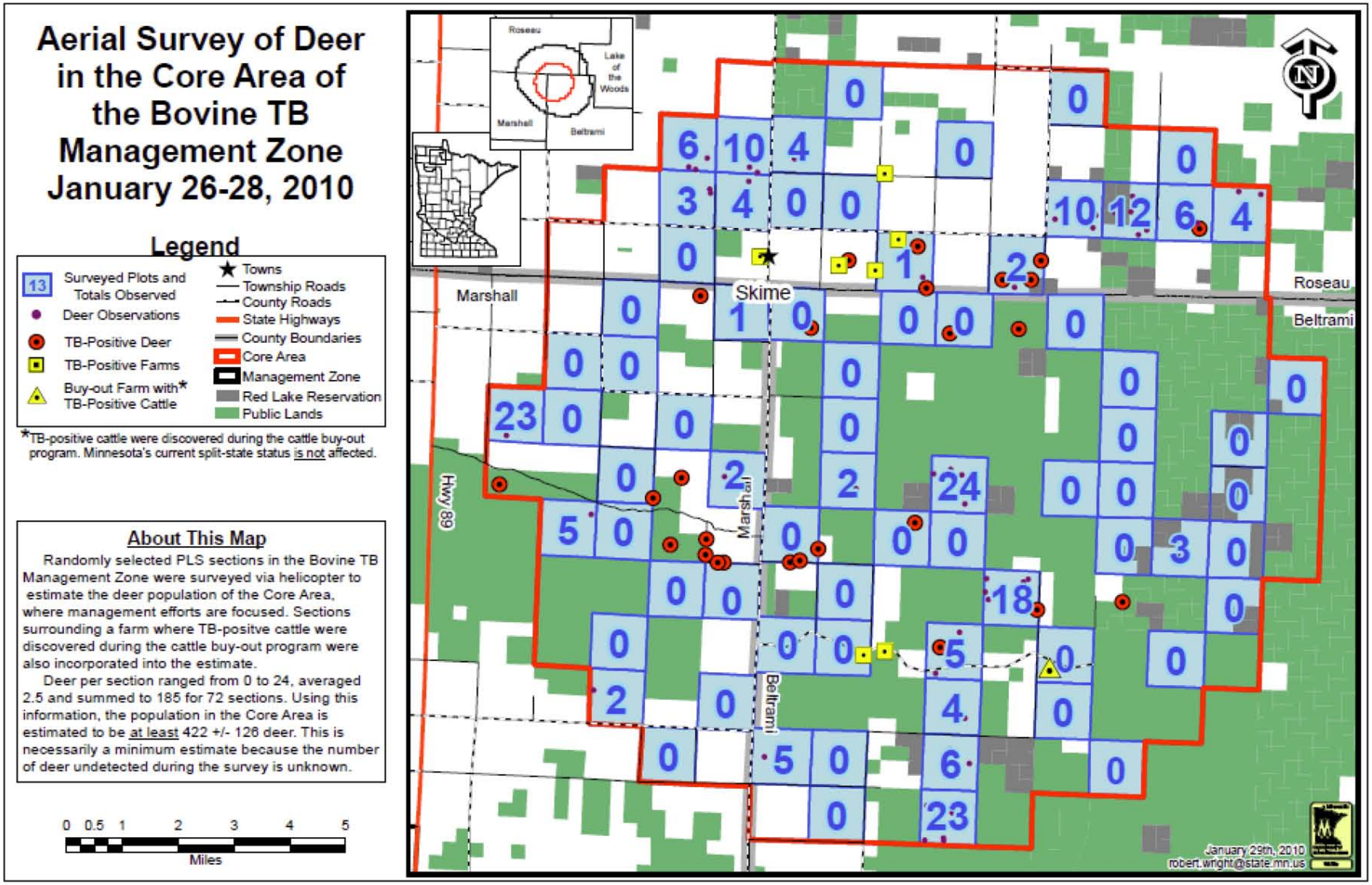


Figure 1. Results of aerial white-tailed deer survey of the Bovine TB Core Area in January 2010, northwestern Minnesota.

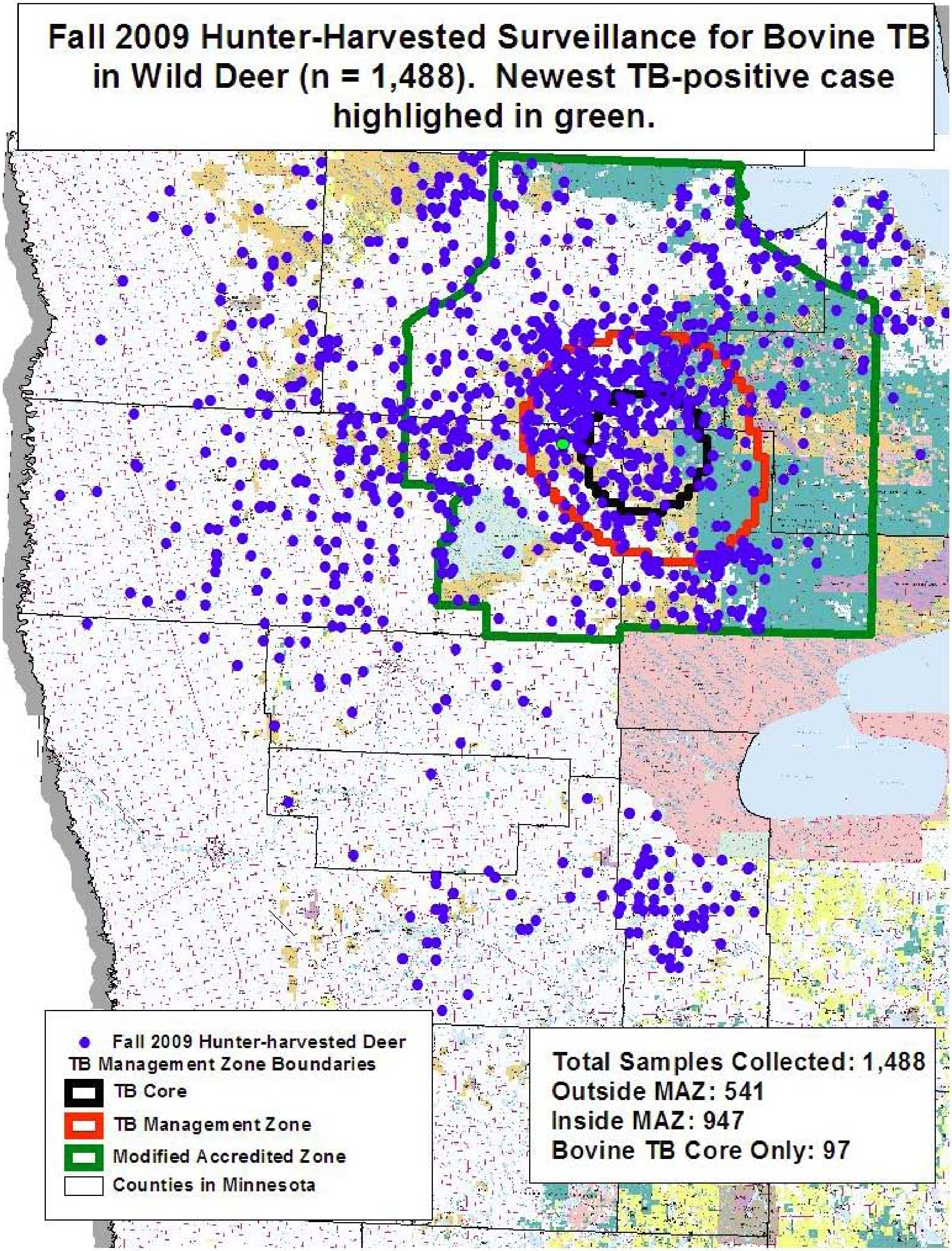
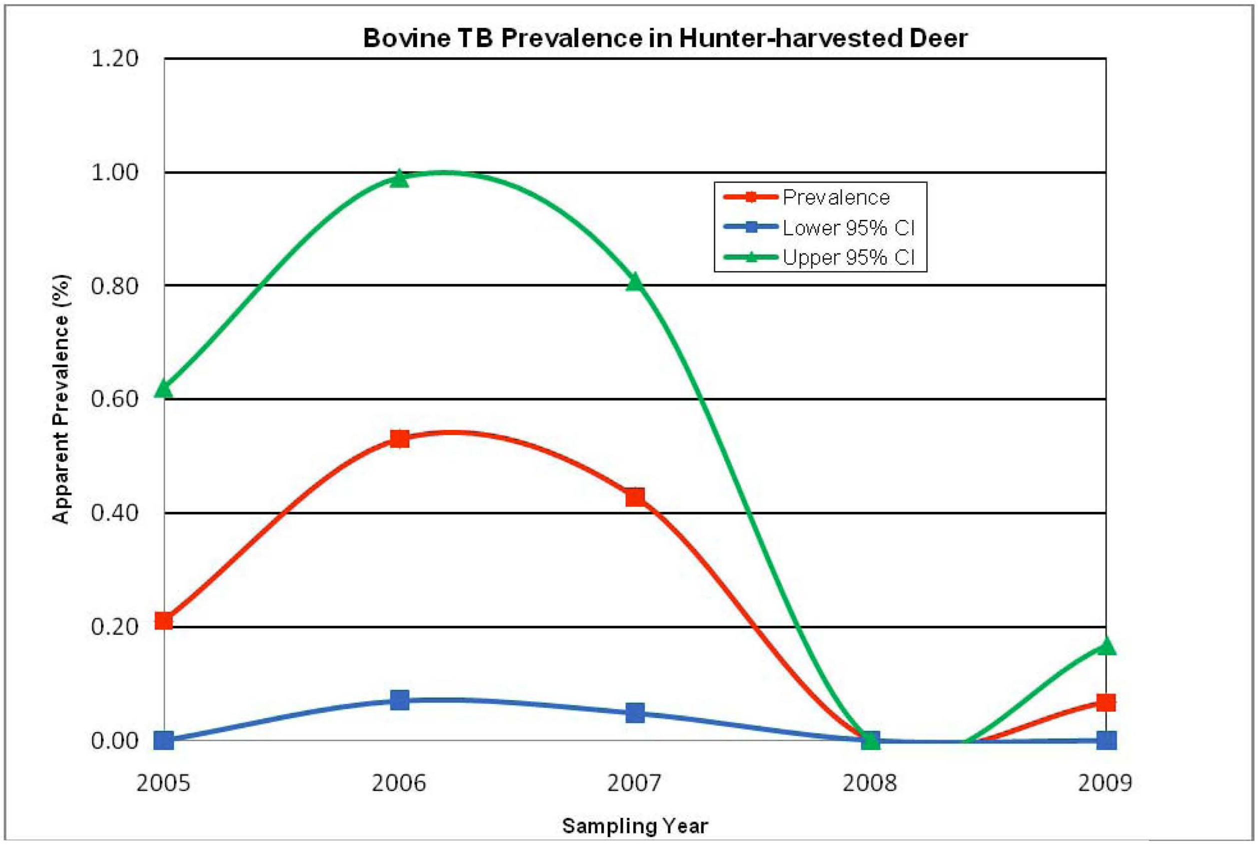


Figure 2. Locations of hunter-harvested deer (*n*=1,246) sampled for Bovine tuberculosis (TB) during fall 2009 in northwestern Minnesota.



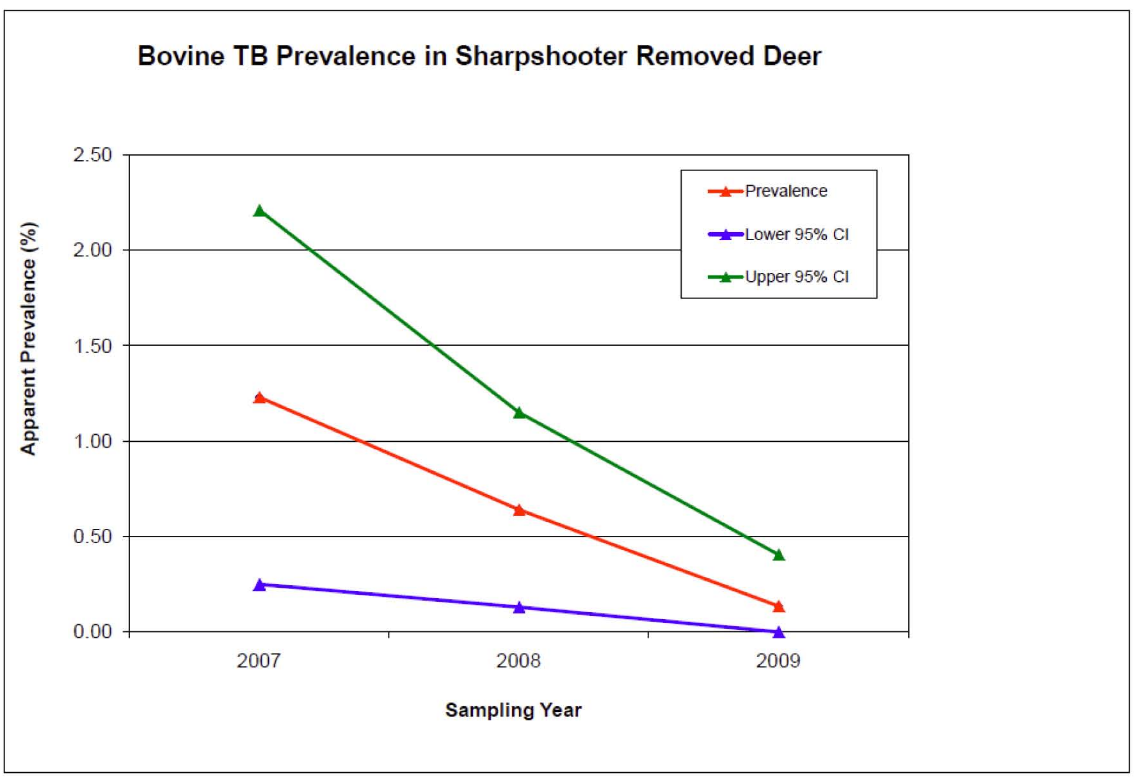


Figure 3. Prevalence of Bovine TB in hunter-harvested deer from 2005–2009 in the Bovine TB Surveillance Zone and disease prevalence from sharpshooter removed deer from 2007–2009 in the Bovine TB Core Area, northwestern Minnesota.

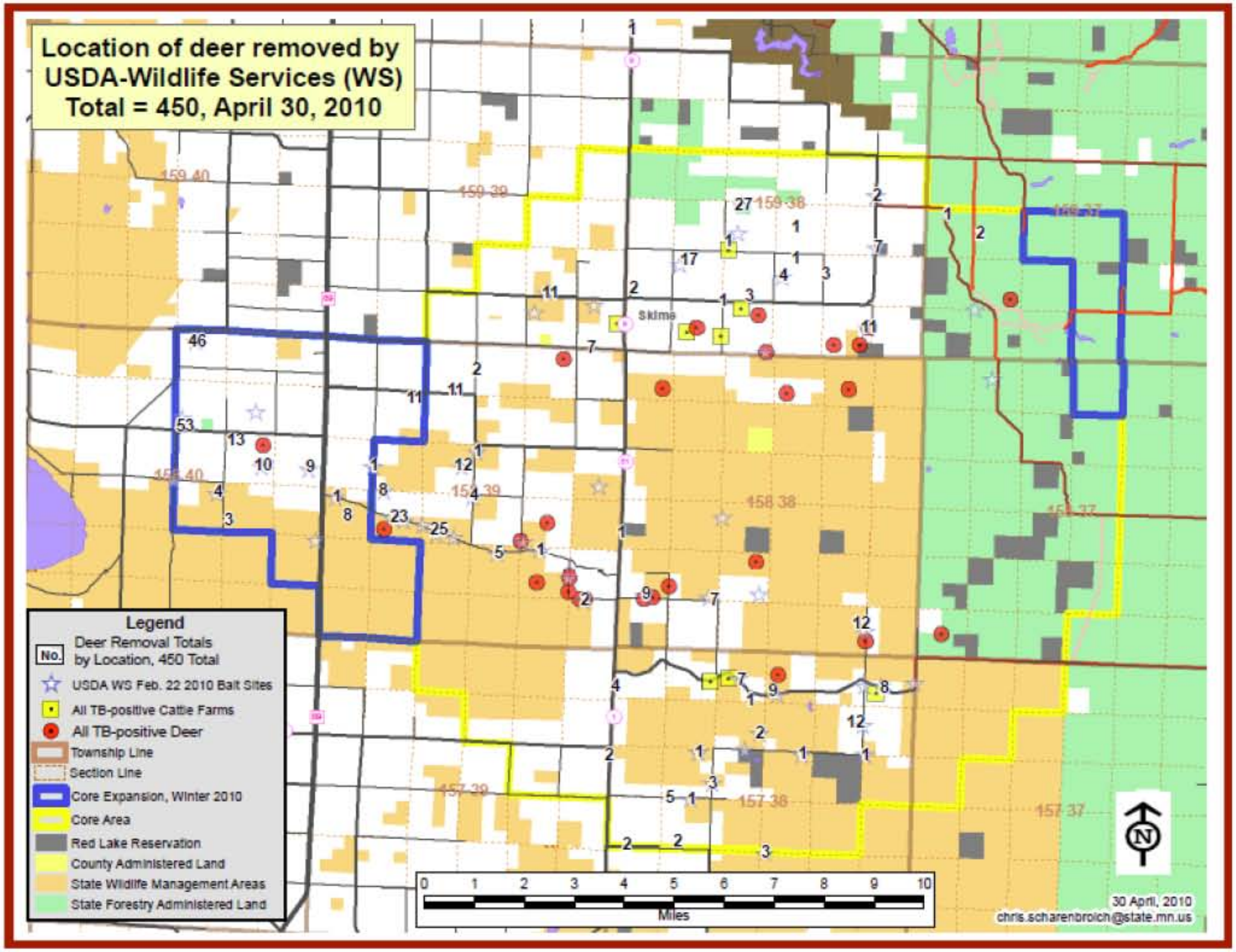


Figure 4. Locations of deer removed (*n*=450) by USDA ground sharpshooters during February-April 2010, in northwestern Minnesota.

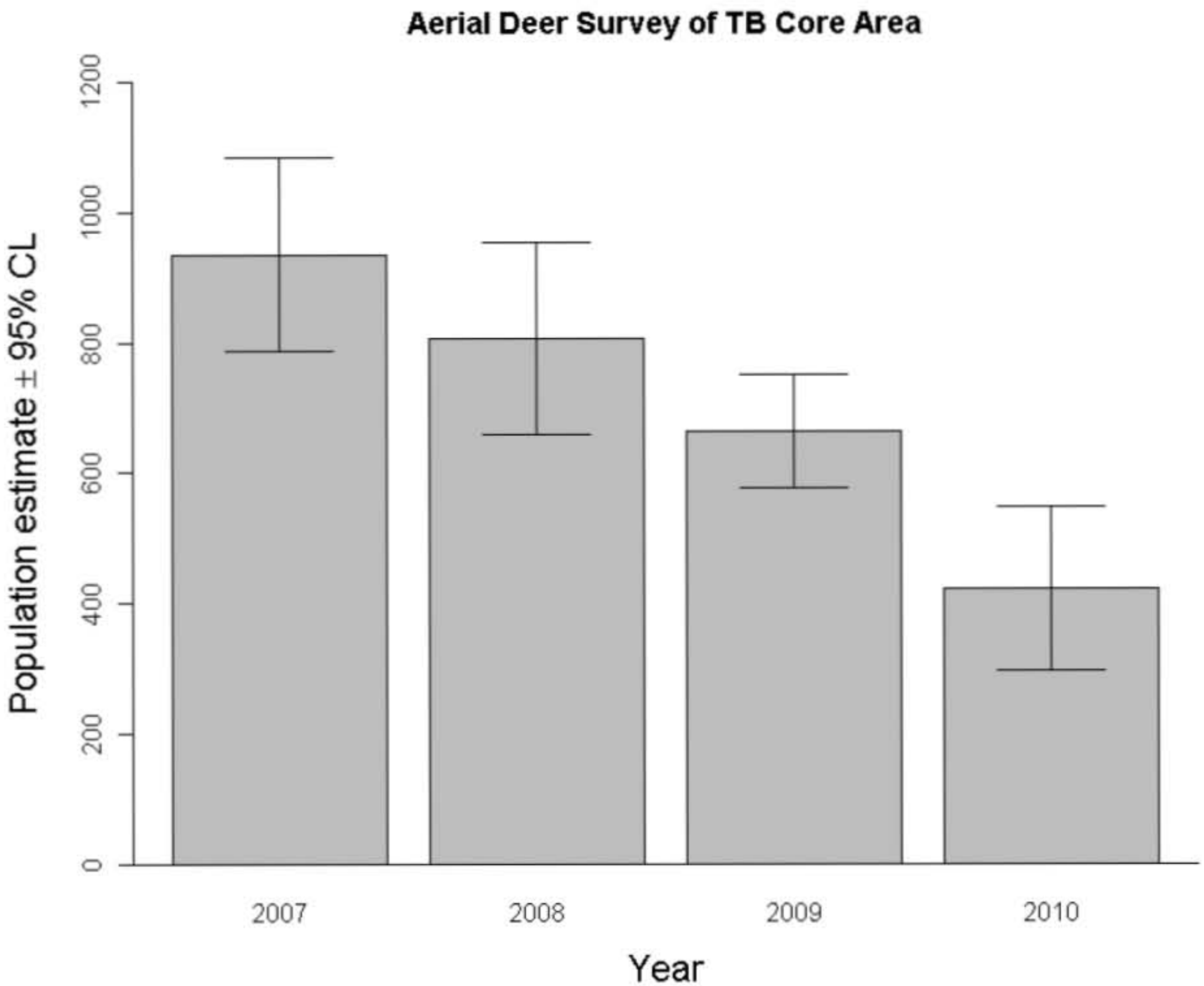


Figure 5. Population estimate of deer within the Bovine TB Core, winters 2007–2010, northwestern Minnesota.

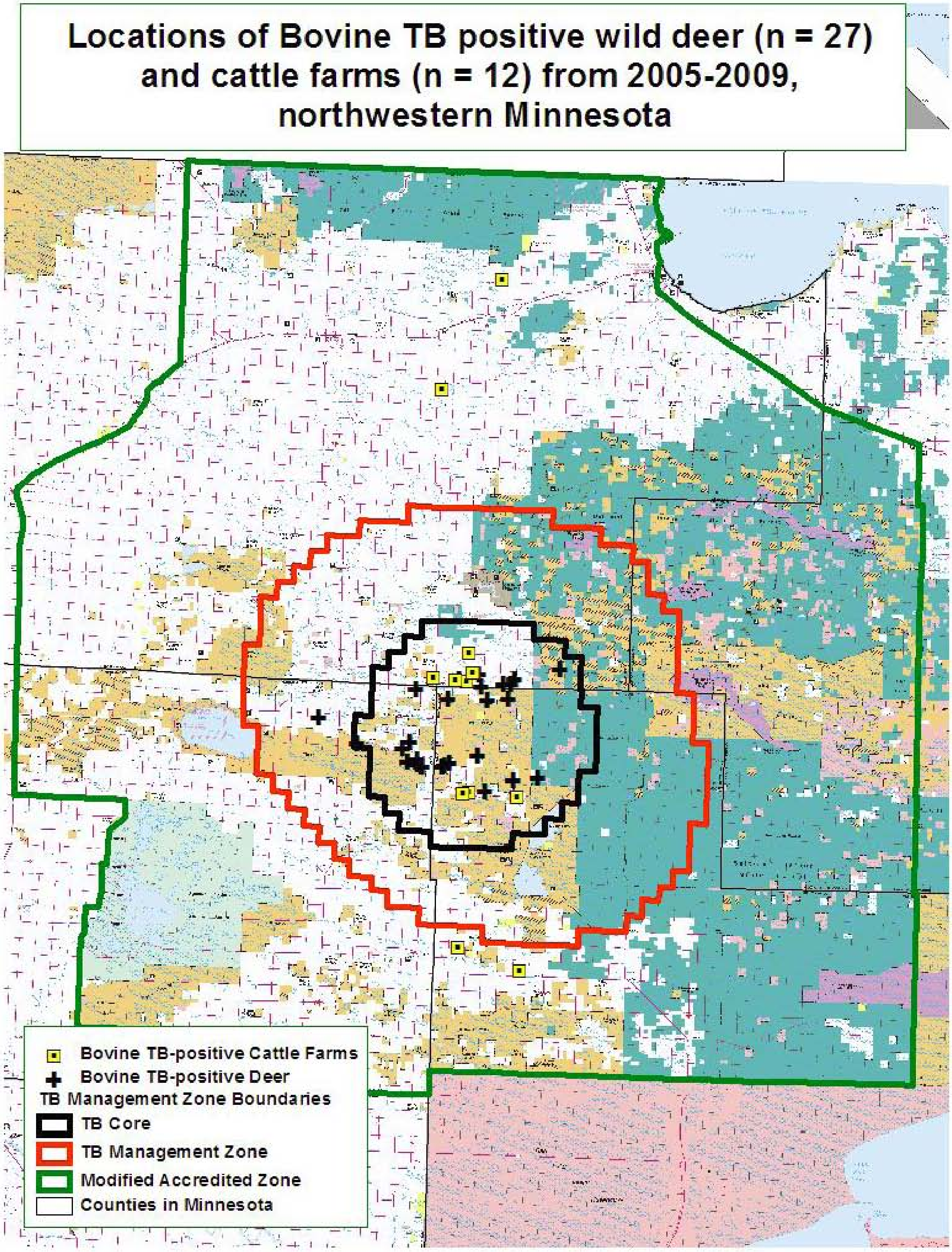
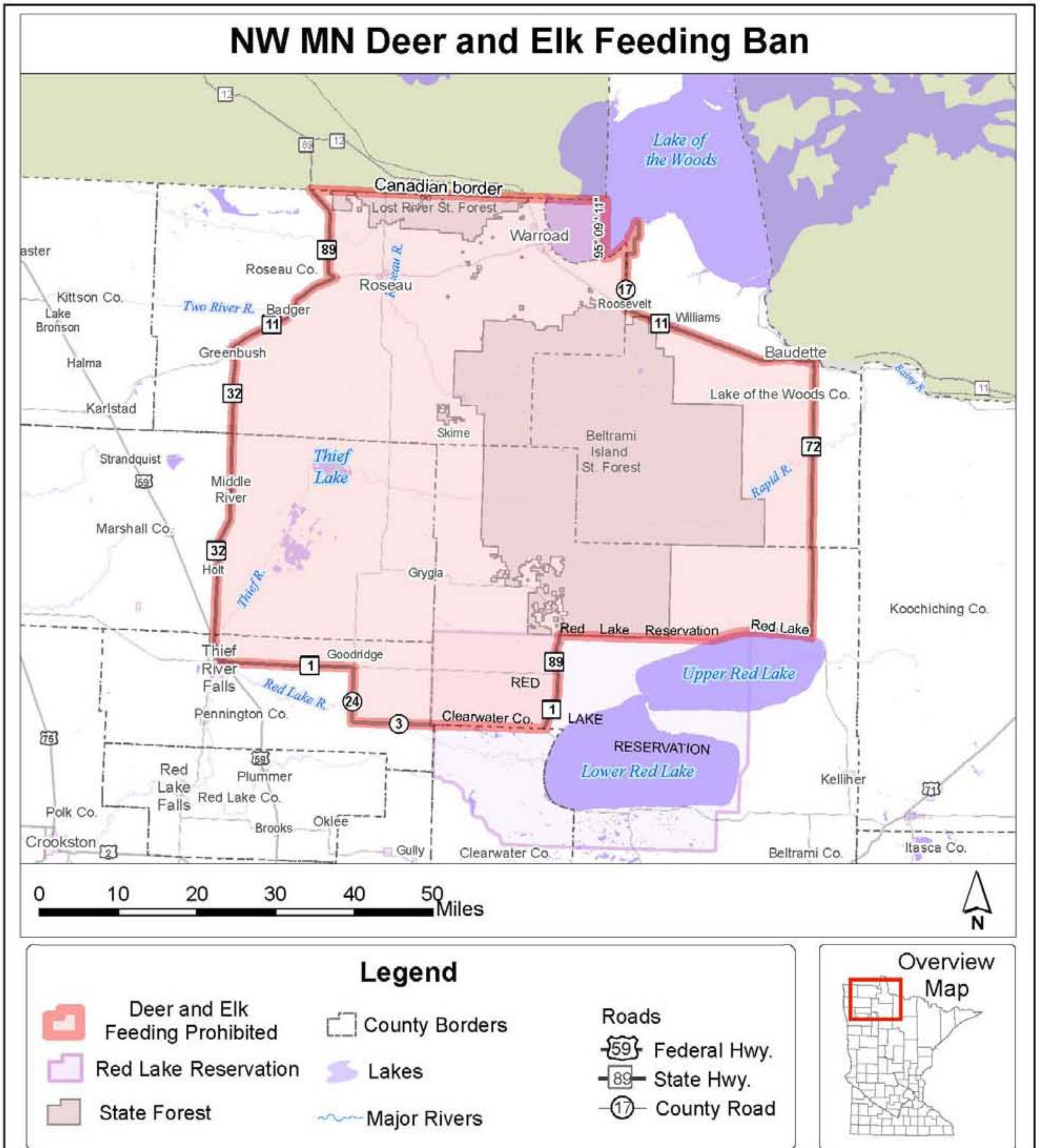


Figure 6. Locations of white-tailed deer found infected (*n*=27) with Bovine TB since fall 2005 in northwestern Minnesota, with the 12 previously-infected cattle operations are also included.



Deer and Elk Feeding Ban Boundary Description

Beginning at State Trunk Highway (STH) 89 and the Canadian border; thence along the Canadian border to longitudinal point 95 degrees, 9 minutes, 11 seconds; thence due south to the shore of Lake of the Woods; thence along the shore of Lake of the Woods to County Road (CR) 17; thence along CR 17 to STH 11; thence along STH 11 to STH 72; thence along STH 72 to the north shore of Upper Red Lake; thence along the north shore of Upper Red Lake to the boundary of the Red Lake Indian Reservation; thence along the boundary of the Red Lake Indian Reservation to STH 89; thence along STH 89 to STH 1; thence south along STH 1 to the Clearwater County line; thence along the Clearwater County line to the Pennington County line; thence due west to the intersection of CR 27 and CR 3; thence along CR 3 to CR 24; thence along CR 24 to STH 1; thence along STH 1 to STH 32; thence along STH 32 to STH 11; thence along STH 11to STH 89; thence along STH 89 to the point of beginning.

Figure 7. Area in northwestern Minnesota where recreational feeding of deer and elk was banned in November 2006, as a preventative measure to reduce risk of disease transmission.

Farmland Wildlife Populations and Research Group
Madelia Research Center
35365 - 800th Avenue
Madelia, Minnesota 56062-9744
(507) 642-8478

ESTIMATING WHITE-TAILED DEER ABUNDANCE USING AERIAL QUADRAT SURVEYS

Brian S. Haroldson

SUMMARY OF FINDINGS

I estimated white-tailed deer (*Odocoileus virginianus)* abundance in select permit areas (PA) using quadrat surveys to recalibrate deer population models and evaluate the impact of deer season regulation changes on population size. With rare exception, precision of population estimates was similar among permit areas. However, because population estimates were not corrected for sightability, estimates represent minimum counts and are biased low. In 2009, I incorporated a sightability estimator to adjust estimates for animals missed during surveys. Sightability estimates were similar during 2009-2010. Additional sightability trials are needed to determine how sightability varies over space and time.

INTRODUCTION

Management goals for animal populations are frequently expressed in terms of population size (Lancia et al. 1994). Accurate estimates of animal abundance allow for documentation of population trends, provide the basis for setting harvest quotas (Miller et al. 1997), and permit assessment of population and habitat management programs (Storm et al. 1992).

The Minnesota Department of Natural Resources (MNDNR) uses simulation modeling to estimate and track changes in deer abundance and, subsequently, to develop harvest recommendations to keep deer populations within goal levels. In general, model inputs include estimates of initial population size and spatial/temporal estimates of survival and reproduction for various age and sex cohorts. Because simulated population estimates are subject to drift as model input errors accumulate over time, it is imperative to periodically recalibrate the starting population within these models with independent deer population estimates (Grund and Woolf 2004).

Minnesota's deer numbers are managed according to numeric population goals within 125 PAs. MNDNR recently revised deer population goals within each PA using a consensus-based, roundtable approach consisting of 15-20 citizens representing varied interest groups (e.g. deer hunters, farmers, foresters, environmental groups, etc.; Stout et al. 1996). Revised goals are used to guide deer-harvest recommendations. Currently, deer populations exceed management goals in many PAs. A conventional approach of increasing the bag limit within the established hunting season framework has failed to reduce deer densities. As a result, MNDNR began testing the effectiveness of 3 non-traditional harvest regulations to increase the harvest of antlerless deer and reduce overall population levels (Grund et al. 2005). Accurate estimates of deer abundance are needed to evaluate these regulations.

My objective in this investigation is to provide independent estimates of deer abundance in select PAs that are within 20% of the true mean with 90% confidence (Lancia et al. 1994). Abundance data will be used to recalibrate population models to improve population management and to evaluate impacts of deer season regulation changes on deer abundance.

METHODS

I estimated deer populations in selected PAs using a quadrat-based, aerial survey design. Quadrat surveys have been used to estimate populations of caribou (*Rangifer tarandus*; Siniff and Skoog 1964), moose (*Alces alces*; Evans et al. 1966), and mule deer (*O. heimonus*; Bartmann et al. 1986) in a variety of habitat types. Quadrats were selected using 1 of 3 sampling designs: (1) stratified random (StRS; Cochran 1977); (2) 2-dimensional (2-D) systematic (Cressie 1993, D'Orazio 2003); or (3) generalized random-tessellation stratified (GRTS; Stevens and Olsen 2004). I used a StRS sampling design in PAs where the local wildlife manager had prior knowledge about deer abundance and distribution. Quadrats were stratified into 2 abundance classes (low, high) based on relative deer densities. Occasionally, additional strata were constructed to encompass management boundaries (e.g., park boundaries). I used a 2-D systematic sampling design in other areas. Systematic designs are typically easier to implement and maximize spatial distribution of the sample. Beginning in 2008, I used the GRTS design to obtain spatially balanced stratified and random samples. This design improves the spatial distribution of StRS and permits replacement of sample quadrats that are lost due to navigation hazard or high human development. Previously, replacement quadrats were unavailable in systematic PAs because of the rigid, 2-D design.

Within each PA, quadrats were delineated by Public Land Survey section boundaries and a 20% sample was selected for surveying. Sample size calculations indicated this sampling rate was needed to meet accuracy and precision objectives. I used OH-58 helicopters during most surveys and attempted to maintain a flight altitude of 60 m above ground level and an airspeed of 64-80 km/hr. A Cessna 182 airplane was used in 3 PAs dominated by intensive row-crop agriculture. To increase visibility, I completed surveys after leaf-drop and when snow cover measured at least 15 cm. A pilot and 2 observers searched for deer along transects spaced at 270-m intervals until they were confident all "available" deer were observed. When animals fled the helicopter, direction of movement was noted to avoid double counting. I used a real-time, moving-map software program (DNR Survey; MNDNR 2005), coupled to a global positioning system receiver and a tablet-style computer, to guide transect navigation and record deer locations, direction of movement, and aircraft flight paths directly to ArcView GIS (Environmental Systems Research Institute 1996) shapefiles. I estimated deer abundance from StRS surveys using PROC SURVEYMEANS (SAS 1999). I used the R programming language (RDCT 2009) and formulas developed by D'Orazio (2003) for 2-D systematic surveys and the R package SPSURVEY (ver. 2.0; RDCT 2009) for GRTS surveys. I evaluated precision using coefficient of variation (CV), defined as standard deviation of the population estimate divided by the population estimate, and relative error (RE), defined as the 90% confidence interval bound divided by the population estimate (Krebs 1999).

I conducted a pilot study in 2 PAs (240, 345) in 2009 to evaluate logistics of using double sampling (Eberhardt and Simmons 1987, Thompson 2002) to estimate sightability (p) of deer from the helicopter. I subjectively selected 10 sightability quadrats (sampling rate = 1.5–3.0%) within each PA where at least 20 deer had been previously observed to help ensure that animals would be available for the evaluation. Immediately after completing the operational survey on each sightability quadrat, a second more intensive survey was flown at reduced speed (48-64 km/hr) to identify animals that were missed (but assumed available) on the first survey (e.g., Gasaway et al. 1986). I used georeferenced deer locations, group size, and movement information from DNR Survey (MNDNR 2005) to "mark" deer (groups) observed in the operational survey and help estimate the number of "new" animals detected in the sightability survey. I defined p as number of "marked" deer / number of "marked" deer + number of "new" deer. I computed $\hat{p}$ for each PA using the arithmetic mean ($\bar{p}$) of quadrat-specific sightability estimates.

During 2010, I implemented double sampling (Eberhardt and Simmons 1987, Thompson 2002) on a subsample of quadrats in 3 permit areas (225, 227, 236) and St Croix State Park (SCSP) to estimate p. For each survey area, I sorted the random sample of survey quadrats (n) by percent woody cover and then selected a random systematic subsample of sightability quadrats (n_s = 7–26 quadrats/survey area; sampling rate = 3.7–11.3%) to help ensure a wide range of covariate values for evaluating the relationship between $\hat{p}$ and percent woody cover (at the quadrat scale). Flight protocol during the operational and sightability surveys was the same as described for 2009 surveys. I computed $\hat{p}$ and $\widehat{var}(\hat{p})$ for each survey area using a generalized linear model (glm function in the R stats package; R Development Core Team 2009) with a logit-link function and an events/trials response for each sightability quadrat where at least 1 deer was observed. Graphical analysis suggested a weak negative relationship between percent woody cover and $\widehat{p}$, but the effect was at least partly confounded with permit area effects. Therefore, I did not include percent woody cover in the logistic model. I used estimates of p from the logistic model to compute population estimates $\hat{\tau}$ adjusted for both sampling and sightability:

$$\hat{\tau} = \frac{N\bar{y}}{\hat{p}},$$

where N = total quadrats in the sampling universe and $\bar{y}$ = average deer count/quadrat in the operational survey (or stratified mean where applicable). With estimated sightability in the denominator, $\hat{\tau}$ is no longer unbiased for τ, although it may be approximately so (Thompson 2002:191). I used Tayor's theorem (Thompson 2002:191, eq. 9) to estimate $var(\hat{\tau})$, assuming p is uncorrelated with $\bar{y}$:

$$\widehat{var}(\hat{\tau}) \approx \frac{N^2}{\hat{p}^2}\left[var(\bar{y}) + \frac{\bar{y}^2}{\hat{p}^2}\widehat{var}(\hat{p})\right]$$

For stratified surveys, I applied the variance estimator to the population total (expanded for sampling; versus summing stratum-specific variance estimates) because when using a single parameter to correct for undetected animals, stratum-specific estimates of abundance will be positively correlated and summing stratum-specific variances will underestimate true uncertainty (Fieberg and Giudice 2008). For 4 permit areas (209, 210, 256, 257) where I did not conduct sightability surveys, I used a simple arithmetic mean ($\bar{p}$) and var($\bar{p}$) to adjust population estimates for estimated sightability.

RESULTS AND DISCUSSION

I completed 4-8 surveys each winter (December-March, 2005-2010; Table 1). Stratified fixed-wing surveys were conducted in PAs 270 and 272. Based on long-term deer harvest metrics, population estimates in these areas were biased low. Several possibilities may explain this result: (1) deer were clustered in unsampled quadrats; (2) deer were wintering outside PA boundaries; (3) sightability was biased using fixed-wing aircraft; and/or (4) kill locations from hunter-killed deer were reported incorrectly. Land cover in these PAs was dominated by intensive row-crop agriculture. After crops were harvested each fall, deer habitat was limited to riparian areas, wetlands, abandoned farm groves, and undisturbed grasslands, including those enrolled in state and federal conservation programs. Although recreational feeding of deer could influence distribution, wildlife managers believed it was not a common practice in these PAs. Thus, I had no evidence to support non-traditional deer distribution in these units. I also had no reason to believe hunter registration errors had greater bias in these units than in other PAs. Although it was possible that deer occupied unsampled quadrats by chance, the use of optimal allocation to increase sampling effort in high strata quadrats because of expected higher deer densities should minimize this possibility. Furthermore, we surveyed 100% of the high-strata quadrats in PA 270, resulting in no unsampled quadrats. Sightability bias, however, is greater in fixed-wing aircraft than helicopters (LeResche and Rausch 1974, Kufeld et al. 1980, Ludwig 1981) and likely explained much of the bias I observed in these PAs. Consequently, beginning in 2007, all surveys have been conducted using a helicopter.

With the exception of PAs 270, 272, and 201, precision (CV, RE) of the population estimates was similar among PAs (Table 1). High precision in PA 270 was, in part, an artifact of sample design. Based on optimal allocation formulas, we selected and surveyed all high strata quadrats. Thus, because no sampling occurred within the high stratum (100% surveyed), sampling variance was calculated only from low strata quadrats. We observed few deer in these low strata quadrats, which resulted in low sampling variance and high precision of the population estimate. It is unlikely that this design (i.e., sampling 100% of high strata quadrats) will be feasible in all areas, especially if deer are more uniformly distributed throughout the landscape.

In contrast, survey precision in PAs 272 and 201 was poor. We observed few deer during either survey (*n*=144 and 56, respectively) and nearly all observations occurred within 1 or 2 quadrats. As a result, associated confidence intervals exceeded 60% of the population estimate (Table 1). Kufeld et al. (1980) described similar challenges with precision due to nonuniformity of mule deer distribution within strata in Colorado.

Prior to 2010, I did not correct population estimates for sightability. Thus, these estimates represent minimum counts and are biased low. Estimates of sightability in 2010 ranged from 0.652 (SE = 0.044) in SCSP to 0.780 (SE = 0.023) in PA 227 and averaged 0.728 (SE = 0.031), which are similar to sightability estimates in 2009 (0.80-0.82). Incorporating uncertainty in the detection process into the population estimates increased relative variance (CV[%]) by 0.4 to 1.6%, which was a reasonable tradeoff between decreased bias and increased variance – although costs associated with the sightability surveys are also important. However, I caution that my estimates of sightability are conditional on animals being available for detection (p_i > 0; *sensu* Johnson 2008). Unfortunately, like many other wildlife surveys, I have no estimates of availability or how it varies over space and time. My approach also assumes that sightability is constant across animals and quadrats. Heterogeneity in detection probabilities can lead to biased estimates of abundance, but the magnitude of the bias will depend on the degree of heterogeneity and the distribution of animal groups (counts) with respect to p_i. Common methods for correcting for heterogeneous detection probabilities include distance sampling, mark–recapture methods (with covariates), and logistic-regression sightability models (based on radio-marked animals). I did not have marked animals in my populations, and relatively high densities of deer in my survey areas would present serious logistical and statistical problems for distance-sampling and double-observer methods. Therefore, my double-sampling approach is a reasonable alternative to using unadjusted counts or applying more complicated methods whose assumptions are tenuous. Nevertheless, my "adjusted" population estimates must still be viewed as approximations to the truth.

Additional sightability trials are needed to determine how $\hat{p}$ varies over space and time. The relationship between $\hat{p}$ and visual obstruction (at the observation scale) will be examined to evaluate heterogeneity in sightability. Future analysis will also include *post-hoc* evaluation of habitat features present in quadrats containing deer. This will provide additional empirical data for use in quadrat stratification. In addition, the impact of winter feeding on deer distribution will be examined to determine if pre-survey stratification flights (Gasaway et al. 1986) are warranted.

ACKNOWLEDGEMENTS

I thank field staff throughout the survey areas for logistical assistance and conducting the surveys. B. Osborn coordinated data collection during 2005. J. Giudice and J. Fieberg provided statistical advice on sample design and analysis. B. Wright and C. Scharenbroich provided GIS technical support and training on DNR Survey. I also thank the enforcement pilots – M. Trenholm, J. Heineman, B. Maas, and T. Buker.

LITERATURE CITED

Bartmann, R. M., L. H. Carpenter, R. A. Garrott, and D. C. Bowden. 1986. Accuracy of helicopter counts of mule deer in pinyon-juniper woodland. Wildlife Society Bulletin 14:356-363.

Cochran, W. G. 1977. Sampling techniques. Second edition. Wiley, New York, New York, USA.

Cressie, N. A. C. 1993. Statistics for spatial data. Second edition. Wiley, New York, New York, USA.

D'Orazio, M. 2003. Estimating the variance of the sample mean in two-dimensional systematic samplings. Journal of Agricultural, Biological, and Environmental Statistics 8:280-295.

Eberhardt, L. L., and M. A. Simmons. 1987. Calibrating population indices by double sampling. Journal of Wildlife Management 51:665–675.

Environmental Systems Research Institute. 1996. ArcView GIS. Version 3.x. Environmental Systems Research Institute, Inc., Redlands, California, USA.

Evans, C. D., W. A. Troyer, and C. J. Lensink. 1966. Aerial census of moose by quadrat sampling units. Journal of Wildlife Management 30:767-776.

Fieberg, J., and J. Giudice. 2008. Variance of stratified survey estimators with probability of detection adjustments. Journal of Wildlife Management 72:837–844.

Gasaway, W. C., S. D. Dubois, D. J. Reed, and S. J. Harbo. 1986. Estimating moose population parameters from aerial surveys. Biological Papers, University of Alaska, Number 22, Fairbanks.

Grund, M., L. Cornicelli, D. Fulton, B. Haroldson, E. Dunbar, S. Christensen, and M. Imes. 2005. Evaluating alternative regulations for managing white-tailed deer in Minnesota – a progress report. Pages 132-137 *in* P. Wingate, R. Kimmel, J. Lawrence, and M. Lenarz, editors. Summaries of Wildlife Research Findings, 2005. Division of Fish and Wildlife, Minnesota Department of Natural Resources, St. Paul.

Grund, M. D., and A. Woolf. 2004. Development and evaluation of an accounting model for estimating deer population sizes. Ecological Modeling 180:345-357.

Johnson, D. H. 2008. In defense of indices: the case study of bird surveys. Journal of Wildlife Management 72:857-868.

Krebs, C. J. 1999. Ecological methodology. Second edition. Benjamin/Cummings, Menlo Park, California, USA.

Kufeld, R. C., J. H. Olterman, and D. C. Bowden. 1980. A helicopter quadrat census for mule deer on Uncompahgre Plateau, Colorado. Journal of Wildlife Management 44:632-639.

Lancia, R. A., J. D. Nichols, and K. H. Pollock. 1994. Estimating the number of animals in wildlife populations. Pages 215-253 *in* T. A. Bookhout, editor. Research and management techniques for wildlife and habitats. Fifth edition. The Wildlife Society, Bethesda, Maryland.

LeResche, R. E., and R. A. Rausch. 1974. Accuracy and precision of aerial moose censusing. Journal of Wildlife Management 38:175-182.

Ludwig, J. 1981. Proportion of deer seen in aerial counts. Minnesota Wildlife Research Quarterly 41:11-19.

Miller, S. D., G. C. White, R. A. Sellers, H. V. Reynolds, J. W. Schoen, K. Titus, V. G. Barnes, Jr., R. B. Smith, R. R. Nelson, W. B. Ballard, and C. C. Schwarz. 1997. Brown and black bear density estimation in Alaska using radiotelemetry and replicated mark-resight techniques. Wildlife Monographs 133.

MNDNR. 2005. DNR Survey: an Arcview GIS 3.x extension for enhancing aerial surveys for wildlife.<http://www.dnr.state.mn.us/mis/gis/tools/arcview/extensions/dnrsurvey/dnrsurvey.html>.

Potvin, F., L. Breton, and L. O. Rivest. 2002. Testing a double-count survey technique for white-tailed deer, *Odocoileus virginianus*, in Quebec. Canadian Field Naturalist 116:488-496.

R Development Core Team [RDCT]. 2009. R: a language and environment for statistical computing. R Foundation for Statistical Computing, Vienna, Austria. <http://www.R-project.org>. April 2010.

SAS Institute. 1999. SAS OnlineDoc, version 8. SAS Institute, Cary, North Carolina, USA. <http://v8doc.sas.com/sashtml/>. April 2006.

Siniff, D. B., and R. O. Skoog. 1964. Aerial censusing of caribou using stratified random sampling. Journal of Wildlife Management 28:397-401.

Stevens, D. L., Jr., and A. R. Olsen. 2004. Spatially balanced sampling of natural resources. Journal of the American Statistical Association 99:262-278.

Storm, G. L., D. F. Cottam, R. H. Yahner, and J. D. Nichols. 1992. A comparison of 2 techniques for estimating deer density. Wildlife Society Bulletin 20:197-203.

Stout, R. J., D. J. Decker, B. A. Knuth, J. C. Proud, and D. H. Nelson. 1996. Comparison of three public-involvement approaches for stakeholder input into deer management decisions: a case study. Wildlife Society Bulletin 24:312-317.

Thompson, S. K. 2002. Sampling. Second edition. John Wiley & Sons, New York, New York, USA.

Table 1. Deer population and density estimates derived from aerial surveys in Minnesota, 2005-2010. Beginning in 2010, estimates were corrected for sightability.

Sampling design	Year	Permit area	Population estimate N	Population estimate 90% CI	CV (%)	Relative error (%)[a]	Density estimate (deer/mi^2) Mean	Density estimate (deer/mi^2) 90% CI	Model estimate (deer/mi^2)
Systematic	2005	252	2,999	2,034 – 3,969	19.5	32.2	2.9	2.0 – 3.8	2
		257	2,575	1,851 – 3,299	16.9	28.1	6.1	4.4 – 7.8	7
	2006	204	3,432	2,464 – 4,401	17.0	28.2	4.5	3.2 – 5.8	5
		209	6,205	5,033 – 7,383	11.4	18.9	9.3	7.6 – 11.1	5
		210	3,976	3,150 – 4,803	12.5	20.8	6.1	4.8 – 7.3	7
		256	4,670	3,441 – 5,899	15.9	26.3	6.8	5.0 – 8.6	5
		236	6,774	5,406 – 8,140	12.1	20.2	15.0	12.0 – 18.0	37
	2007	225	5,341	4,038 – 6,645	14.7	24.4	7.7	5.8 – 9.6	24
		227	5,101	4,245 – 5,960	10.1	16.8	9.7	8.1 – 11.3	13
		346	7,896	5,736 – 10,062	16.4	27.4	21.6	15.7 – 27.6	31
	2008	266	3,853	2,733 – 4,977	17.5	29.1	5.9	4.2 – 7.6	n/a[b]
Stratified	2005	206	2,486	1,921 – 3,051	13.7	22.5	5.3	4.1 – 6.5	5
		270	631	599 – 663	3.0	5.0	0.8	0.8 – 0.9	5
		342	3,322	2,726 – 3,918	10.8	17.7	8.9	7.3 –10.4	10
	2006	201	274	100 – 449	37.6	61.9	1.5	0.6 – 2.5	6
		269	1,740	1,301 – 2,180	15.2	25.1	2.6	1.9 – 3.2	3
		272	472	179 – 764	37.4	61.5	0.9	0.3 – 1.4	5
		SCSP[c]	765	587 – 944	14.2	23.4	12.3	9.5 – 15.2	n/a[c]
	2007	343	6,982	5,957 – 8,006	8.9	14.6	10.0	8.6 – 11.5	29
		344	4,116	3,375 – 4,857	10.7	17.7	19.4	15.9 – 22.9	49
		347	5,482	4,472 – 6,492	11.1	18.2	12.6	10.3 – 14.9	13
		349	10,103	8,573 – 11,633	9.1	15.0	20.2	17.1 – 23.2	35
	2008	262	2,065	1,692 – 2,437	10.9	17.9	2.9	2.4 – 3.4	n/a[b]
		271	1,019	848 – 1,189	10.1	16.6	1.6	1.3 – 1.8	8
		SCSP[c]	1,271	989 – 1,554	13.5	22.2	20.5	16.0 – 25.1	n/a[c]
	2010	SCSP[c]	1,686	1,253 – 2,120	15.6	25.7	27.2	20.2 – 34.2	n/a[c]
GRTS[d]	2008	265	4,575	3,766 – 5,384	10.7	17.7	9.2	7.5 – 10.8	n/a[b]
	2009	240	11,041	9,799 – 13,003	8.5	14.1	16.7	14.4 – 19.1	28
		261	1,721	1,450 – 1,992	9.6	15.7	2.2	1.8 – 2.5	4
		345	4,247	3,678 – 4,806	8.0	13.2	12.8	11.1 – 14.5	21
		348	5,717	4,953 – 6,480	8.1	13.4	17.8	15.4 – 20.1	13
	2010	209	6,180	4,923 – 7,438	12.4	20.4	9.6	7.6 – 11.5	7
		210	4,083	3,106 – 5,061	14.6	24.0	6.4	4.8 – 7.9	10
		225	10,271	8,853 – 11,690	8.4	13.8	15.9	13.7 – 18.1	13
		227	9,318	7,810 – 10,827	9.8	16.2	19.4	16.2 – 22.5	18
		236	7,787	6,487 – 9,088	10.2	16.7	19.3	16.1 – 22.5	19
		256	3,076	2,174 – 3,979	17.8	29.4	4.8	3.4 – 6.2	3
		257	2,810	2,089 – 3,532	15.6	25.7	6.7	4.9 – 8.4	6

[a]Relative precision of population estimate. Calculate as 90% CI bound/*N*.
[b]Permit area boundaries were recently modified. No model estimate is available.
[c]St Croix State Park. No model estimate is available.
[d]Generalized Random-Tessellation Stratified sample design.

NEST SITE SELECTION AND NESTING ECOLOGY OF GIANT CANADA GEESE IN CENTRAL TENNESSEE[1]

Jason S. Carbaugh, Daniel L. Combs, and Eric M. Dunton

ABSTRACT

Little information is available on giant Canada goose (*Branta canadensis maxima*) nest site selection on isolated nesting ponds. We monitored 46 island and 72 shoreline nests in the Upper Cumberland (UC) region of central Tennessee during 2002 and 2003. We measured 6 habitat variables at nesting ponds and randomly selected non-nesting ponds, and we used logistic regression to determine which measured habitat variables were important in nest site selection. Presence of an island was the most important variable but was excluded from the final analysis because of quasi-separation (i.e., geese nested on all known islands in the study area). Geese that nested on shorelines generally selected larger ponds which may have offered a larger foraging base and more escape options from predators. Nest success rates were similar for island and shoreline nests. Management actions in the UC region and similar areas should be concentrated on ponds with islands because of higher goose nesting densities and ease in finding nests.

[1]Human-Wildlife Interactions 4(2): Fall 2010 *in press*

CONSERVATION RESERVE PROGRAM GRASSLANDS AND RING-NECKED PHEASANT ABUNDANCE IN MINNESOTA

James F. Drake, Richard O. Kimmel, J. David Smith, and Gary Oehlert

ABSTRACT

Ring-necked pheasant (Phasianus colchicus) abundance was measured on 15 study areas using roadside counts during the summers of 1990-1994 to examine possible relationships to permanent grasslands and 9 other cover types. The majority of permanent grasslands was enrolled in the Conservation Reserve Program (CRP) and likely would have been actively used for agriculture if not for the CRP. Roads were divided into 300 m segments and the proportion of each cover type was determined within 200 m and 800 m of each segment. A non-parametric procedure was used to determine the most significant predictors of number of pheasants observed on each road segment during roadside surveys. Year, study area, and proportion of cover type were used as predictor variables. Proportion of permanent grassland cover was the most significant predictor in every model examined. Numbers of pheasants, predominantly broods, were approximately 10 times higher in samples that had >30% grassland compared to samples with 10%. There was no statistically significant increase in number of pheasants as grassland increased from 30 to 100%. Year-to-year variation and differences among study areas were the second most significant factors in predicting the number of pheasants observed. Small grains and pasture were also positively correlated to pheasant numbers. If CRP grassland had not been available, pheasant abundance would have been significantly lower in the study areas.

Gamebird 2006: Quail VI and Perdix XII. Published 2009

ECOLOGY, HUNTING SEASONS, AND MANAGEMENT OF GRAY AND FOX SQUIRRELS IN MINNESOTA

Emily J. Dunbar, Richard O. Kimmel, and Eric M. Walberg

SUMMARY OF FINDINGS

We conducted a pilot study to identify potential problems for squirrel hunters and to explore if squirrel-hunting opportunities could be enhanced by harvest regulation changes or management activities. We surveyed Minnesota squirrel hunters to provide an understanding of how hunting opportunities could be improved and to determine if perception of squirrel hunting problems differed among hunter groups. We also surveyed personnel from state and provincial wildlife agencies in the U.S. and Canada to provide information on squirrel season management in other jurisdictions. Finally, a literature review and summary was completed to gain a better understanding of the management and ecology of gray and fox squirrels.

Results of the hunter survey suggest most Minnesota squirrel hunters spent ≤ 7 days hunting and harvested <10 squirrels during the 2008 squirrel-hunting season. In general, hunters have not changed hunting areas in the past 5 years. The most frequently cited obstacle for squirrel hunters was private land access. Most hunters do not believe hunting regulations or squirrel habitat management need to change, but many stated that the Minnesota Department of Natural Resources (MNDNR) could improve their hunting experience by providing additional public hunting land and improving access to public land. There were numerical differences among metro Hmong hunters and non-Hmong hunters regarding number of years hunting squirrels, types of properties being hunted, obstacles faced by hunter groups in gaining access to hunting land, squirrel population trends, and which regulations changes, if any, should take place to change squirrel population trends.

The survey of other wildlife agencies in the U.S. and Canada suggest that, although squirrel hunting season structure varies from state to state, Minnesota's season length and bag/possession limits are similar to most other states/provinces. Most agencies do not have declining squirrel populations, but feel that the resource, due to declining hunter participation, is underutilized.

Indications from this pilot study would suggest that future discussions on squirrels in Minnesota should consider: 1) providing increased hunter access to land for squirrel hunting; 2) managing for higher squirrel populations and hunter harvest through habitat improvement and hunting season management; and/or 3) conducting additional surveys with larger sample sizes to examine the numerical differences observed in this pilot study in more detail and determine if they should be taken into account in future management decisions.

INTRODUCTION

Gray and fox squirrel (*Sciurus carolinensis* and *S. niger*) hunting provides recreational opportunities for an estimated 26,000 hunters annually in Minnesota (Dexter 2008). The reported harvest for gray and fox squirrels has declined by 50% in the past 2 decades. The number of hunters has also declined significantly, from an estimated 39,000 gray squirrel hunters in 1985 to 26,000 hunters in 2008. MNDNR recognizes that hunter participation has declined and wants to encourage greater participation by outreach efforts directed at various groups, including those who “experience language or cultural barriers” (Minnesota Department of Natural Resources 2009). One such cultural community is the Hmong, an ethnic group from Southeast Asia. Hmong hunters have expressed concern about perceived low populations of squirrels on public hunting land near population centers (Tim Bremicker, personal communication).

MNDNR initiated this pilot study to determine if squirrel-hunting opportunities could potentially benefit from harvest regulation changes and/or management activities and to determine which changes could increase the huntable squirrel population on public land.

OBJECTIVES

1. Survey a sample of non-Hmong and Hmong Minnesota squirrel hunters to determine if perception of squirrel hunting problems differed between the hunter groups and to provide an indication of how hunting opportunities could be improved;
2. Survey other state/provincial wildlife agencies to gain knowledge about their squirrel hunting seasons and management programs; and
3. Conduct a literature review and summary to gain a better understanding of the management and ecology of gray and fox squirrels.

METHODS

A survey of Minnesota squirrel hunters was conducted April-May 2009. We collected names and addresses of hunters who indicated they had harvested squirrels on small game hunter surveys from 2005-2008. We sampled 400 hunters; 200 hunters were selected from the metro region and 200 hunters were selected from greater Minnesota. Hunters from the 'metro' area were those hunters with mailing addresses from the 7 county area (Anoka, Carver, Dakota, Hennepin, Ramsey, Scott, and Washington) surrounding Minneapolis and St. Paul, MN. Statewide non-metro hunters had addresses outside of the metro counties previously indicated. Note that these designations do not necessarily indicate where those surveyed hunted, but where they received mail. One hundred metro hunters were assumed Hmong hunters and 100 hunters were assumed non-Hmong hunters based on surname. Hmong names were selected by choosing hunters with traditional Hmong clan surnames provided by the Southeast Asian Community Liaison. Non-Hmong hunters were those hunters whose last name did not match the list of Hmong traditional clan surnames. The sample size gathered from the 2005-2008 small game hunter survey did not meet the desired sample size for the metro Hmong hunter group, so additional names and addresses of Hmong hunters were included from the 2008-2009 small game hunter survey. The survey instrument (Appendix 1) consisted of questions relating to harvest, counties and land ownership of properties hunted, hunter experiences, hunter access, hunter perception of squirrel populations, and suggestions for improving squirrel hunting experiences. Responses were entered into Microsoft Access and converted to percentages.

A survey of state/provincial wildlife agencies was conducted during April-May 2009. An email survey was sent to 47 state wildlife agencies (Alaska, Hawaii, and Minnesota were excluded) and 4 Canadian provincial wildlife agencies (Quebec, Ontario, Manitoba, and Saskatchewan). The survey instrument (Appendix 2) consisted of questions relating to season opening/closing dates, bag and possession limits, management, research, population estimation, and issues concerning squirrel hunting.

A list of manuscripts relating to the ecology, management, and hunting mortality of gray and fox squirrels was compiled and selected manuscripts summarized.

RESULTS

Hunter Survey

Surveys were mailed on 10 April 2009 with a second mailing to non-respondents on 1 May 2009. The overall response rate was 80% for 2 mailings. Non-response bias was not evaluated due to the high response rate. Overall, a majority (58%) of the respondents indicated that they had hunted squirrels during the last hunting season (2008). A higher percentage of metro Hmong respondents (73%) hunted squirrels in 2008 as compared to metro non-Hmong and statewide non-Hmong hunters (55% and 52%). Metro non-Hmong hunters (hereafter referred to as MNH hunters) hunted squirrels in 33 of the state's 87 counties (Table 1). The 3 most often hunted areas were Washington (19%), Anoka (15%), and Pine (13%) counties (Table 1). Metro Hmong hunters (hereafter referred to as MH hunters) hunted squirrels in 26 of the state's 87 counties (Table 1). The 3 most often hunted areas were Winona (37%), Anoka (13%), and Houston (13%) counties (Table 1). Statewide non-Hmong hunters (hereafter referred to as SNH hunters) hunted squirrels in 49 of the state's 87 counties (Table 1). The 3 most often hunted areas were Morrison (7%), Stearns (7%), and Wright (6%) counties (Table 1).

Roughly half of squirrel hunters harvested between 1 - 5 squirrels during the last hunting season (46%; Table 2). Forty-eight percent of MNH and MH hunters harvested between 1 - 5 squirrels, while 43% of SNH hunters harvested the same amount (Table 2). Overall, about 1/3 of the hunters harvested between 6 and 10 squirrels (27%; Table 2). Thirty-three percent of SNH hunters harvested between 6 and 10 squirrels, while 17% and 28% of MNH and MH hunters harvested squirrels at this same level (Table 2). The 3 hunter groups spent a similar amount of time hunting squirrels, with most squirrel hunters (70%) reporting spending 7 days or less hunting squirrels (Table 3). Over half (53% and 63%) of MNH and SNH hunters have hunted squirrels for at least 21 years, while over half (58%) of the MH hunters have hunted for 10 years or less (Table 4). Most MH hunters hunted exclusively on public land (98%), while only 17% and 5% of MNH and SNH hunted exclusively on public land (Table 5).

Hunters were also asked if they had hunted new properties in the past 5 years. Approximately half (49%) of hunters indicated hunting the same properties, while other half hunted new properties in addition to traditional properties (Table 6). Few hunters have switched to different properties in the past 5 years (Table 6). MH hunters were the only group where some hunters had completely switched to new properties, with 9% of MH hunters indicating that they were hunting on different properties in the past 5 years (Table 6).

Hunters were asked whether they encountered obstacles to gain access to hunting land. Sixty-two percent of MNH hunters reported obstacles, 78% of MH hunters reported obstacles, and 46% of SNH hunters reported obstacles (Table 7). Private land access was the most frequently cited obstacle for all groups (62%). While private land access was the most frequently cited obstacle for both MNH (64%) and SNH (71%) hunters, it was not the most frequently cited obstacle for MH hunters (50%; Table 7). The most frequently cited obstacles for MH hunters (64%) were how to find additional public hunting land and not comfortable asking for permission to access private land (Table 7). Many hunters (40%) reported 2 different types of obstacles, while 3% reported 5 or more different types of obstacles (Table 8). The highest percentage of MNH (50%) and SNH (43%) reported 2 different types of obstacles, but the highest percentage of MH hunters (31%) reported 3 different types of obstacles (Table 8).

Regarding squirrel population trends in areas used by hunters, 51% of the hunters surveyed reported that squirrel populations have remained stable over the past 5 years (Table 9). Few hunters (13%) felt that squirrel populations were increasing (Table 9). More MH hunters (65%) felt the populations had declined than MNH and SNH hunters (37% and 19%; Table 9).

Hunters were also asked about changes to hunting regulations and habitat management based on their perception of squirrel populations in the areas where they hunt. Most hunters (62%) recommended no changes (n=103; Table 10). MH hunters recommended no change to regulations less often (35%) than MNH (68%) and SNH (74%) hunters (Table 10). Hunters that indicated that changes were needed cited habitat management more frequently (n=56) than hunting regulation changes (n=34; Table 10).

The final question asked hunters what the DNR could implement to improve their squirrel hunting experience. Most respondents reported additional public hunting land and improved access to hunting land would improve their hunting experience (Table 11).

Agency Survey

Surveys were emailed on 27 April 2009 with a second emailing to non-respondents on 15 May 2009. The overall response rate was 86% for 2 emailings. Eighty-two percent of wildlife officials that responded to the survey indicated that their state/province does have a fox and/or gray squirrel-hunting season (n = 38). Squirrel seasons varied widely among the different agencies. Two agencies reported year-round squirrel-hunting seasons (Manitoba and

Washington); while 7 other agencies (Arkansas, Connecticut, Kentucky, Louisiana, Pennsylvania, Tennessee, Wyoming) reported split seasons. Split seasons refer to the states/provinces that provide multiple hunting seasons throughout the year (for example, having a spring and fall squirrel hunting season). Agencies that reported using different hunting zones include Massachusetts, Mississippi, New York, Oregon, and Texas. Pennsylvania and West Virginia have youth hunts before the regular season.

Opening and closing dates varied widely. Agencies that did not have a split season or different hunting zones responded that opening dates occurred during the months of May (n = 1), June (n = 1), August (n = 4), September (n = 9), October (n = 2), and November (n = 1) and closing dates occurred during December (n = 3), January (n = 6), and February (n = 8), and March (n = 1). States that had a fall and spring season had opening dates in August (n = 1), September (n = 2), October (n = 1) for the fall season and opening dates in January (n = 1) and May (n = 3) for the spring season. Closing dates for states that have a fall and spring season are December (n = 1), February (n = 3), March (n = 1), May (n = 1), and June (n = 2), respectively. States that have hunting zones generally have different start dates, but have the same end date (Massachusetts, Mississippi, and New York). Connecticut and Pennsylvania each have 3 regular seasons that vary in length and occur in the fall and winter months (September to February). Length of squirrel-hunting seasons differs from 75 days to 365 days (average = 172).

Bag and possession limits for squirrel harvest varies. Most states (n = 21) reported a single bag limit (range = 4-12, average = 7) and possession limit (range = 4-40, average = 15). Some states (n = 9) reported a single bag (range = 5-10, average = 6) with no possession limit. State-specific bag and possession limits include:

- having different limits for gray and fox squirrels;
- having different limits for different parts of the state;
- having different limits for different seasons;
- having no limits; and
- intending to raise limits.

Most shooting hours are 1/2 hr before sunrise to 1/2 hr after sunset (n = 21), or 1/2 hr before sunrise until sunset (n = 7), although 4 states/provinces have no restrictions.

Wildlife officials were asked if they specifically manage for fox and/or gray squirrels. Maryland was the only state that monitored population trends and harvest to establish seasons and bag limits. Other states/provinces manage for squirrels indirectly by managing forest habitat (n = 8). None of the states responded that they estimate squirrel populations, although 3 states (Illinois, Maryland, and Michigan) monitor population trends based on hunter surveys, and 2 states (South Carolina and North Dakota) monitor population trends based on spotlight and rural postal carrier sighting surveys, respectively. Four states are currently conducting squirrel research (South Carolina, Tennessee, and Washington) or are proposing a study (Mississippi).

Wildlife officials were asked about issues concerning squirrel hunting. Forty-seven percent of states/provinces indicated concerns. The most common concerns were declining hunter participation, habitat loss/declining populations, and hunters requesting season changes.

Literature Review

The list of manuscripts and summaries follows the tables and appendices.

DISCUSSION

The results of the surveys imply that squirrels in most states (including Minnesota), are for the most part, an under-utilized resource. The majority of Minnesota squirrel hunters we surveyed spent no more than 7 days/year hunting squirrels. Minnesota’s current squirrel season is 163 days long, indicating that few squirrel hunters take advantage of the lengthy season. Minnesota's squirrel season is slightly shorter than the average season length (172 days) reported by the other states/provinces. Minnesota's daily bag and possession limits (7/14) are similar to the average limits of other states/provinces. Most hunters in Minnesota are not considered avid squirrel hunters, with a majority harvesting only 10 squirrels or less per year. Squirrel hunters in Minnesota tend to hunt the same areas from year to year with nearly half hunting on the same properties. Finally, squirrel hunters seem to be content with the current season, but would prefer easier access to hunting areas or new public areas to hunt.

Survey responses of concern are related to the types of properties (public vs. private) that hunter groups are using, the obstacles faced by hunter groups in gaining access to hunting land, squirrel population trends, and which regulation changes, if any, could improve squirrel populations and harvest. For each of these questions, responses from the MH hunter group differed from the MNH and SNH hunter groups. MH hunters use public hunting land almost exclusively, while MNH and SNH hunters use either private or a mix of public and private hunting land. Obstacles that impact MH hunters are not the same as those that impact MNH and SMH hunters. MH hunters are unsure how to find additional public land for hunting and are uncomfortable asking permission to hunt on private land, while MNH and SNH hunters face difficulties with land that is posted as no hunting or trespassing. Most MH hunters surveyed feel that squirrel populations are decreasing on lands they hunt, but many MNH and SNH hunters believe they are stable. More MH hunters believe that changes are needed based on squirrel population trends than MNH or SNH hunters. A higher percentage of MH hunters favor hunting regulation changes than MNH and SNH hunters. MH hunters favor habitat management changes at a rate similar to MNH hunters. This question in the survey was vague in regards to the type of changes needed. If future surveys are conducted, they should separate questions concerning habitat management changes and hunting regulation changes (bag limit, season timing and season length, etc.) to better determine support among squirrel hunters. Future discussions for land access issues should focus on providing MH hunters with maps or lists of different types of public hunting land and developing a contact list of landowners willing to allow squirrel hunters on their property. A landowner list would also benefit MNH and SMH hunters who want to find additional hunting land.

It appears that low squirrel populations are a concern to some hunter groups in other parts of the U.S. A few states mentioned that populations are declining due to habitat loss (Indiana and New Jersey). Mississippi noted complaints from hunters about low populations in some regions of the state. However, many states replied that squirrels are an under-utilized resource (Illinois, Maryland, Missouri, Pennsylvania, and Vermont). It is possible that squirrel-hunting is more popular in some states than others, and hunting pressure (in addition to habitat loss) may be responsible for low squirrel numbers. A more in depth look at hunter pressure (number of hunters, number of days spent hunting, types of property hunted, etc.) for each state/province may help to explain the variable responses from this survey. Information is also needed for the demographics of squirrel hunters in other states/provinces. If hunters are restricted to an area or a type of property that is scarce, there are likely to be differing effects on squirrel populations.

If further squirrel hunter surveys are desired, we suggest surveying a larger number of hunters and expanding upon questions relating to types of property hunted, access obstacles, and acceptable regulation changes. Determining when squirrels are being harvested in the season would also provide information on potential seasonal framework changes. A survey with a larger sample size may find further differences among the hunter groups, or strengthen the existing numerical differences with statistical significance. Power analysis should be used to determine sample size. Additional factors to consider in determining an adequate sample size include an estimated response rate and percentage of small game license holders that hunt squirrels. A Hmong hunter sample could be pulled by selecting the last names of hunters that correspond to the clan surnames. MNH and SNH hunters could be selected based upon zip codes. If statistical difference could be found between the 3 groups, then metro Hmong squirrel hunters' perceptions should be recognized as being different than non-Hmong squirrel hunter' perceptions. This difference in perception between metro Hmong and non-Hmong could be taken in account in future season management decisions.

ACKNOWLEDGMENTS

We would like to thank T. Bremicker, J. Johnson, and T. Vang for bringing attention to the issue and supplying background information.

LITERATURE CITED

Dexter, M. H., editor. 2008. Status of wildlife populations, fall 2008. Unpublished report, Division of Fish and Wildlife, Minnesota Department of Natural Resources, St. Paul, Minnesota. 300 pp.

Minnesota Department of Natural Resources. 2009. A Strategic Conservation Agenda 2009-2013. Minnesota Department of Natural Resources, St. Paul, MN. Accessed August 7, 2009. Available online: http://www.dnr.state.mn.us/conservationagenda.index.html

Table 1. Response for question 2: Which county/counties did you hunt squirrels? Squirrel hunter survey, 2009, Minnesota.

	Response (%)		
County	Metro non-Hmong (n=47)	Metro Hmong (n=38)	Statewide non-Hmong (n=83)
Aitkin	9	3	2
Anoka	15	13	
Becker			1
Beltrami			1
Benton			4
Blue Earth	2		1
Brown		3	2
Carlton	2	3	2
Carver	4	3	
Cass	4		5
Chisago	4	5	1
Cottonwood			1
Crow Wing	4		2
Dakota	9	3	
Dodge			2
Douglas	4		5
Fillmore	2	5	2
Goodhue	4	8	1
Grant	2		1
Hennepin	4	5	
Houston	4	13	5
Hubbard			1

Table 1. continued.

Isanti			1
Itasca			1
Jackson	2		
Kanabec			2
Kandiyohi			2
Kittson	2		
Lake			1
Le Sueur	4	3	1
Lincoln			1
Mcleod	1		1
Mille Lacs	4	3	1
Morrison	2		7
Mower			4
Murray			1
Nicollet		4	1
Olmstead	2	4	5
Ottertail			2
Pine	13	3	
Ramsey	4	3	
Redwood			1
Renville			1
Rice		3	4
Roseau			1
Scott	11	5	2
Sherburne	4	5	4
Stearns	4	3	7
Steele			4
St. Louis	2		5
Todd	6	3	2
Wabasha		11	2
Wadena			1
Waseca			2
Washington	19	3	
Winona	2	37	4
Wright		3	6

Table 2. Response for question 3: Approximately, how many squirrels did you harvest last season? Squirrel hunter survey, 2009, Minnesota.

	Response (%)			
Number of squirrels	Overall response	Metro non-Hmong (n=46)	Metro Hmong (n=46)	Statewide non-Hmong (n=83)
1-5	46	48	48	43
6-10	27	17	28	33
11-15	15	17	15	14
16-20	7	11	4	6
21-25	3	2	2	4
26-30	1	4	0	0
31+	1	0	2	0

Table 3. Response for question 4: Approximately, how many days did you spend hunting squirrels last season? Squirrel hunter survey, 2009, Minnesota.

	Response (%)			
Number of days	Overall response	Metro non-Hmong (n=47)	Metro Hmong (n=47)	Statewide non-Hmong (n=83)
1-7	70	68	70	71
8-14	19	21	17	19
15-21	7	6	11	5
21-28	3	4	0	4
29+	1	0	2	1

Table 4. Response for question 5: How long have you hunted squirrels in Minnesota? Squirrel hunter survey, 2009, Minnesota.

	Response (%)			
Number of years	Overall response	Metro non-Hmong (n=47)	Metro Hmong (n=46)	Statewide non-Hmong (n=83)
1-5	15	11	39	10
6-10	16	15	28	10
11-15	12	11	17	10
16-20	10	11	13	8
21+	47	53	11	63

Table 5. Response for question 6: Do you hunt squirrels on public land, private land, or both? Squirrel hunter survey, 2009, Minnesota.

	Response (%)			
Type of land	Overall response	Metro non-Hmong (n=47)	Metro Hmong (n=47)	Statewide non-Hmong (n=83)
Public land	33	17	98	5
Private land	28	28	0	43
Both public and private land	40	55	2	52

Table 6. Response for question 7: In the past 5 years, have you hunted squirrels on the same properties, new properties, or both? Squirrel hunter survey, 2009, Minnesota.

	Response (%)			
Types of properties	Overall response	Metro non-Hmong (n=47)	Metro Hmong (n=46)	Statewide non-Hmong (n=83)
Same	49	38	48	55
New	2	0	9	0
Both (same and new)	49	62	43	49

Table 7. Response for question 8: What are the main obstacles for gaining access to property for squirrel hunting? Squirrel hunter survey, 2009, Minnesota.[1]

	Response (%)			
Types of obstacles	Overall response	Metro non-Hmong (n=45)	Metro Hmong (n=46)	Statewide non-Hmong (n=81)
No obstacles[2]	42	38	22	54
Land is posted [3]	62	64	50	71
Not sure how to find additional public land	36	29	64	14
Not comfortable asking for permission	42	36	64	24
Denied access by landowner	36	46	19	43
Difficulty finding owners of private land	54	57	58	46
Other	14	14	21	11

[1] Respondents could report > 1 type of obstacle
[2] Percent response was calculated using number of respondents indicating no obstacles divided by the total number of respondents for that hunter group
[3] Obstacles' response was calculated using number of respondents indicating a certain obstacle divided by the number of respondents reporting obstacles for that hunter group

Table 8. Number of access obstacles reported by hunters for question 8. Squirrel hunter survey, 2009, Minnesota.[1]

	Response (%)			
Number of obstacles	Overall response	Metro non-Hmong (n=28)	Metro Hmong (n=36)	Statewide non-Hmong (n=37)
1	23	4	7	12
2	40	14	10	16
3	20	6	11	3
4	15	4	5	6
5	2	0	2	0
6	1	0	1	0

[1] Respondents could report > 1 type of obstacle

Table 9. Response for question 9: Over the past 5 years, do you think squirrel populations in areas where you hunt are decreasing, about the same, or increasing? Squirrel hunter survey, 2009, Minnesota.

	Response (%)			
Population trend	Overall response	Metro non-Hmong (n=47)	Metro Hmong (n=46)	Statewide non-Hmong (n=83)
Decreasing	36	37	65	19
Same	51	48	35	62
Increasing	13	15	0	20

Table 10. Response for question 10: Based on your perception of squirrel population trends, which changes would you recommend? Squirrel hunter survey, 2009, Minnesota.[1]

	Response (%)			
Changes	Overall response	Metro non-Hmong (n=44)	Metro Hmong (n=46)	Statewide non-Hmong (n=77)
No Changes[2]	62	68	35	74
Hunting Regulations[3]	34	29	50	15
Habitat Management	56	64	60	45
Enforcement of Regulations	20	36	20	10
Other	27	14	27	35

[1] Respondents could report > 1 type of change
[2] Percent response was calculated using number of respondents indicating no changes divided by the total number of respondents for that hunter group
[3] Changes' response was calculated using number of respondents indicating a certain change divided by the number of respondents reporting changes for that hunter group

Table 11. Response for question 11: How could the DNR improve your hunting experience? Squirrel hunter survey, 2009, Minnesota.

Types of improvements	Number of responses
Additional public land/improved access to hunting land	20
Better information about public hunting lands	7
Habitat management	9
Change season management	14
Create sanctuaries	4
Increase number of squirrels	10
Better enforcement of regulations	3
No blaze orange requirements	2
Predator control	2
Increase safety	3
Hmong concerns	3
Other	6

Appendix 1.

Minnesota Fox and Gray Squirrel Hunter Survey

You have been selected from a group of hunters that harvested squirrels as indicated on past Small Game Hunter surveys. Because this survey is only being sent to a small number of hunters, your input is extremely valuable, please complete and return the following survey as soon as possible. Your identity will be kept confidential.

1. Did you hunt fox and/or gray squirrels last season (September 2008-February 2009)?

   Yes ___ No* ___

   *If No, then you do not need to continue the survey; please answer question 1 and return the survey.

2. In which county/counties did you hunt squirrels? ______________________________

3. Approximately, how many squirrels did you harvest last season?

   1-5 ____ 6-10 ____ 11-15 ____ 16-20 ____ 21-25 ____ 26-30 ____ 31+_____

4. Approximately, how many days did you spend hunting squirrels last season?

   1-7 days ____ 8-14 days ____ 15-21 days _____ 22-28 days ____ 29+ days ____

5. How long have you hunted squirrels in Minnesota?

   1-5 years ____ 6-10 years ____ 11-15 years _____ 16-20 years ____ 21+ years ____

6. Do you hunt squirrels on public land ___, private land ___, or both ___?

7. In the past 5 years, have you hunted squirrels:

   on the same properties ___, on new properties ___, or both ___?

(Survey continues on next side)

8. What are the main obstacles for gaining access to property for squirrel hunting? (check all that apply)

 _____ I have not encountered obstacles

_____ Land is posted as no hunting or trespassing

_____ Not sure how to find additional public hunting lands

_____ Not comfortable asking for permission to access private land from landowner

_____ Denied access by landowner(s) in the past

_____ Difficulty finding owners of private land

_____ Other (please specify)____

9. Over the past 5 years, do you think squirrel populations in areas where you hunt are: decreasing____, about the same____, or increasing_____?

10. Based on your perception of squirrel population trends, would you recommend changes to
(check all that apply):
_____ Hunting regulations
_____ Habitat management
_____ Enforcement of regulations
_____ Other (Please Specify) ____
_____ I don't recommend any changes

11. How could the DNR improve your squirrel hunting experience?

____

____

____

Thank you for completing the survey! Please return the survey in the enclosed, postage-paid envelope.

Appendix 2.

Squirrel-Hunting Survey

Minnesota DNR is being asked by our hunters to evaluate our squirrel-hunting season. As part of this effort we are interested in what your state wildlife agency is doing regarding squirrel hunting seasons. Attached is a very short survey, which we hope you can complete and return via email by May 8, 2009.

Please forward to the correct person to complete this survey, if that is not you.

Thanks, in advance, for your help.

Emily Dunbar
emily.dunbar@dnr.state.mn.us
Wildlife Biologist
Minnesota DNR
35065 800th Ave
Madelia, MN 56062

1. Your name________________________
   Your position title __________________________
   Your email address_________________________________
   Your phone number _________________________________

2. Does your state/province have a fox and/or gray squirrel hunting season?
   Yes ____ No____
   If No, then the survey is complete. Please send this survey back.

3. When does the hunting season open?

4. When does the season close?

5. What is the daily bag limit and possession limit?

6. What are the shooting/hunting hours?

7. Does your state/province manage specifically for fox and/or gray squirrels ?
   Yes ____ No____
   If Yes, what are the management activities?

8. Does your state/province estimate fox and/or gray squirrel populations?
   Yes ____ No____
   If Yes, what techniques is used?

9. Is there currently any research being conducted by your agency on fox and/or gray squirrels?

   Yes _____ No______

   If so, please describe the study/studies:

10. Are there any issues surrounding squirrel populations or squirrel hunting in your state/province?

11. Other things we should know about your squirrel season:

Gray and Fox Squirrel Manuscripts

The following is a list of manuscripts relating to ecology, harvest mortality, and habitat management for fox and gray squirrels. Many of the manuscripts (marked with an asterisk) are summarized following this list.

Adams, C. E. 1976. Measurements and characteristics of fox squirrel, *Sciurus niger rufiventer*, home ranges. The American Midland Naturalist 95:211-215.

*Allen, D. L. 1943. Michigan fox squirrel management. Michigan Department of Conservation, Game Division Publication 100. 404pp.

Allen, J. M. 1952a. Gray and fox squirrel management in Indiana. Indiana Department of Conservation, Division of Fish and Game. Pittman-Robertson Bulletin 1. 112pp.

Allen, J. M. 1952b. Gray and fox squirrel management in Indiana. Revised edition. Indiana Department of Conservation, Division of Fish and Game. Pittman-Robertson Bulletin 1. 101pp.

Allen, J. M. 1954. Gray and fox squirrel management in Indiana, 2nd edition. Indiana Department of Conservation, Division of Fish and Game. Pittman-Robertson Bulletin 1. 112pp.

Allen, D. S., and W. Aspey. 1986. Determinants of social dominance in eastern grey squirrels (*Sciurus carolinensis*): a quantitative assessment. Animal Behaviour 34:81-89.

Armitage, K. B., and K. S. Harris. 1982. Spatial patterning in sympatric populations of fox and gray squirrels. American Midland Naturalist 132:227-233.

Baker, R. H. 1944. An ecological study of fox squirrels in eastern Texas. Journal of Mammalogy 25:8-23.

Bakken, A. 1952. Interrelationships of *Sciurus carolinensis* (Gmelin) and *Sciurus niger* (Linnaeus) in mixed populations. Ph.D. Dissertation, University of Wisconsin, Madison. 190 pp.

Barber, H.L. 1954. Gray and fox squirrel food habits investigations. Proceedings of the Annual Conference of the Southeastern Association of the Game and Fish Commission 8:191-197.

Barkalow, F. S., Jr. and R. F. Soots. 1965. An analysis of the effect of artificial nest boxes on a gray squirrel population. Transactions of the North American Wildlife Natural Resources Conference 30:349-360.

Barkalow, F. S., Jr. 1967. A record gray squirrel litter. Journal of Mammalogy 48:141.

Barkalow, F. S., Jr., R. B. Hamilton, and R. F. Soots, Jr. 1970. The vital statistics of an unexploited gray squirrel population. Journal of Wildlife Management 34(3):489-500.

Barkalow, F. S., Jr., and R. F. Soots, Jr. 1975. Life span and reproductive longevity of the gray squirrel, *Sciurus c. carolinensis* Gmelin. Journal of Mammalogy 56:522-524.

Baumgartner, L. L. 1938. Population studies of the fox squirrel in Ohio. Transactions of the North American Wildlife Natural Resources Conference 3:685-689.

*Baumgartner, L. L. 1939. Fox squirrel dens. Journal of Mammalogy 20(4):456-465.

Baumgartner, L. L. 1940a. The fox squirrel: its life history, habits, and management in Ohio. Abstracts of Documents. 33. The Ohio State University Press, Columbus. 8pp.

Baumgartner, L. L. 1940b. The fox squirrel: its life history, habits, and management. Ph.D. dissertation. Ohio State University, Columbus, OH. 257pp.

*Baumgartner, L.L. 1943. Fox Squirrels of Ohio. Journal of Wildlife Management 7:193-202.

Baumgras. P. 1944. Experimental feeding of captive fox squirrels. Journal of Wildlife Management 8:296-300.

Bendell, J. F. 1972. Population dynamics and ecology of the Tetraonidae. Proceedings of the International Ornithological Congress 15:81-89.

Benson, B. N. 1980. Dominance relationships, mating behaviour and scent marking in fox squirrels (*Sciurus niger*). Mammalia 44:143-160.

Boulware, J. T. 1941. Eucalyptus tree utilized by fox squirrel in California. American Midland Naturalist 26:696-697.

Bowers, M. A., and B. Breland. 1996. Foraging of gray squirrels on an urban-rural gradient: Use of the GUD to assess anthropogenic impact. Ecological Applications 6:1135-1142.

Brauer, A., and A. Dusing. 1961. Sexual cycles and breeding seasons of the gray squirrel, *Sciurus carolinensis* Gmelin. Transactions of the Kentucky Academy of Science 22(1-2):16-27.

Brown, L. G. and L. E. Yeager. 1945. Fox squirrels and gray squirrels in Illinois. Illinois Natural History Survey Bulletin 23:449-536.

Brown, B. W., and G. O. Batzli. 1984. Habitat selection by fox and gray squirrels: a multivariate analysis. The Journal of Wildlife Management 48:616-621.

Brown, B. W., and G. O. Batzli. 1985a. Field manipulations of fox and gray squirrel populations: how important is interspecific competition? Canadian Journal of Zoology 63:2134-2140.

Brown, B. W., and G. O. Batzli. 1985b. Foraging ability, dominance relations and competition for food by fox and gray squirrels. Transactions of the Illinois Academy of Science 78:61-66.

Bugbee, R. E. and A. Riegel. 1945. Seasonal food choices of the fox squirrel in western Kansas. Transactions of the Kansas Academy of Science 48:199-203.

*Burger, G. V. 1969. Response of gray squirrels to nest boxes at Remington Farms, Maryland. Journal of Wildlife Management 33:796-801.

Christisen, D. M. 1964. Squirrel management. Missouri Department of Conservation. 7pp.

Conner, L. M. L. J. Landers, and W. K. Michener. 1999. Fox squirrel density, habitat, and interspecific association with gray squirrels with minimally disturbed longleaf pine forests. Proceedings of the Annual Conference of the Southeastern Association of Fish and Wildlife Agencies 53:364-374.

*Conner, L. M. 2001. Survival and cause-specific mortality of adult fox squirrels in southwestern Georgia. Journal of Wildlife Management 65(2):200-204.

*Conner, L. M., and I. A. Godbois. 2003. Habitat associated with daytime refugia of fox squirrels in a longleaf pine forest. The American Midland Naturalist. 150(1):123-129.

Connolly, M.S. 1979. Time-tables in home range usage by gray squirrels (*Sciurus carolinensis*). Journal of Mammalogy 60:814-817.

Cordes, C. L. and F. S. Barkalow, Jr. 1972. Home range and dispersal in a North Carolina gray squirrel population. Proceedings of the Annual Conference of the Southeastern Association of the Game Fish Commission 26:124-135.

Davison, V. E. 1964. Selection of foods by gray squirrels. Journal of Wildlife Management 28:346-352.

Derge, K. L. 1997. Habitat use by sympatric eastern fox squirrels (*Sciurus niger vulpinus*) and gray squirrels (*Sciurus carolinensis*) at forest-farmland interfaces of the Valley and Ridge Province, Pennsylvania. M.S. thesis. Pennsylvania State University. 99pp.

*Derge, K. L., and R. H. Yahner. 2000. Ecology of Sympatric Fox Squirrels (*Sciurus niger*) and Gray Squirrels (S*ciurus carolinensis*) at Forest-farmland interfaces of Pennsylvania. The American Midland Naturalist 143(2):355-369.

Dexter, M. H., editor. 2008. Status of wildlife populations, fall 2008. Unpublished report, Division of Fish and Wildlife, Minnesota Department of Natural Resources, St. Paul, Minnesota. 300 pp.

Doebel, J. H. and B. S. McGinnes. 1974. Home range and activity of a gray squirrel population. Journal of Wildlife Management 38:860-867.

Don, B. A. C. 1983. Home range characteristics and correlates in tree squirrels. Mammal Review 13:123-132.

Drake, J. C., and F. J. Brenner. 1995. Comparison of habitat preferences of gray and fox squirrels in northwestern Pennsylvania. Journal of the Pennsylvania Academy of Science 69:73-76.

Ducharme, M. B., J. Larochelle, and D. Richard. 1989. Thermogenic capacity in gray and black morphs of the gray squirrel, *Sciurus carolinensis*. Physiological Zoology 62:1273-1292.

Dueser, R. D., J. L. Dooley, Jr., and G. J. Taylor. 1988a. Habitat structure, forest composition, and landscape dimensions as components of habitat suitability for the Delmarva fox squirrel. Pages 414-421 *in* R.C. Sczaro, K.E. Severson, and D.R. Patton,editors. Management of amphibians, reptiles, and small mammals in North America. U.S. Department of Agriculture, Forest Service, Technical Report RM, 1-166, 458 pp.

Dueser, R. D., J. L. Dooley, Jr., and G. J. Taylor. 1988b. Habitat structure, forest composition, and landscape dimensions as components of habitat suitability for the Delmarva fox squirrel. Pages 417-421 *in* Symposium for management of amphibians, reptiles, and small mammals in North America. Flagstaff, Arizona.

Edwards, J. W., D. C. Guynn, and M. R. Lennartz. 1989. Habitat use by southern fox squirrels in coastal South Carolina. Proceedings of the Annual Conference of the Southeastern Association of Fish and Wildlife Agencies 43:337-345.

*Edwards, J. W. and D. C. Guynn, Jr. 1995. Nest characteristics of sympatric populations of fox and gray squirrels. Journal of Wildlife Management 59:103-110.

Eveland, C. G. 1974. The dynamics of a protected population of fox squirrels (*Sciurus niger*). M.S. thesis. Southern Illinois University, Carbondale. 99pp.

Faeth, S. H., P. S. Warren, E. Shochat, and W. A. Marussich. 2005. Tropic dynamics in urban communities. Bioscience 55:399-407.

Fitzgibbon, C. D. 1993. The distribution of grey squirrel dreys in farm woodland: the influence of wood area, isolation and management. British Ecological Society 30(4):736-742.

Flyger, V., and R. Cooper. 1967. The utilization of nesting boxes by gray squirrels. Proceedings of the Annual Conference of the Southeast Association Game and Fish Commission 21:113-117.

Flyger, V. F., and D. A. Smith. 1980. A comparison of Delmarva fox squirrel and gray squirrel habitats and home range. Transactions of the Northeast Section of The Wildlife Society 37:19-22.

Flyger, V. F., and J. E. Gates. 1982. Fox and gray squirrels. Pages 209-229 *in* J.A. Chapman and G.A. Feldhamer, editors. Wild Mammals of North America. John Hopkins University Press, Baltimore, MA.

Fouch, W. R. 1961. Mast crops and fox squirrel populations at the Rose Lake Wildlife Experiment Station. Game Division Report 2332, Michigan Department of Conservation, Lansing. 4pp.

Fouch, W. R. 1962. Mast crops and fox squirrel populations at the Rose Lake Wildlife Experiment Station. Papers of the Michigan Academy of Science Arts Letters 47:211-217.

Fox, J. F. 1982. Adaptation of gray squirrel behavior to autumn germination by white oak acorns. Evolution 36(4):800-809.

Gates, J. E. 1991. Power line corridors, edge effects, and wildlife in forested landscapes of the central Appalachians. Pages 15-32 *in* J.E. Rodick and E.G. Bolen, editors. Wildlife and habitats in managed landscapes. Island Press, Washington, D.C.

Geeslin, H. G., Jr. 1970. A radio-tracking study of home range, movements, and habitat uses of fox squirrel (*Sciurus niger*) in east Texas. M.S. thesis. Texas A&M University, College Station. 118 pp.

Glennon, M. J., and W. F. Porter. 2007. Impacts of land-use management on small mammals in the Adirondack Park, New York. Northeastern Naturalist 14(3):323-342.

Goheen, J. R. and R. K. Swihart. 2003. Food hoarding behavior of gray squirrels and North American red squirrels in the central hardwoods: implications for forest regeneration. Canadian Journal of Zoology 81:1636-1639.

Goheen, J. R., R. K. Swihart, T. M. Gehring, and M. S. Miller. 2003. Forces structuring tree squirrel communities in landscapes fragmented by agriculture: species differences in perceptions of forest connectivity and carrying capacity. Oikos 102:95-103.

Goodrum, P. 1938. Squirrel management in east Texas. Transactions of the North American Wildlife Conference 3:670-676.

Goodrum, P.D. 1961. The gray squirrel in Texas. Texas Parks and Wildlife Department Bulletin 42:1-43.

Gorman, O. T., and R. R. Roth. 1989. Consequences of a temporally and spatially variable food supply for an unexploited gray squirrel (*Sciurus carolinensis*) population. American Midland Naturalist 121(1):41-60.

Gurnell, J. 1983. Squirrel numbers and the abundance of tree seeds. Mammal Review 13:133-148.

Gurnell, J. 1987. The natural history of squirrels. Facts on File, New York. 201pp.

Gurnell, J., L. A. Wauters, D. Preatoni, and G. Tosi. 2001. Spacing behavior, kinship, and population dynamics of grey squirrels in a newly colonized broadleaf woodland in Italy. Canadian Journal of Zoology 79:1533-1543.

Hansen, L. P. and C. M. Nixon. 1985. Effects of adults on the demography of fox squirrels (*Sciurus niger*). Canadian Journal of Zoology 63:861-867.

*Hansen, L. P., C. M. Nixon, and S. P. Havera. 1986. Recapture rates and length of residence in an unexploited fox squirrel population. The American Midland Naturalist 115:209-215.

Harnishfeger, R. L., J. L. Roseberry and W. D. Klimstra. 1978. Reproductive levels in unexploited woodlot fox squirrels. Transactions of the Illinois State Academy of Science 71:342-355.

Havera, S. P. 1978. Nutrition, supplemental feeding, and body composition of the fox squirrel, *Sciurus niger*, in central Illinois. Ph.D. dissertation. University of Illinois, Urbana. 130pp.

Havera, S. P., and C. M. Nixon. 1978. Interaction among adult female fox squirrels during their winter breeding season. Transactions of the Illinois State Academy of Sciences 71:24-38.

Havera, S. P., and C. M. Nixon. 1979. Energy and nutrient cost of lactation in fox squirrels. Journal of Wildlife Management 43:958-965.

*Havera, S. P. and K. E. Smith. 1979. A nutritional comparison of selected fox squirrel foods. Journal of Wildlife Management 43:691-704.

*Havera, S. P. and C. M. Nixon. 1980. Winter feeding of fox and gray squirrel populations. Journal of Wildlife Management 44:41-55.

Healy, W. M. and C. J. Welsh. 1992. Evaluation line transects to monitor gray squirrel populations. Wildlife Society Bulletin 20:83-90.

Heaney, L. R. 1984. Climatic influences on the life-history tactics and behavior of North American tree squirrels. Pages 43-78 *in* The biology of ground-dwelling squirrels, J. O. Murie and G. R. Michener, editors. University of Nebraska Press, Lincoln.

Hedrick, L. D. 1973. Silvicultural practices and tree squirrels (Sciurus L.) in east Texas. M.S. thesis. Texas A&M University, College Station. 97pp.

Hench, J. E., G. L. Kirkland, Jr., H. W. Setzer, L. W. Douglass. 1984. Age Classification for the gray squirrel based on eruption, replacement, and wear of molariform teeth. The Journal of Wildlife Management 48(4):1409-1414.

Herbers, J. and W. Klenner. 2007. Effect of logging pattern and intensity on squirrel demography. Journal of Wildlife Management 71(8):2655-2663.

Herkert, J. R., C. M. Nixon, and L. P. Hansen. 1992. Dynamics of exploited and unexploited fox squirrel (*Sciurus niger*) populations in midwestern United States. Pages 864-874 *in* Wildlife 2001: populations, D. R. McCullough and R. H. Barrett, editors. Elsevier Applied Science, New York, USA.

Hilliard, T. H. 1979. Radio-telemetry of fox squirrels in the Georgia coastal plain. M.S. thesis. University of Georgia, Athens. 121 pp.

Hoover, R. L. and L. E. Yeager. 1953. Status of the fox squirrel in northeastern Colorado. Journal of Mammalogy 34:359-365.

Horwich, R. H. 1972. The ontogeny of social behavior in the gray squirrel (*Sciurus carolinensis*). Berlin, Germany. 103pp.

Hungerford, K. E., and N. G. Wilder. 1941. Observations on the homing behavior of the gray squirrel (*Sciurus carolinensis*). Journal of Wildlife Management 5:458-460.

Ims, R. A. 1987. Male spacing systems in microtine rodents. American Naturalist 130:475-484.

Innes, S., and D. M. Lavigne. 1979. Comparative energetics of coat colour polymorphs in the eastern gray squirrel, *Sciurus carolinensis*. Canadian Journal of Zoology 57:585-592.

Jannett Jr., F. J., M. R. Broschart, L. H. Grim, and J. P. Schaberl. 2007. Northerly range extensions of mammalian species in Minnesota. American Midland Naturalist 158:168-176.

Jodice, P. G. R., and S. R. Humphrey. 1992. Activity and diet of an urban population of Big Cypress fox squirrels. Journal of Wildlife Management 56:685-692.

Jordan, J. S. 1971a. Yield from an intensively hunted population of eastern fox squirrels. U.S. Forest Service Research Paper NE-186, Northeast Forest Experimental Station. 8pp.

Jordan, J. S. 1971b. Dispersal period in a population of eastern fox squirrels (Sciurus niger). U.S. Forest Service Research Paper NE-216. 8pp.

Kantola, A. T., and S. R. Humphrey. 1990. Habitat use by Sherman's fox squirrel (*Sciurus niger shermani*) in Florida. Journal of Mammalogy 71:411-419.

Kemp G. A., and L. B. Keith. 1970. Dynamics and regulation of red squirrel (Tamiasciurus hudsonicus) populations. Ecology 51:765-779.

Kenward, R. 1985. Ranging behavior and population dynamics in grey squirrels. Symposia of the British Ecological Society 25:319-330.

Kline, P. D. 1964. Iowa squirrels: hunting statistics, sex and age ratios, and the influence of mast and agriculture. Proceedings of the Iowa Academy of Science 71:216-227.

Knee, C. 1983. Squirrel energetics. Mammal Review 13:113-122.

Koprowski, J. L., J. L. Roseberry, and W. D. Klimstra. 1988. Longevity records for the fox squirrel. Journal of Mammalogy 69:383-384.

*Koprowski, J. L. 1991. Response of fox squirrels and gray squirrels to a late spring-early summer food shortage. Journal of Mammalogy 72:367-372.

Koprowski, J. L. 1993a. Alternative reproductive tactics in male eastern gray squirrels: "Making the best of a bad job." Behavioral Ecology 4:165-171.

Koprowski, J. L. 1993b. Behavioral tactics, dominance, and copulatory success among male fox squirrels. Ethology, Ecology, and Evolution 5:169-176.

*Koprowski, J. L. 1994a. *Sciurus niger*. Mammalian Species 479:1-9.

*Koprowski, J. L. 1994b. *Sciurus carolinensis*. Mammalian Species 480:1-9.

Koprowski, J. L. 1996. Natural philopatry, communal nesting, and kinship in fox squirrels and gray squirrels. Journal of Mammalogy 77:1006-1016.

Koprowski, J. L. 1998. Conflict between the sexes: a review of social and mating systems of the tree squirrels. Pages 33-41 *in* Ecology and evolutionary biology of tree squirrels, M. A. Steele, J. F. Merritt, and D. A. Zegers, editors. Virginia Museum of Natural History, Martinsville.

Koprowski, J. L. 2005a. The response of tree squirrels to fragmentation: A review and synthesis. Animal Conservation 8:369-376.

Koprowski, J. L. 2005b. Management and conservation of tree squirrels: the importance of endemism, species richness, and forest condition. In: Connecting mountain islands and desert seas: biodiversity and management of the Madrean Archipelago II. U.S. Forest Service, Rocky Mountain Research Station, Ft. Collins, Colorado, USA.

Koprowski, J. L., and M. C. Corse. 2005. Time budgets, activity periods, and behavior of Mexican fox squirrels. Journal of Mammalogy 86(5): 947-952.

*Korschgen, L. J. 1981. Foods of fox and gray squirrels in Missouri. Journal of Wildlife Management 45:260-266.

Kotler, B. P., J. S. Brown, and M. Hickey. 1999. Food storability and the foraging behavior of fox squirrels (*Sciurus niger*). The American Midland Naturalist 142(1): 77-86.

Lee, C. L., D. A. Osborn, and K. V. Miller. 2007. Characteristics of a high density population of southern fox squirrels. American Midland Naturalist 159:385-393.

Lewis, A. R. 1980. Patch use by gray squirrels and optimal foraging. Ecology 61:1371-1379.

Lewis, A. R. 1982. Selection of nuts by gray squirrels and optimal foraging theory. American Midland Naturalist 107:250-257.

Lima, S. L., and T. J. Valone. 1986. Influence of predation risk on diet selection: A simple example in the grey squirrel. Animal Behaviour 34:536-544.

Linders, M. J., S. D. West, and M. Vander Haegen. 2004. Seasonal variability in the use of space by western gray squirrels in south central Washington. Journal of Mammalogy 85:511-516.

Liscinsky, S. A. 1951. The life history and habitat of the northern gray squirrel in relation to some plant communities in Centre County, PA. M.S. thesis. Pennsylvania State University, University Park. 178pp.

Ludwick, R. L., J. P. Fontenot, and H. S. Mosby. 1969. Energy metabolism of the eastern gray squirrel. Journal of Wildlife Management 33:569-575.

Madison, J. 1964. Gray and fox squirrels. Conservation Department, Olin Mathieson Chemical Corporation, East Alton, Illinois. 112pp.

Mankin, P. C. and R. E. Warner. 1997. Mammals of Illinois and the Midwest: ecological and conservation issues for human-dominated landscapes. Pages 135-153 *in* Conservation in highly fragmented landscapes , M. W. Schwartz, editor. Chapman and Hall, New York, USA.

Manski, D. A., L. W. VanDruff, and V. Flyger. 1981. Activities of gray squirrels and people in a downtown Washington, D.C., park: Management implications. Transactions of the North American Wildlife and Natural Resources Conference 46:439-454.

McCleery, R. A., R. R. Lopez, N. J. Silvy, and S. N. Kahlick. 2007. Habitat use of fox squirrels in an urban environment. Journal of Wildlife Management 71(4):1149-1157.

*McCleery, R. A., R. R. Lopez, N. J. Silvy, and S. N. Kahlick. 2008. Fox squirrel survival in urban and rural environments. Journal of Wildlife Management 72(1):133-137.

McCloskey, R. J., and P. A. Vohs, Jr. 1971. Chronology of reproduction of the fox squirrel in Iowa. Proceedings of the Iowa Academy of Science 78:12-15.

McComb, W. C. and R. E. Noble. 1981. Nest box and natural cavity use in three mid-south forest habitats. Journal of Wildlife Management 45:93-101.

McComb, W. C. 1984. Managing urban forests to increase or decrease gray squirrel populations. Southern Journal of Applied Forestry 8:31-34.

Mech, S. G., and P. A. Zollner. 2002. Using body size to predict perceptual range. Oikos 98:47-52.

Merritt, J. E. 1987. Guide to the mammals of Pennsylvania. University of Pennsylvania. Pittsburgh Press, Pennsylvania. 408pp.

Merson, M. H., C. J. Cowles, and R. L. Kirkpatrick. 1978. Characteristics of captive gray squirrels exposed to cold and food deprivation. The Journal of Wildlife Management 42:202-205.

Moncrief, N. D., R. A. Van Den Bussche, R. D. Dueser, D. Loftis, N. E. Cockett, and M. Culver. 2008. Diagnostic genetic marker that differentiates eastern fox squirrels from eastern gray squirrels. Journal of Wildlife Management 72(1):320-323.

Moore, E. M., A. B. McEuen, R. K. Swihart, T. A. Contreras, and M. A. Steele. 2007. Determinants of seed removal distance by scatter-hoarding rodents in deciduous forests. Ecology 88(10): 2539-2540.

Moore, J. C. 1957. The natural history of the fox squirrel, *Sciurus niger shermani*. Bulletin of the American Museum of Natural History 113:1-71.

Moore, J. E. and R. K. Swihart. 2005. Modeling patch occupancy by forest rodents: incorporating detectability and spatial autocorrelation with hierarchically structured data. Journal of Wildlife Management 69(3):933-949.

Morgan, R. A., J. S. Brown, and J. M. Thorson. 1997. The effect of spatial scale on the functional response of fox squirrels. Ecology 78(4):1087-1097.

*Mosby, H. S. 1969. The influence of hunting on the population dynamics of a woodlot gray squirrel population. Journal of Wildlife Management 33(1):59-73.

Mumford, R. E, and J. O. Whitaker Jr. 1982. Mammals of Indiana. Indiana University Press, Bloomington, USA.

Newton, I. 1994. The role of nest sites in limiting the numbers of hole-nesting birds: A review. Biological Conservation 70:265-276.

Nixon, C. M. 1968. Squirrel management guidelines for wildlife areas. Ohio Department of Natural Resources, Division of Wildlife In-service Document 55. 8pp.

*Nixon, C. M. and M. W. McClain. 1969. Squirrel population decline following a late spring frost. Journal of Wildlife Management 33:353-357.

Nixon, C. M., and W. J. Harper. 1972. Composition of gray squirrel milk. Ohio Journal of Science 72:3-6.

*Nixon, C. M., R. W. Donohoe, and T. Nash. 1974. Overharvest of fox squirrels from two woodlots in western Ohio. Journal of Wildlife Management 38(1):67-80

*Nixon, C. M. and M. W. McClain. 1975. Breeding seasons and fecundity of female gray squirrels in Ohio. Journal of Wildlife Management 39:426-438.

*Nixon, C. M., M. W. McClain, and R. W. Donohoe. 1975. Effects of hunting and mast crops on a squirrel population. Journal of Wildlife Management 39:1-25.

*Nixon, C. M., and R. Donohoe. 1979. Squirrel nest boxes-are they effective in young hardwood stands? Wildlife Society Bulletin 7:283-284.

Nixon, C. M., S. P. Havera, and L. P. Hansen. 1980a. Initial response of squirrels to forest changes associated with selection cutting. Wildlife Society Bulletin 8:298-306.

*Nixon, C. M., M. W. McClain, and R. W. Donohoe. 1980b. Effects of clear-cutting on gray squirrels. The Journal of Wildlife Management 44:403-412.

*Nixon, C. M., S. P. Havera, and L. P. Hansen. 1984. Effects of nest boxes on fox squirrel demography, condition and shelter use. American Midland Naturalist 112:157-171.

Nixon, C. M., and L. P. Hansen. 1987. Managing forests to maintain populations of gray and fox squirrels. Illinois Department of Conservation Technical Bulletin 5:1-35.

Nupp, T. E. 1997. Population dynamics and community structure of granivorous forest rodents in fragmented landscape. PhD. dissertation. Purdue University, West Lafayette, IN.

Nupp, T. E., and R. K. Swihart. 2000. Landscape-level correlates of small-mammal assemblages in forest fragments of farmland. Journal of Mammalogy 81:512-526.

Nupp, T. E., and R. K. Swihart. 2001. Assessing competition between forest rodents in a fragmented landscape of midwestern USA. Mammalian Biology 66:1-12.

Ostfeld, R. S. 1990. The ecology of territoriality in small mammals. Trends of Ecology and Evolution 5:411-415.

Packard, R. L. 1956. The tree squirrels of Kansas: ecology and economic importance. University of Kansas, Museum of Natural History, Miscellaneous Publication 11:1-67.

Pasch, B., and J. L. Koprowski. 2006. Sex differences in space use of Chiricahua fox squirrels. Journal of Mammalogy 87(2):380-386.

Pasitschniak-Arts, M., and J. F. Bendell. 1990. Behavioural differences between locally recruiting and dispersing gray squirrels, *Sciurus carolinensis*. Canadian Journal of Zoology 68:935-941.

*Perkins, M. W., and L. M. Conner. 2004. Habitat use of fox squirrels in southwestern Georgia. Journal of Wildlife Management 68(3): 509-513.

Perkins, M. W., L. M. Conner, and M. B. Howze. 2008. The importance of hardwood trees in the longleaf pine forest ecosystem for Sherman's fox squirrels. Forest Ecology and Management 255:1618-1625.

*Peterle, T. J., and W. R. Fouch. 1959. Exploitation of a fox squirrel population on a public shooting area. Michigan Department of Conservation Report 2251. 4pp.

Peterson, J. C., R. N. Hasenyager and A. W. Heggen. 1978. Use of artificial nest boxes by Alberts squirrels. Southwestern Naturalist 23:700-702.

Redmond, H. R. 1953. Analysis of gray squirrel breeding studies and their relation to hunting season, gunning pressure, and habitat conditions. Transactions of the North American Wildlife Conference 18:378-389.

Robb, J. R., M. S. Cramer, A. R. Parker, and R. P. Urbanek. 1996. Use of tree cavities by fox squirrels and raccoons in Indiana. Journal of Mammalogy 77:1017-1027.

Root, J. J., P. T. Oesterle, H. J. Sullivan, J. S. Hall, N. L. Marlenee, R. G. McLean, J. A. Montenieri, and L. Clark. 2007. Short Report: Fox squirrel (*Sciurus niger*) associations with West Nile virus. The American Society of Tropical Medicine and Hygiene 76(4):782-784.

Rosenblatt, D. L., Heske, E. J. Nelson, S. L. Barber, D. M. Miller, A., and B. MacAllister. 1999. Forest fragments in east-central Illinois: islands or habitat patches for animals? American Midland Naturalist 141:115-123.

Rushton, S. P., J. Gurnell, P. W. W. Lurz, and R. M. Fuller. 2002. Modeling impacts and costs of gray squirrel control regimes on the viability of red squirrel populations. The Journal of Wildlife Management 66(3):683-697.

Ryan, L. A., and A. B. Carey. 1995. Distribution and habitat of the western gray squirrel (*Sciurus griseus*) on Ft. Lewis, Washington. Northwest Science 69(3):204-216.

*Salsbury, C. M., R. W. Dolan, and E. B. Pentzer. 2004. The distribution of fox squirrel (*Sciurus niger*) leaf nests within forest fragments in central Indiana. The American Midland Naturalist 151(2):369-377.

*Salsbury, C. M. 2008. Distribution patterns of *Sciurus niger* (Eastern fox squirrel) leaf nests within woodlots across a suburban/urban landscape. Northeastern Naturalist 15(4):485-496.

*Sanderson, H. R. 1975. Den-tree management for gray squirrels. Wildlife Society Bulletin 3:125-131.

Sanderson, H. R., C. M. Nixon, R. W. Donohoe, and L. P. Hansen. 1980. Grapevines-an important component of gray and fox squirrel habitat. Wildlife Society Bulletin 8:307-310.

Schmidt, K. A. 2000. Interactions between food chemistry and predation risk in fox squirrels. Ecology 81(8): 2077-2085.

Sexton, O. J. 1990. Replacement of fox squirrels by gray squirrels in a suburban habitat. American Midland Naturalist 124:198-205.

Sharp, W. M. 1959. A commentary on the behavior of free-running gray squirrels. Proceedings of the Southeastern Association Game and Fish Commissioners 13:382-387.

Shaw, S. P. 1971. Wildlife and oak management. Pages 84-89 *in* Proceedings of the Oak Symposium, Northeast Forest Experimental Station, U.S. Department of Agriculture, Forest Service, Upper Darby, PA.

Shaw, W. W., and W. R. Mangun. 1984. Nonconsumptive use of wildlife in the United States. United States Fish and Wildlife Service Resource Publication 154. 20 pp.

Shepard, B. F., and R. K. Swihart. 1995. Spatial dynamics of fox squirrels (*Sciurus niger*) in fragmented landscapes. Canadian Journal of Zoology 73:2098-2105.

Shifley, S. R., F. R. Thompson III, W. D. Dijak, M. A. Larson and J. J. Millspaugh. 2006. Simulated effects of forest management alternatives on landscape structure and habitat suitability in the midwestern United States. Forest Ecology and Management 229:361-377.

Short, H. L., and W. B. Duke. 1971. Seasonal food consumption and body weights of captive tree squirrels. The Journal of Wildlife Management 35:435-439.

Short, H. L. 1976. Composition and squirrel use of acorns of black and white oak groups. Journal of Wildlife Management
40:479-480.

Shorten, M. 1951. Some aspects of the biology of the grey squirrel (*Sciurus carolinensis*) in Great Britain. Proceedings of the Zoological Society of London 121:427-459.

Shorten, M., and F. A. Courtier. 1955. A population study of the gray squirrel (*Sciurus carolinensis*) in May 1954. Annuals of Applied Biology 43(3):494-510.

Smallwood, P. D., and W. D. Peters. 1986. Grey squirrel preferences: The effects of tannin and fat concentration. Ecology 67(1):168-174.

Smith, N. B. 1967. Some aspects of reproduction in the female gray squirrel, *Sciurus* carolinensis Gmelin, in Wake County, North Carolina. M.S. thesis. North Carolina State University, Raleigh. 92pp.

Smith, N. B., and F. S. Barkalow, Jr. 1967. Precocious breeding in the gray squirrel. Journal of Mammalogy 48:328-330.

*Smith, C. C., and D. Follmer. 1972. Food preferences of squirrels. Ecology 53(1):82-91.

Spritzer, M. D. 2002. Diet, microhabitat use and seasonal activity patterns of gray squirrels (*Sciurus carolinensis*) in hammock and upland pine forest. The Midland Naturalist 148(2):271-281.

Spurgin, R. J., and J. A. Hnida. 2002. *Eimeria lancasterensis* and *Eimeria ontarioensis* from fox squirrels, *Sciurus niger*, in southeastern Nebraska, U.S.A. Comparative Parasitology 69(2):211-212.

Steele, M. A. 1988. Patch use and foraging behavior by the fox squirrel (Sciurus niger): tests of theoretical predictions. Ph.D. dissertation, Wake Forest University, Winston Salem, NC. 220pp.

Steele, M. A., and J. L. Koprowski. 2001. North American tree squirrels. Smithsonian Institution Press, Washington, D. C., USA.

Swihart, R. K., and T. E. Nupp. 1998. Modeling population responses of North American tree squirrels to agriculturally induced fragmentation of forests. Pages 1-19 *in* M. A. Steele, J. F. Merritt and D. A. Zegers, editors. Ecology and evolutionary biology of tree squirrels. Virginia Museum of Natural History Special Publication 6.

Tappe, P. A., and D. C Guynn. 1998. Southeastern fox squirrels: r- or K-selected? Implications for management. Pages 239-249 *in* M. A Steele, J. F. Merritt, and D. A. Zegers, editors. Ecology and evolutionary biology of tree squirrels. Virginia Museum of Natural History Special Publication 6.

Taylor, G. J. 1973. Present status and habitat survey of the Delmarva fox squirrel (*Sciurus niger cinereus*) with a discussion of reasons for its decline. Proceedings of the Southeastern Association of Game and Fish Commissioners 27:278-289.

Terrill, H. V. 1941. A preliminary study of the western fox squirrel, *Sciurus niger rufiventer* (Geoffrey), in Missouri. M.A. thesis. University of Missouri, Columbia, MO. 164pp.

Thompson, D. C. 1977a. Diurnal and seasonal activity of the grey squirrel (*Sciurus carolinensis*). Canadian Journal of Zoology 55:1185-1189.

Thompson, D. C. 1977b. Reproductive behavior of the grey squirrel. Canadian Journal of Zoology 55:1176-1184.

*Thompson, D. C. 1978. Regulation of a northern grey squirrel (*Sciurus carolinensis*) population. Ecology 59:708-715.

Thompson, D. C. and P. S. Thompson. 1980. Food habits and caching behavior of urban grey squirrels. Canadian Journal of Zoology 58:701-710.

Uhlig, H. G. 1955. The gray squirrel: its life history, ecology, and population characteristics in West Virginia. W. Virginia Conservation Commission. Pittman-Robertson Project, 31-R. 175pp.

Uhlig, H. 1956. The gray squirrel in West Virginia. West Virginia Conservation Commission, Division of Game Management Bulletin 3. 83pp.

U.S. Department of Agriculture. 1948. Woody-plant seed manual. U.S. Department Of Agriculture, Forest Service Miscellaneous Publications 654. 416pp.

Webley, G. E., and E. Johnson. 1983. Reproductive physiology of the grey squirrel (*Sciurus carolinensis*). Mammal Review 13:149-154.

Weigl, P. D., M. A. Steele, L. J. Sherman, and J. C. Ha. 1989. The ecology of the fox squirrel (*Sciurus niger*) in North Carolina: implications for survival in the Southeast. Bulletin of Tall Timbers Research Station 24:1-93.

Weigl, P. D., L. J. Sherman, A. I. Williams, M. A. Steele and D. S. Weaver. 1998. Geographic variation in the fox squirrel (*Sciurus niger*): a consideration of size cline, habitat vegetation, food habits and historical biogeography. Pages 171-184 *in*: Ecology and evolutionary biology of tree squirrels, M. A. Steele, J. F. Merritt, and D. A. Zegers, editors. Virginia Museum of Natural History Special Publication 6.

Whitaker, J. O., and W. J. Hamilton. 1998. Mammals of the eastern United States. Third edition. Cornell University Press, Ithaca, New York, USA.

Wilson, D. E. and S. Ruff. 1999. The Smithsonian book of North American mammals. Ronald Press Company, New York. 750pp.

Yeager, L. E. 1959. Status and population trend in fox squirrels on fringe range, Colorado. Journal of Wildlife Management 23:102-107.

Zegers, D. A. 1985. Eastern fox squirrel. Pages 399-402 *in*: Species of special concern in Pennsylvania, H. H. Genoways and F. J. Brenner, editors. Carnegie Museum of Natural History, Special Publication, No.11, Pittsburgh, PA.

Zollner, P. A. 2000. Comparing the landscape level perceptual abilities of forest sciurids in fragmented agricultural landscapes. Landscape Ecology 15:523-533.

Summaries of Selected Manuscripts

Allen, D.L. 1943. Michigan fox squirrel management. Michigan Department of Conservation, Game Division Publication 100. 404pp.

- Oak-hickory forests can sustain high numbers of fox squirrels.
- Sanctuaries do not guarantee continuous abundance.
- Disease is an important mortality factor.
- The "breeding potential" of fox squirrels is such that the population can more than triple itself by late summer.
- Mating activity of the fox squirrel occurs primarily in January and February, and again in May and June.
- The average production per year is four young per female.
- The normal fall ratio is made up of two-thirds juvenile animals, and one-third adult animals.
- Tree dens give squirrels more protection from weather, natural enemies, and man than do leaf nests.
- Long-time management program should aim to provide tree dens.
- The domestic dog is probably the most important mammalian predator of squirrels.
- The remedy for overshooting is to limit harvest.
- The percentage of the total harvest is fairly constant across the entire season, so adding a week to the season would increase the harvest substantially, and subtracting a week would decrease the harvest.

Baumgartner, L.L. 1939. Fox squirrel dens. Journal of Mammalogy 20(4):456-465.

- Den formation depends upon 1) rate of tree growth, 2) rate of tree decay, 3) age of tree, and 4) method of scar tissue formation.
- Den cavity formation can take between 8-30 years depending on the type of tree.
- Dens are typically in use for 10-20 years and become less appealing as the cavity becomes larger.
- Dens will naturally form in older forests if mature trees are left standing.
- The artificial production of dens by girdling limbs is not practical or economically feasible.
- Observations have shown that young forests can be depleted of fox squirrels in 1 hunting season due to lack of den trees.

Baumgartner, L.L. 1943. Fox squirrels of Ohio. Journal of Wildlife Management 7:193-202.

- Fox squirrel habitat in Ohio consists of maple-beech-oak woodlots of 5-300 acres in size, while gray squirrel habitat consists of maple-beech forests.
- Oak-hickory habitat is the most preferred habitat of squirrels.
- Isolated, mature forest remnants support fewer squirrels than sub-climax woodlots because 1) old trees appear to supply fewer food resources than young trees and 2) the remaining mature beech-maple forests supply a good mast crop only once every 3-5 years.
- Selective cutting, pasturing, and planted woodlots create temporary squirrel habitat that can support a squirrel population seasonally or for 1 year out of 3 - 4.
- Male squirrels moved an average of 154 yds/day, and females moved 130 yd/day.
- Squirrels may infiltrate a field that is planted adjacent to a woodlot and will go further into a cut soybean field than a corn or wheat field.
- Woodlots that are unhunted become over populated in the fall, and squirrels will emigrate to other less populated woodlots.
- Leaf nests are temporary and are used as escape cover and to bear and rear young.

Burger, G.V. 1969. Response of gray squirrels to nest boxes at Remington Farms, Maryland. Journal of Wildlife Management 33:796-801.

- Artificial dens increased the number of squirrels trapped 1 year after installation. Populations in control units remained essentially unchanged over the same period.
- On control units, age ratios averaged 1.3-1.8 young per adult. Units with den structures averaged 2.4 and 2.6 young per adult.
- Costs of construction and installation may make erection of artificial dens uneconomical in large habitat units.
- Artificial dens would appear to furnish a useful and productive tool where intensive management is needed.

Conner, L. M. 2001. Survival and cause-specific mortality of adult fox squirrels in southwestern Georgia. Journal of Wildlife Management 65(2): 200-204.

- Adult squirrel mortality in an unhunted population is about 10% each season.
- No differences in survival between sexes or seasons.
- Fox squirrels survival is higher than gray squirrels.
- Survival rates estimated during this study indicate that fox squirrels may have higher survival rates in the southeastern U.S. than in the midwestern U.S.

Conner, L.M., and I.A. Godbois. 2003. Habitat associated with daytime refugia of fox squirrels in a longleaf pine forest. American Midland Naturalist 150(1):123-129.

- Hardwood trees had a 56% greater chance of being used as a refuge tree than pine trees.
- Tree height and density was positively associated with probability of use as refugia.
- Understory vegetation appears to be an unimportant variable for fox squirrels when selecting daytime refuge sites.
- Mature hardwood trees within an open-canopy pine stand are significant refuge sites.

Derge, K. L., and R. H. Yahner. 2000. Ecology of sympatric fox squirrels (*Sciurus niger*) and gray squirrels (S*ciurus carolinensis*) at forest-farmland interfaces of Pennsylvania. The American Midland Naturalist 143(2):355-369.

- Habitat use differed between fox and gray squirrels.
- Fox squirrels occurred closer to forest edge and in areas with fewer short shrubs than gray squirrels.
- Gray squirrels used habitats with fewer understory logs and trees and habitats with greater basal areas of snags.

Edwards, J.W. and D.C. Guynn, Jr. 1995. Nest characteristics of sympatric populations of fox and gray squirrels. Journal of Wildlife Management 59:103-110.

- Gray squirrel use of cavities was greatest during winter.
- Fox squirrel use of cavities was greatest during fall and winter.
- Gray squirrels use tree dens more often than fox squirrels for all seasons.
- Fox squirrels rarely included vines in their leaf nests while gray squirrels often used them.
- Fox squirrel leafs nests were higher and located in taller trees of larger diameter than gray squirrels.
- Both squirrels built leaf nests in pine and oak trees more frequently than expected.
- Placement of artificial cavities in pines and at heights of 15-20m may increase use by fox squirrels.

Hansen, L. P., C. M. Nixon, and S. P. Havera. 1986. Recapture rates and length of residence in an unexploited fox squirrel population. The American Midland Naturalist 115:209-215.

- Adults consisted of 79% of the population during autumn and 86% during spring.
- Length of residence of fox squirrels that were juveniles at first capture was shorter than for squirrels that were subadults at first capture.

- Length of residence of both juveniles and subadult squirrels was shorter than that of yearling and adult squirrels
- Recapture rates of yearling and adult fox squirrels also were greater than juveniles and subadults.
- 99.1% population turnover in 11.7 years indicates high survival of fox squirrel on the study area.
- Results suggest that resident adults may limit recruitment of squirrels into a population, thereby controlling maximum densities.
- Annual adult survivorship is generally >60% with average annual mortality estimated as 34% for males and 37% for females.

Havera, S.P. and K.E. Smith. 1979. A nutritional comparison of selected fox squirrel foods. Journal of Wildlife Management 43:691-704.

- Shagbark hickory and mockernut hickory had the highest caloric values, whereas white oak and corn embryos had the lowest values.
- Oaks have similar coefficients of energy metabolism that fell between the lower values of the grains and the higher values of walnuts, shagbark, and mockernut hickory.
- Squirrels fed a white oak diet had the highest mean levels of food intake, metabolized energy and weight gain.
- Red oak was the only mast with a lower amount of metabolized energy per day than corn and soybeans.
- The higher coefficients of metabolism and the higher caloric values for shagbark and mockernut hickory nuts, as compared to acorns, require that adult female fox squirrels eat approximately 59% more white, bur, and black oak acorns than hickory nut.
- An average white oak crop in addition to black oak acorns, shagbark and mockernut hickories, and black walnuts, should supply fox squirrels with sufficient energy throughout the winter.
- Corn should not be used in winter as supplemental feeding due to its low energy and mineral nutritional values.

Havera, S.P. and C.M. Nixon. 1980. Winter feeding of fox and gray squirrel populations. Journal of Wildlife Management 44:41-55.

- Study found few beneficial effects on squirrel populations from winter feeding with corn during 3 winters with above-average mast crops in a mature mixed hardwood forest and an even-aged oak-hickory forest.
- In mature oak-hickory study areas, winter feeding did not increase density, survival, or reproduction, nor did it decrease the number of squirrels with mange.

- In oak-hickory study areas, the recapture rate on the feed area were significantly higher than on the control area.
- Supplemental feeding in even-aged timber may be more justified than feeding in mature timber, because fox squirrels collected on the even-aged timber control area had better physiological indices than fox squirrels collected from areas in mature timber.
- To benefit reproduction, winter feeding should be started before December.

Koprowski, J. L. 1991. Response of fox squirrels and gray squirrels to a late spring-early summer food shortage. Journal of Mammalogy 72:367-372.

- Late spring frost killed available fruit crop in 1987.
- Mulberries and hackberries are the major component of the diet of squirrels in a year of typical abundance, but declined from 79% of identified food items in 1988 to 8% of identified food items in 1987.
- In 1987, juvenile survival was lower (39%) than adult survival (94%).
- Juvenile and adult survival were similar during 1988.
- Weights of adults and juveniles were significantly less in 1987 compared with 1988.

Koprowski, J. L. 1994a. *Sciurus niger*. Mammalian Species 479:1-9.

- Article describes taxonomy, physical characteristics, distribution, and genetics of fox squirrels.
- Author conducts literature review of fox squirrel reproduction, ecology, and behavior.

Koprowski, J.L. 1994b. *Sciurus carolinensis*. Mammalian Species, 480:1-9.

- Article describes taxonomy, physical characteristics, distribution, and genetics of gray squirrels.
- Author conducts literature review of gray squirrel reproduction, ecology, and behavior.

Korschgen, L.J. 1981. Foods of fox and gray squirrels in Missouri. Journal of Wildlife Management 45:260-266.

- Fox squirrels ate 109 identified plant foods, 18 which made up > 80% of all foods eaten.
- For fox squirrels, hickories of 7 species were selected most often and in the greatest amount (29%).
- For fox squirrels, oaks of 11 species made up 23% of diet.
- Gray squirrels ate 97 identified plant foods and 14 animal foods, 18 which made up > 86% of all foods eaten.

- The most effective management for oak-hickory forest was to allow it to reach maturity.
- Uneven-aged stands that include a variety of oaks and hickories are the most dependable food producers.
- Gray squirrels rely more on oak-hickory mast (73% of annual diet) than fox squirrels (52% of annual diet).
- Mulberry, osage orange, white elm, maples, and grapes are important as seasonal or supplemental food sources.

McCleery, R.A., R.R. Lopez, N.J. Silvy, and S.N. Kahlick. 2008. Fox squirrel survival in urban and rural environments. Journal of Wildlife Management 72(1):133-137.

- Monthly survival rates of rural fox squirrels was lower than urban fox squirrels over same 12-month period.
- When comparing 24-month period for urban fox squirrels with 18-month period for rural squirrels, survival rates were similar between urban and rural squirrels.
- Data suggests that sex and season may influence survival of urban squirrels but not rural squirrels.
- >60% of fatalities on rural site caused by predation.
- >5% of fatalities on urban site caused by predation and >60% of urban squirrel fatalities caused by motor vehicle collisions.
- Management efforts should not assume demographic rates of rural populations are pertinent in the management of urban populations.

Mosby, H.S. 1969. The influence of hunting on the population dynamics of a woodlot gray squirrel population. Journal of Wildlife Management 33(1):59-73.

- Turnover period for the control group was 6.2 years compared with 7.2 for the hunted population.
- The mean annual mortality rate for the control population was 42.4% and 47.6% for the hunted population.
- Natural losses accounted for 25.2% of the mean annual mortality for the control population and only 10.2% for the hunted population.
- Study suggests that hunting removed a proportion (37.4%) of the population that would have been lost to “natural” causes.
- 37.4% of squirrels were removed from the hunted population due to harvest and 17.3% of squirrels were removed from the control population due to trap mortality.
- Author concludes that harvest of 38% of the squirrel population did not affect recruitment in the hunted population, had no significant influence on the mean annual mortality rate,

and probably removed a segment of the population that would normally be lost to "natural losses."

Nixon, C.M. and M.W. McClain. 1969. Squirrel population decline following a late spring frost. Journal of Wildlife Management 33:353-357.

- A late spring frost killed available seed crop.
- The following fall, the population had a lower reproductive rate, lighter body weight, and increased dispersal rate (for subadults).
- Breeding stopped for the rest of the year and did not resume until late summer of the next year.
- In a year with a good mast crop, 36% of females produced two litters while none did so in years of mast failure.

Nixon, C. M., R. W. Donohoe, and T. Nash. 1974. Overharvest of fox squirrels from two woodlots in western Ohio. Journal of Wildlife Management 38(1):67-80.

- Results of harvest data (1965-71) for 54 acres of public hunting woodlots in western Ohio.
- Harvest of 1 squirrel/acre for 1st 4 years, and 0.60 squirrel/acres last 3 years.
- Harvest related to preseason squirrel density, but not to hunting pressure.
- High yield due to high, sustained hunting pressure (mean= 4.3 gun-hours per acre for all years) and easy access.
- Of 759 hunters surveyed, 66% killed no squirrels, 6.0% harvested > 1 squirrel per trip, and only four hunters killed a bag limit of four squirrels.
- 73% of squirrel hunting occurred in September, 23% in October, and only 4.4% in November.
- 35.6% of September hunters were successful, but only 25.1% of October hunters were successful.
- Nearly 75% of total harvest was young-of-the-year 59.8% subadults, and 14.0% juveniles.
- Annual mortality rate of 91.8% for adults and subadults (75.2% of mortality was from hunting)
- The early hunting season was harmful to population maintenance because nursing females were harvested, resulting in the loss of nestlings that could have contributed to the harvest.
- In western Ohio, weaning of litters is almost complete by September 15th.

- Starting hunting season after September 15th would make juveniles available and would reduce the harvest of nursing adult females, who are more vulnerable to shooting at this time.

Nixon, C.M. and M.W. McClain. 1975. Breeding seasons and fecundity of female gray squirrels in Ohio. Journal of Wildlife Management 39:426-438.

- Winter breeding peaks in January.
- Spring-summer breeding peaks between mid-May and mid-June.
- The two breeding peaks occur when photoperiod and temperature are increasing.
- Mortality from implantation to mid fall of spring-born young averaged 15.9%, while summer-born young averaged 68.1%.
- 56% of yearling females breed during either the winter or summer breeding period.
- The age ratio of gray squirrels in the study area was 48% young-of-the-year.
- 60% of adult females breed during each breeding period, even in years when food availability is high.
- Gray squirrels are organized into social units that are dominated by adult females.

Nixon, C.M., M.W. McClain, and R.W. Donohoe. 1975. Effects of hunting and mast crops on a squirrel population. Journal of Wildlife Management 39:1-25.

- In Waterloo, Ohio, 63% hunters surveyed killed no squirrels per trip, 8.5% shot 1 squirrel, 8.5% shot 2 squirrels, and 6% shot 3 or more.
- Hunters were more successful in September (40.1%), than October (35.9%), or November (24.6%).
- Composition of gray squirrels was 51.4% adults and 48.6% juveniles; for fox squirrels 53.2% of the population were adults and 46.8% were juveniles.
- Juveniles made up 5.4% of the September harvest and 35% of the October harvest.
- Adults declined from 42.4% of the harvest in September to 25.3% in October.
- The average annual mortality rate for adult fox squirrels was 80%; for subadults it was 69%.
- Hunting accounted for 55.2% of the annual mortality.
- Population densities of squirrels were dependent upon the mast crop of the previous fall and harvest.
- Harvest and densities affected the survival of adult female gray squirrels.
- A good mast crop improved the survival of summer-born young, increased fecundity of breeding females, reduced emigration rates for juveniles and subadults, and increased survival of adult gray squirrels due to the larger size of hickory crop.
- An opening date after September 20th would eliminate the killing of nursing females by hunters.

Nixon, C.M., and R. Donohoe. 1979. Squirrel nest boxes-are they effective in young hardwood stands? Wildlife Society Bulletin 7:283-284.

- 10 nest boxes were placed in a 21 year-old, 1.9 ha clear-cut, located in Vinton County, Ohio, and in April 20 nest boxes were placed in a 4.0 ha, 32-36 year-old stand located in Pope County, Illinois.
- Gray squirrels made little use of boxes located in the 21-year-old woods, while gray squirrels were using 7 boxes by August and 16 boxes by December of that year.

Nixon, C.M., M.W. McClain, and R.W. Donohoe. 1980b. Effects of clear-cutting on gray squirrels. The Journal of Wildlife Management 44:403-412.

- The number of adult and subadult squirrels captured declined about 54% and densities decreased 44% 1 year after 12.9 ha clear-cut.
- Number of spring-born subadults decreased after clear-cutting from 36.6% of the squirrels captured in 1968 and 1970 to only 8.6% of the squirrels captured in 1972-1973.
- The loss of a large number of tree cavities within the clear-cut is believed to have contributed to the decline in subadults captured after 1971.
- The reluctance of squirrels to exploit clear-cuts appears to decrease after 15 years following clear-cutting.
- Food production < 22 years post-cut was lower than the uncut forest.
- Smaller clear-cuts (7.9 and 3.8 ha) did not affect squirrel densities, recovery rates, breeding rates, movements, or body weights.
- Clear-cuts kept narrower than 160m should allow most squirrels to retain some portion of their original home range, and should improve the likelihood of the squirrels tolerating the logging operation.
- The authors recommend keeping uncut travel lanes of mature trees, 50-100m wide if cutting units > 8 ha.
- Use of small (<8 ha) and narrow (<160m) clear-cuts where 40-60% of the stands are retained in a seed-producing age should not significantly reduce squirrel populations.

Nixon, C.M., S.P. Havera, and L.P. Hansen. 1984. Effects of nest boxes on fox squirrel demography, condition and shelter use. American Midland Naturalist 112:157-171.

- No evidence that use of artificial shelters by fox squirrels improves either survival or birth rate.
- The infrequent use of boxes by fox squirrels may be the result of a low rate of acceptance by breeding females.
- Survival and density of adult males increased on both study areas in the presence of boxes.
- There was a positive correlation between use by breeding females and the height of boxes.

- Male squirrels preferred boxes in mature, mixed species forests, whereas females preferred boxes close the edge of young forests with low stem density.
- The authors believe that fox squirrels have adapted to more resource-limited environments than have gray squirrels. Adaptations developed to exploit these environments may limit their ability to respond to sudden increases in a particular resource, such as shelter.
- If used, nest boxes should be place in forests <50 years old, high (>12m) in the canopy, and close to the forest edge.

Perkins, M.W., and L.M. Conner. 2004. Habitat use of fox squirrels in southwestern Georgia. Journal of Wildlife Management 68(3): 509-513

- Fox squirrels populations are decreasing in southeastern United States due to habitat loss of mixed pine-hardwood forests.
- Fox squirrels selected mature pine and mixed pine-hardwood habitats when choosing a home range.
- Results suggest that management strategies should provide a combination of mature longleaf pine-hardwood and mature pine-hardwood habitats to promote fox squirrel habitat.
- Uneven-aged forest management strategies can be supplemented with prescribed fire to support pine dominance while retaining a hardwood component.

Peterle, T.J., and W.R. Fouch. 1959. Exploitation of a fox squirrel population on a public shooting area. Michigan Department Conservation Report 2251. 4pp.

- Harvest from 1952 to 1958 was 0.68 squirrels/acre.
- Hunters recovered 67% of juvenile males, 59% of juvenile females, 46% of adult males, and 58% of the adult females.
- Hunters removed up to 60% of population without reducing the reproductive potential.
- 10 squirrel nest boxes/acre in a 10-acre oak-hickory woodlot failed to increase the squirrel population.
- 24,000-board foot cut in same 10-acre woodlot did not change the number of squirrels harvested.
- 3-year study of the mast production shows no relationship between mast crops and squirrel population.
- Public pressure on this public shooting area is 4 to 5 times greater than on private land.

Salsbury, C.M., R.W. Dolan, and E.B. Pentzer. 2004. The distribution of fox squirrel (*Sciurus niger*) leaf nests within forest fragments in central Indiana. The American Midland Naturalist 151(2):369-377.

- Squirrels preferred certain tree species over others to build leaf nests in. Preference was not the same at all 6 study sites.
- The presence of vines in the tree canopy and large trees are important factors influencing nest tree choice for fox squirrels.
- The higher leaf nest densities at the disturbed sites suggest that fox squirrels prefer to nest in woodlots with a dense shrub layer, and may not negatively affected by habitat disturbance and fragmentation of urbanization.

Salsbury, C.M. 2008. Distribution patterns of *Sciurus niger* (Eastern Fox Squirrel) leaf nests within woodlots across a suburban/urban landscape. Northeastern Naturalist 15(4):485-496.

- 8.0% of leaf nests were found in trees with at least one other nest.
- Nest density was negatively related to woodlot area.
- The leaf-nest density within woodlots was not influenced by woodlots size, approximate age, shape, or degree of isolation.
- Leaf nests were not more likely to be located near the woodlot edge than in the woodlot interior.
- Forest fragmentation does not appear to negatively affect abundance of fox squirrels as shown by leaf-nest density.

Sanderson, H. R. 1975. Den-tree management for gray squirrels. Wildlife Society Bulletin 3:125-131.

- Den requirements and formation are discussed.
- A mixture of tree species will decay and develop den cavities at varying rates.
- Clear-cut management options to supply tree dens are discussed.
- Option A: Retention of existing den trees. In the regeneration cut, leave 2 den trees per 5 acres, and keep all known den trees in intermediate treatment. This option has the lowest management intensity and highest risk of not attaining the mast or timber yield.
- Option B: Retention of existing and potential den trees. Follow Option A in the regeneration cut, or leave no den trees. For intermediate treatments, retain all known den trees, and thin to keep a mixture of tree species for mast. This option has low management intensity and high risk of not achieving the squirrel management goal.
- Option C: Treatment of selected trees to form dens. In the regeneration cut, follow options A and B; as intermediate treatments, follow Option B and treat selected trees to

form dens. This option provides medium management intensity and moderate risk of not attaining the squirrel management goal.

- Option D: Provision of artificial dens. The regeneration cut is the same as for all other options; intermediate treatments include those of Option B plus providing artificial dens. High management intensity and low risk of not attaining the squirrel management goal.
- A minimum of 1 den/0.8 ha is required to maintain a density of 1 squirrel/1.6 ha.

Smith, C.C., and D. Follmer. 1972. Food preferences of squirrels. Ecology 53(1):82-91.

- Gray and fox squirrels show similar food preferences.
- Squirrels' preferences are based on speed at which they can ingest food energy and digestibility of the food eaten.
- Niche distinction between squirrel species probably related to differences in foraging behavior and predator escape behavior, not food preferences or feeding efficiency.
- Fox squirrels are adapted to open forest and forest edges and gray squirrels to dense forests.
- Activity patterns of gray and fox squirrels make hickory nuts and walnuts most efficient in fall and spring, and acorns most efficient in winter.
- A squirrel will have to ingest almost twice the weight of white oak or bur oak acorn kernels as hickory nut kernels in order to obtain the same daily energy requirements.
- Both fox and gray squirrels prefer hickory nuts and oak acorns to other food sources.

Thompson, D.C. 1978. Regulation of a northern grey squirrel (*Sciurus carolinensis*) population. Ecology 59:708-715.

- 70.7% of females gave birth during each breeding period.
- 48.8% of females weaned a litter averaging 3.1 young.
- 51.9% of adult females weaned litters averaging 3.2 young, and 42.9% of the yearlings weaned an average of 2.8 young.
- 11% of spring-born animals emigrated each fall.
- The sex ratio was 1:1.
- Dispersal losses were estimated at 10.4% of young.
- Adult annual mortality was 54%
- Spring-born and summer-born juveniles had similar annual mortality (63-66%).
- The dispersal of surplus young squirrels is considered a major factor in the determination of local population size.

WILD TURKEY FOOD HABITS ON THE NORTHERN FRINGE OF THEIR RANGE IN MINNESOTA

Eric M. Dunton, Darren Mayers, John Fieberg, and Kurt J. Haroldson

SUMMARY OF FINDINGS

The purpose of this study was to evaluate diet selection and body condition of eastern wild turkeys (*Meleagris gallopavo silvestris)* in agricultural and forested areas on the northern fringe of their range in Minnesota. We collected 15 turkeys in forested habitat (7 in 2009 and 8 in 2010) and 55 turkeys in agricultural habitat (24 in 2009 and 31 in 2010). Diets of turkeys consisted of a mixture of high energy (acorns) and low energy (grass, leaf litter, and sensitive fern) food items in forested habitat, while diets of turkeys located in agricultural habitats consisted primarily of high energy (corn) food items. In 2009, adult females in forested habitat had 32% less body weight, 72% less body fat, and were assigned to lower body condition classes than adult females in agricultural habitat. In 2010, adult females in forested habitat had 24% less body weight, 49% less body fat, and most birds were assigned to a lower body condition class than adult females in agricultural habitat. Further range expansion of wild turkeys in Minnesota's northern forests may be limited by availability of high energy food sources during winter, which are generally associated with agricultural practices.

INTRODUCTION

The current range of the eastern wild turkey extends far north of what was identified by Schorger (1966) as their historical range. This northern expansion has been associated with increased availability of food during winter (Wunz 1992, Wunz and Pack 1992, Kubisiak et al. 2001), which was considered limiting prior to settlement by European farmers. Wild turkey range in Minnesota and throughout the northeastern United States and southeastern Canada is currently expanding northward beyond agricultural areas (Kimmel and Krueger 2007). It is unknown how far turkeys will expand outside of mixed forest-agriculture areas into northern forest areas, and what their diet will include. Understanding winter diet selection of turkeys on the northern periphery of their range and the interaction of agriculture, snow conditions, and food habits will provide managers with improved information on wild turkey management needs outside of an agriculturally dominated landscape.

The eastern wild turkey is a food generalist with a winter diet ranging from >20 species (Korschgen 1967) to a restricted diet of only corn (Porter et al. 1980). As wild turkey range expanded north through mixed forest-agricultural habitats, Porter (2007) concluded, "Looking back at the field studies of the 1970s, it is clear that they were telling us more than we realized: snow and cold are not the issue, the key is food." Survival of wild turkeys in northern habitats was enhanced for birds with access to agricultural foods (Porter et al. 1980, Vander Haegen et al. 1989, Kane et al. 2007, Restani et al. 2009), but information is lacking on turkey food habits in northern non-agricultural areas. In central Minnesota, Restani et al. (2009) demonstrated that wild turkeys that had access to food plots had higher survival than turkeys without access.

Our objectives in this study were to: (1) identify winter foods used by wild turkeys on the northern fringe of their range in Minnesota; (2) describe diet as a function of agriculture and snow condition; and (3) compare body condition of wild turkeys with access to high-energy agricultural diets to those without.

STUDY AREA

We conducted this study within the Western Superior Uplands and Northern Minnesota Drift and Lake Plain Ecological Sections of the Laurentian Mixed Forest Ecological Province (MNDNR 2003). The study area is located north of Minnesota's historical wild turkey range (Leopold 1931, Schorger 1966, Snyders 2009) where wild turkey populations were established by translocation during the 1990s - 2008. The 25,959 km^2 study area is comprised of 35% upland deciduous forest, 31% crop/grass, 16% aquatic environment, 10% shrubland, 4% upland conifer forest, 2% lowland conifer forest, 2% lowland deciduous forest, and 1% non-vegetated (GAP Analysis Program MNDNR 2008).

METHODS

Using fixed wing aircraft, we located wintering flocks of turkeys in agricultural areas. To aid in locating wild turkeys with access to agricultural foods, we stratified the study area using a 500 ha grid and classified each cell to 1 of 3 habitat categories based on reclassified GAP land cover data: agricultural cells contained ≥30% cropland and ≥20% forested habitat; forested cells contained ≥50% forested habitat and 0% cropland; and other cells contained all other combinations of habitats. We used real-time, moving-map software (MNDNR 2005) coupled to a global positioning system receiver and a tablet-style computer to guide transect navigation and record turkey locations and aircraft flight paths directly to a geographic information system (Haroldson 2007). Turkeys located from aircraft were then relocated on the ground within 1-3 days. We attempted to collect by shooting 1-5 turkeys from each flock in late afternoon or early evening, when their crops were most likely to contain food (Hillerman et al. 1953). We did not conduct aerial surveys of forested strata in 2010 because no turkeys were observed in 122 surveys of forested strata during 2009. Instead, forested turkeys were located by soliciting observations from MNDNR employees and private landowners, and searching areas where turkey flocks were observed before winter.

At each collection site, we recorded date, snow depth, snow condition (e.g., crusted versus powder snow), habitat class (agricultural versus forested), and geographic coordinates. We verified habitat class by plotting collection sites on Farm Service Agency 2008 aerial imagery, and identifying presence or absence of cropland within a 1,545-m radius buffer (based on the 750 ha winter home range of wild turkeys in Minnesota reported by McMahon and Johnson [1980]). Habitat was classified as forested if no cropland was located within the buffer; otherwise habitat was classified as agricultural.

We determined frequency of occurrence and weight of food items present in the crops and gizzards according to the methods of Korschgen (1967). We determined dry matter content of foods by drying to a constant weight at 50°C (Decker et al. 1991). We assigned each food item 1 of 4 classes (high, medium, low, and unknown) based on estimated energy content of individual food items (Decker et al. 1991).

We evaluated body condition of wild turkeys collected in forested and agricultural habitats based on relative body weight and 3 estimates of body fat. We estimated total body fat of adult hens using a formula from Pekins (2007). We also assigned turkeys to 1 of 4 body condition classes based on amount and color of visible fat (Carter 1970). Finally, we assigned turkeys to 1 of 3 classes based on the amount of fat visible on the gizzard. We tested for differences in estimated weights and in the distribution of body condition classes among birds collected in agricultural and forested habitats using permutation tests that controlled for age, sex, and year. Similarly, we tested for a difference in mean estimated body fat for adult females in agricultural and forested habitats using a permutation test that controlled for year. Lastly, we calculated mean weights by age, sex, and year and 95% confidence intervals for these means. Tests were conducted using the independence-test function in the coin package (Hothorn et al. 2006) in the R programming language (R Development Core Team 2008).

RESULTS

In 2009, we aerially surveyed 122 forested strata and 103 agricultural strata and located 0 turkeys in forested strata and 1,130 turkeys (mean flock size = 23) in agricultural strata. In 2010, we aerially surveyed 50 agricultural grids and observed 289 turkeys (mean flock size = 14). Over the 2 year study period we collected 70 turkeys, including 15 from forested habitat (7 in 2009 and 8 in 2010) and 55 from agricultural habitat (24 in 2009 and 31 in 2010). Mean collection dates were similar between years for turkeys collected in forested habitat (24 January 2009 versus 31 January 2010) and agricultural habitat (30 January 2009 versus 13 January 2010). The geographic distribution of collected birds was similar between years, but included a larger proportion of the study area in 2010 (Figure 1). Snow depth in forested habitat averaged 39 cm in 2009 and 24 cm in 2010, and we classified snow conditions as crusted at 6 sites (0 in 2009 and 6 in 2010) and powder at 9 sites (7 in 2009 and 2 in 2010). Snow depth in agricultural habitat averaged 27 cm in 2009 and 22 cm in 2010, and we classified snow conditions as crusted at 29 sites (14 in 2009 and 15 in 2010) and powder at 26 sites (10 in 2009 and 16 in 2010).

High energy food (e.g., acorn [*Quercus spp.*]) was found in 86% and 63% of the crops from forest-habitat turkeys but formed only 47% and 25% of the crop contents by weight in 2009 and 2010, respectively (Table 1). For agricultural habitat turkeys, high energy foods (e.g., corn [*Zea mays*]) was found in 92% and 77% of the crops and formed 86% and 62% of the crop contents by weight in 2009 and 2010, respectively (Table 2).

In both years, wild turkeys from forested habitats were generally in poorer condition than birds from agricultural habitats. In 2009, adult females in forested habitat had 32% less body weight, 72% less body fat, and were assigned to lower body condition classes than adult females in agricultural habitat (Table 3, Figure 2). In 2010, adult females in forested habitat had 24% less body weight, 49% less body fat, and most birds were assigned to a lower body condition class than adult females in agricultural habitat (Table 3, Figure 2). Based on body condition we classified 4 forested turkeys as thin (3 in 2009 and 1 in 2010), 9 lean (4 in 2009 and 5 in 2010), 1 fat (2010), and 1 very fat (2010). In agricultural habitat we classified 1 thin (2010), 19 lean (5 in 2009 and 14 in 2010), 28 fat (12 in 2009 and 16 in 2010), and 7 very fat (2009). Body fat estimates were slightly higher in forested and agricultural strata in 2010 compared to 2009, but year effects were larger for forested turkeys (Figure 3). Observed differences in mean weights, estimated body fats (adult females only), and body condition classes were statistically significant (alpha=0.05 level). We classified gizzard fat for 8 forested turkeys as no fat (6 in 2009 and 2 in 2010), 3 fat (1 in 2009 and 2 in 2010), and 4 very fat (0 in 2009 and 4 in 2010. In agricultural habitat we classified gizzard fat for 1 turkey as no fat (2010), 19 fat (10 in 2009 and 8 in 2010), and 36 very fat (14 in 2009 and 22 in 2010).

DISCUSSION

Although the wild turkeys that we collected in this study consumed a wide variety of foods, birds collected in agricultural habitats consumed a larger amount of corn and other high energy foods than birds collected in forested habitats. The habitat-specific difference in consumption of high energy foods would likely have been greater if 3 of the 8 turkeys (2 adult females that weighed 3.28 and 3.81 kg and 1 adult male 8.913 kg) collected in forested habitat did not have access to residential bird feeders containing high energy food. The opportunistic feeding behavior of wild turkeys has long been known (Porter et al. 1980, 1983; Vander Haegen et al. 1989, Healy 1992). Because the distribution and abundance of bird feeders and other anthropogenic food sources is unknown, we consider them unreliable as a management strategy for maintaining turkey populations, particularly on the fringe of their current range.

Body weights of adult and juvenile hens collected in forested habitats in this study were below average whereas body weights of adult and juvenile hens collected in agricultural habitats were within the average range reported by Porter (1980) in Minnesota, Vander Haegen et al (1989) in Massachusetts, and Coup and Pekins (1999) in New Hampshire. Pekins (2007) suggested that adult hens weighing <3.0 kg have minimal body fat and were approaching a critical threshold of malnutrition. Thus, most adult hens collected from forested habitats in this study were showing signs of food deprivation. As supporting evidence, in 2009 we frequently observed turkeys in forested habitats remaining in their roosts late in the morning. This behavior was only occasionally observed in 2010 and is generally considered an indication of stress (Hayden and Nelson 1963).

Findings from this study indicate that turkeys in agricultural areas were able to find sufficient food (primarily corn) to maintain energy balance and fat reserves throughout the winter, even in 2009 when snow depth was >25 cm. In contrast, turkeys using exclusively forested habitats in deep snow were in poor body condition with little to no fat reserves. Even in 2010, when mean snow depth at collection sites was 24 cm, birds were in poor condition compared to their counterparts in agricultural habitat. Powder snow >15-20 cm hinders mobility, and >30 cm can prevent movement of wild turkeys (Austin and DeGraff 1975, Porter 1977, Healy 1992). Deep persistent snow cover can ultimately result in starvation. Wild turkeys began starving when snow depth was >30cm for >2 weeks in Pennsylvania (Wunz and Hayden 1975), 49 days in Wisconsin (Wright et al. 1996), and 40-59 days in New York (Roberts et al. 1995). Wright et al. (1996) documented starvation when deep snow restricted movements even though food was available within 0.8 km.

Further range expansion of wild turkeys in Minnesota's northern forests may be limited by availability of high energy food sources during winter, which are generally associated with agricultural practices. Wild turkey range may expand during periods with consecutive mild winters and then contract during severe winters. Because opportunities for agriculture are limited in this region, unharvested crops, stored crops, and livestock feeding operations may attract large concentrations of wintering turkeys, resulting in depredation complaints. Our inability to detect any wild turkeys during aerial surveys of 122 forested strata during 2009 leads us to suspect that some wild turkeys in forested areas moved to agricultural habitats to take advantage of high energy food sources.

ACKNOWLEDGEMENTS

Funding for this project was provided by MNDNR with contributions from Minnesota State Chapter of the National Wild Turkey Federation. J. Ream assisted with data collection in 2009. R. Wright provided GIS support. MNDNR wildlife, parks, and forestry staff and Camp Ripley staff located several forested turkeys and assisted with data collection. We thank private landowners in the study area for allowing access to their lands. T. Buker, R. Geving, and J. Jensen piloted survey aircraft. B. Haroldson provided assistance with survey software. R. Schindle and W. Smith assisted with food item identification. P. Pekins commented on the study design and R. Kimmel reviewed an earlier draft of this report.

LITERATURE CITED

Austin, D. E., and L. W. DeGraff. 1975. Winter survival of wild turkeys in the southern Adirondacks. Proceedings of the National Wild Turkey Symposium 3:55-60.

Carter, A. V. 1970. Pheasant body condition as related to winter weather factors. Progress report. South Dakota Department of Game, Fish, and Parks.

Coup, R. N., and P. J. Pekins. 1999. Field metabolic rate of wild turkeys in winter. Canadian Journal of Zoology 77:1075-1082.

Decker, S. R., P. J. Pekins, and W. M. Mautz. 1991. Nutritional evaluation of winter foods of wild turkeys. Canadian Journal of Zoology 69:2128-2132.

GAP Analysis Program MNDNR. 2008. Minnesota Department of Natural Resources Data Deli Homepage. < http://deli.dnr.state.mn.us/>. Accessed January 2008.

Haroldson, B. S. 2007. Estimating white-tailed deer abundance using aerial quadrat surveys.*In* M. W. DonCarlos, R. Kimmel, M. Lenarz, and J. Lawrence, editors. Summaries of Wildlife Research Findings, 2007. Minnesota Department of Natural Resources, St. Paul.

Hayden, A.H., and E. Nelson. 1963. The effects of starvation and limited rations on reproduction of game-farm wild turkeys. Transactions of the Northeast Section of The Wildlife Society 20:1-11.

Healy, W. M. 1992. Behavior. Pages 46-65 *in* J. G. Dickson, editor. The wild turkey: biology and management. Stackpole Books, Harrisburg, Pennsylvania, USA.
Hillerman, J. P., F. H. Kratzer, and W. O. Wilson. 1953. Food passage through chickens and turkeys and some regulatory factors. Journal of Poultry Science 32:332-335.
Hothorn, T., K. Hornik, M. A. Van de Wiel, and A. Zeileis. 2006. A lego system for conditional inference. The American Statistician 60:257-263.
Kane, D. F., R. O. Kimmel, W. E. Faber. 2007. Winter survival of wild turkeys in central Minnesota. Journal of Wildlife Management 71:1800-1807.
Kimmel, R. O., and W. J. Krueger. 2007. Northern wild turkeys: issues or opportunity, Proceedings of the National Wild Turkey Symposium 9:263-272.
Korschgen, L. J. 1967. Feeding habits and food. Pages 137-198 *in* O.H. Hewitt, ed., The wild turkey and its management. Washington, DC: The Wildlife Society. 589 pp.
Kubisiak, J., N. Paisley, and R. Wright. 1991. Wisconsin 1991 wild turkey studies report: research update. Proceedings Midwest Deer and Wild Turkey Study Group. Des Moines: Iowa Department of Natural Resources 174pp.
Leopold, A. 1931. Game Survey of the north central states. American Game Association. 189-193pp.
McMahon G. L. and R. N. Johnson. 1980. Introduction of the wild turkey into the Carlos Avery Wildlife Management Area. Proceedings of the National Wild Turkey Symposium 4:32–44.
MNDNR. 2003. Field Guide to the Native Plant Communities of Minnesota: the Laurentian Mixed Forest Province. Ecological Land Classification Program, Minnesota County Biological Survey, and Natural Heritage and Nongame Research Program. MNDNR St. Paul, MN.
MNDNR. 2005. DNR Survey: an Arcview GIS 3.x extension for enhancing aerial surveys for wildlife.http://www.dnr.state.mn.us/mis/gis/tools/arcview/extensions/dnrsurvey/dnrsurvey.html.
Pekins, P. J. 2007. Winter bioenergetics of eastern wild turkeys: understanding energy balance and survival in northern populations, Proceedings of the National Wild Turkey Symposium 9:273-280.
Porter, W. F. 1977. Home range dynamics of wild turkeys in southeastern Minnesota. Journal of Wildlife Management 41:434-437.
Porter, W. F., R. D. Tangen, G. C. Nelson, and D. A. Hamilton. 1980. Effects of corn food plots on wild turkeys in the upper Mississippi Valley. Journal of Wildlife Management 44:456-462.
Porter, W. F., G. C. Nelson, and K. Mattson. 1983. Effects of winter condition on reproduction in a northern wild turkey populations. Journal of Wildlife Management 47:281-290.
Porter, W. F. 2007. Understanding the ecology of wild turkeys in northern latitudes, Proceedings of the National Wild Turkey Symposium 9:307-313.
R Development Core Team. 2008. R: A language and environment for statistical computing. R Foundation for Statistical Computing, Vienna, Austria. ISBN 3-900051-07-0, URL http://www.R-project.org.
Restani, M., R. O. Kimmel, J. Fieberg, and S. Goetz. 2009. Effects of supplemental food and experience on winter survival of transplanted wild turkeys. The Wilson Journal of Ornithology 121:366-377.
Roberts, S. D., J. M. Coffey, and W. F. Porter. 1995. Survival and reproduction of female wild turkeys in New York. Journal of Wildlife Management 59:437-447.
Schorger, A. W. 1966. The wild turkey: its history and domestication. Norman: University of Oklahoma Press. 625pp.
Snyders, J. R. 2009. History and range expansion of wild turkeys (*Meleagris gallopavo*) in Minnesota. Thesis, Minnesota State University, Mankato, Minnesota, USA.
Vander Haegen, W. M., M. W. Sayre, and W. E. Dodge. 1989. Winter use of agricultural habitats by wild turkeys in Massachusetts. Journal of Wildlife Management 53:30-33.

Wright, R. G., R. N. Paisely, and J. F. Kubisiak. 1996. Survival of wild turkey hens in southwestern Wisconsin. Journal of Wildlife Management 60:313-320.
Wunz, G. A., and A. H. Hayden. 1975. Winter mortality and supplemental feeding of turkeys in Pennsylvania. Proceedings of the National Wild Turkey Symposium 3:61-69.
Wunz, G. A. 1992. Wild turkeys outside their historic range. Pages 361-384 *in* J. G. Dickson, editor. The wild turkey: biology and management. Stackpole, Books, Harrisburg, Pennsylvania, USA.
Wunz, G. A, and J. C. Pack. 1992. Eastern turkey in eastern oak-hickory and northern hardwood forest. Pages 232-264 *in* J. G. Dickson, editor. The wild turkey: biology and management. Stackpole, Books, Harrisburg, Pennsylvania, USA.

Table 1. Crop contents and estimated energetic value of food items for 8 wild turkeys collected in forested habitats on the northern fringe of their range in Minnesota during winter, 2010.

	Weight			
Food item	Total (g)	% of total	Frequency (%)	Estimated energetic value of food item
Crab apple (*Malus spp*.)	51.5	36	12.5	Medium
Acorn (*Quercus spp*.)	35.64	24.9	62.5	High
Grass stem and leaves (*Poa spp*.)	18.02	12.6	87.5	Low
Unknown forb seed	10.18	7.1	25	High
Smartweed seed (*Polygunum spp*.)	7.79	5.4	25	High
Bittersweet plant and seed (*Celastrus sp.*)	7.29	5.1	12.5	Medium
Sensitive fern frond (*Onoclea sensibilis*)	6.78	4.7	12.5	Low
Oat seed (*Avena sativa*)	2.03	1.4	25	High
Brome grass seed (*Bromus inermis*)	1.7	1.2	25	High
Curly dock plant and seed (*Rumex crispus*)	0.42	0.3	12.5	Medium
Poison ivy fruit (*Toxicodendron spp.*)	0.22	0.2	12.5	Medium
Soybean (*Glycine spp*.)	0.22	0.2	12.5	High
Sunflower seed (*Helianthus spp*.)	0.09	0.1	12.5	High
White cockle plant and seed (*Silene latifolia)*	0.14	0.1	12.5	Medium
Hazel catkin (*Corylus spp*.)	0.93	TR[a]	50	Low
Beetle (*Coleoptera spp.*)	0.01	TR[a]	12.5	Unknown
Club moss (*Lycopodium spp*.)	0.01	TR[a]	12.5	Low
Total	142.97			

[a] Trace (TR) amount of food item present in diet < 0.1

Table 2. Crop contents and estimated energetic value of food items for 31 wild turkeys collected in agricultural habitats on the northern fringe of their range in Minnesota during winter, 2010.

Food item	Weight		Frequency (%)	Estimated energetic value of food item
	Total (g)	% of total		
Corn kernel (*Zea mays*)	1038.77	61.9	77.4	High
Acorn (*Quercus spp*.)	213.16	12.7	12.9	High
Sunflower seed (*Helianthus spp*.)	78.83	4.7	29	High
Soybean (*Glycine spp*.)	76.12	4.5	12.9	High
Ash samara (*Fraxinus spp*.)	33.54	2	22.6	Low
Soybean plant parts (*Glycine spp*.)	33.08	2	9.7	Low
Buckthorn fruit (*Rhamnus cathartica*)	30.25	1.8	6.5	Medium
Crab apple (*Malus spp*.)	26.09	1.6	6.5	Low
Grass plant parts (*Poa spp*.)	25.71	1.5	77.4	Low
Manure	21.67	1.3	22.6	Unknown
Oat seed (*Avena sativa*)	19.54	1.2	16.1	High
Millet seed (*Panicum spp*.)	14.85	0.9	9.7	High
Black cherry fruit (*Prunus serotina*)	13.07	0.8	3.2	Medium
Corn plant parts (*Zea mays*)	13.54	0.8	22.6	Low
Brome grass seed (*Bromus inermis*)	11.64	0.7	9.7	High
Smartweed seed (*Polygunum spp*.)	5.9	0.4	6.5	High
Club moss (*Lycopodium spp*.)	5.04	0.3	16.1	Low
Unknown forb seed	3.86	0.2	12.9	High
Canada lettuce plant and seed (*Lactuca canadensis*)	2.8	0.2	6.5	Medium
Sweet cicely plant and seed (*Osmorhiza berteroi*)	2.93	0.2	9.7	Medium
Unknown legume seed	2.71	0.2	3.2	High
Grit	1	0.1	3.2	N/A
Basswood fruit (*Tilia americana*)	0.36	TR[a]	3.2	Medium
Curly dock plant and seed (*Rumex crispus*)	0.11	TR[a]	3.2	Medium
Hazel catkin (*Corylus spp*.)	0.04	TR[a]	3.2	Low
Beetle (*Coleoptera spp.*)	0.04	TR[a]	6.5	Unknown
Leaf litter	0.38	TR[a]	3.2	Low
Pine needle (*Pinus spp*.)	0.23	TR[a]	12.9	Low
Quack grass seed (*Elymus repens*)	0.69	TR[a]	3.2	High
Ragweed seed (*Ambrosia spp*.)	0.36	TR[a]	3.2	High
Red clover seed (*Trifolium spp*.)	0.25	TR[a]	3.2	High
Sensitive fern (*Onoclea sensibilis*)	0.35	TR[a]	6.5	Low
Thistle seed (*Cirsium spp*.)	0.43	TR[a]	6.5	High
Unknown forb seed	0.08	TR[a]	3.2	High
Wheat seed (*Triticum spp*.)	0.28	TR[a]	3.2	High
Total	1677.7			

[a] Trace (TR) amount of food item present in diet < 0.1

Table 3. Estimates of body fat for 70 wild turkeys collected in forested versus agricultural habitats on the northern fringe of their range in Minnesota during winter, 2009 - 2010.

						Body condition class				Mean estimated total body fat[e]	
Habitat	Year	Gender	Age	*n*	Mean weight (kg)	Very fat[a]	Fat[b]	Lean[c]	Thin[d]	Kg	%
Forest	2009	F	A	5	3.24	0	0	2	3	0.21	5.80
Forest	2009	F	J	1	3.81	0	0	1	0	-	-
Forest	2009	M	A	1	6.48	0	0	1	0	-	-
Forest	2009	M	J	0	0	0	0	0	0	-	-
Forest	2010	F	A	3	3.71	0	0	2	1	0.43	11.46
Forest	2010	F	J	0	0	0	0	0	0	-	-
Forest	2010	M	A	4	8.8	1	1	2	0	-	-
Forest	2010	M	J	1	3.57	0	0	1	0	-	-
Ag	2009	F	A	14	4.75	5	5	4	0	1.02	20.57
Ag	2009	F	J	3	3.91	0	2	1	0	-	-
Ag	2009	M	A	3	9.26	2	1	0	0	-	-
Ag	2009	M	J	4	6.41	0	4	0	0	-	-
Ag	2010	F	A	18	4.9	0	10	8	0	1.1	22.49
Ag	2010	F	J	2	4.83	0	1	1	0	-	-
Ag	2010	M	A	4	8.23	0	2	1	1	-	-
Ag	2010	M	J	7	5.87	0	3	4	0	-	-
Total				70		8	29	28	5		

[a] Large deposits of fat on mid-line of breast, thighs, back, around crop, at the posterior of the body cavity, and immediately beneath skin. Fat is bright yellow (Carter 1970).
[b] Large fat deposits on back and thighs and reduced deposits elsewhere. Fat may be orange in color. (Carter 1970).
[c] Fat deposits are completely resorbed. Breast muscle has "normal" contour. Dark orange color in cellular framework of resorbed fat deposits (Carter 1970).
[d] Breast muscle attains wedge-like appearance ("hatchet-breast"). Skin resembles parchment (Carter 1970).
[e] Body fat (g) = 571.3 x (kg body weight) – 1696; $R^2 = 0.59$, $P < 0.05$. Applies to adult females only (Pekins 2007)

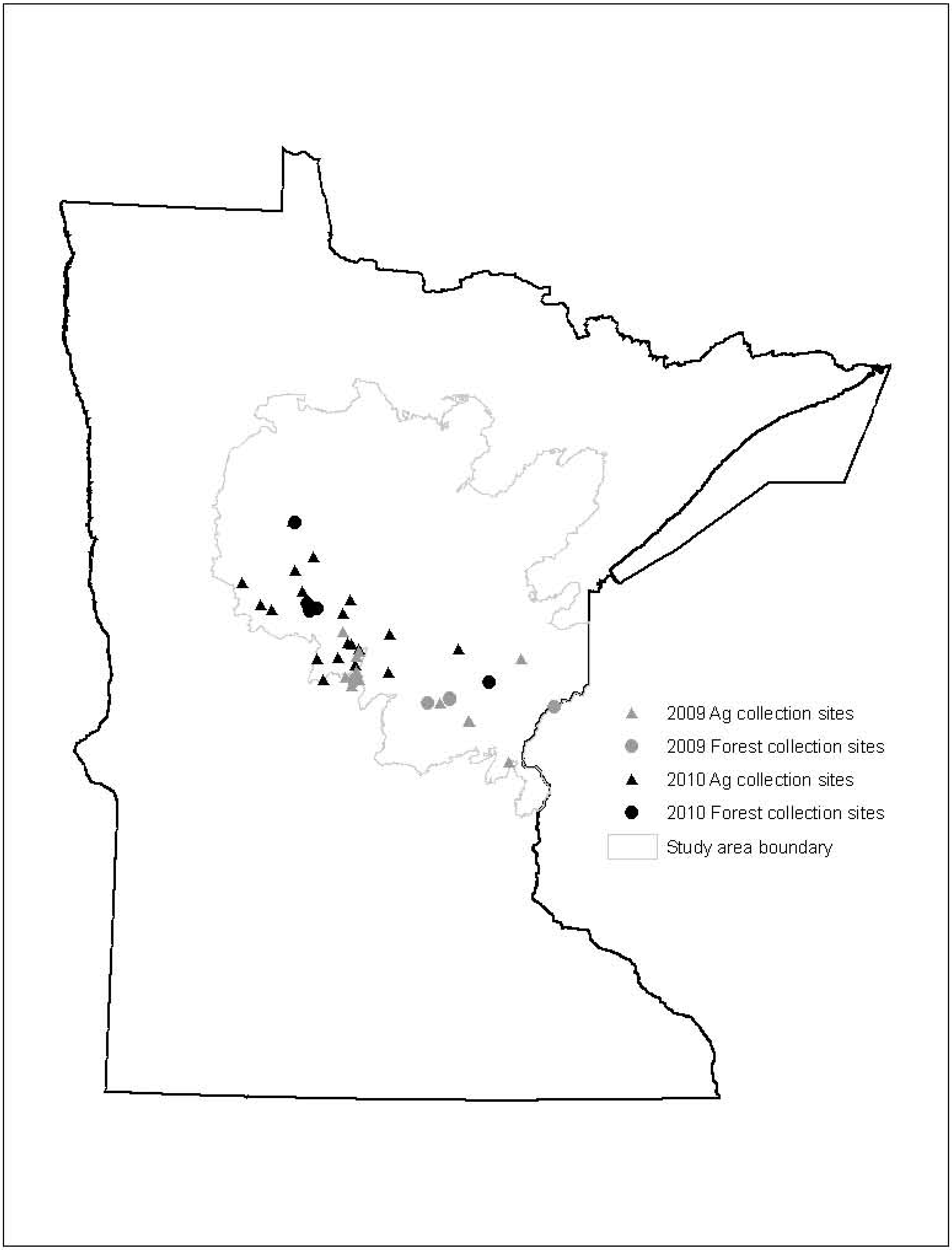


Figure 1. Study area location and collection sites for 15 wild turkeys collected in forested habitat and 55 turkeys collected in agricultural habitat as part of the winter food habits project on the northern fringe of wild turkey range in Minnesota, 2009 – 2010.

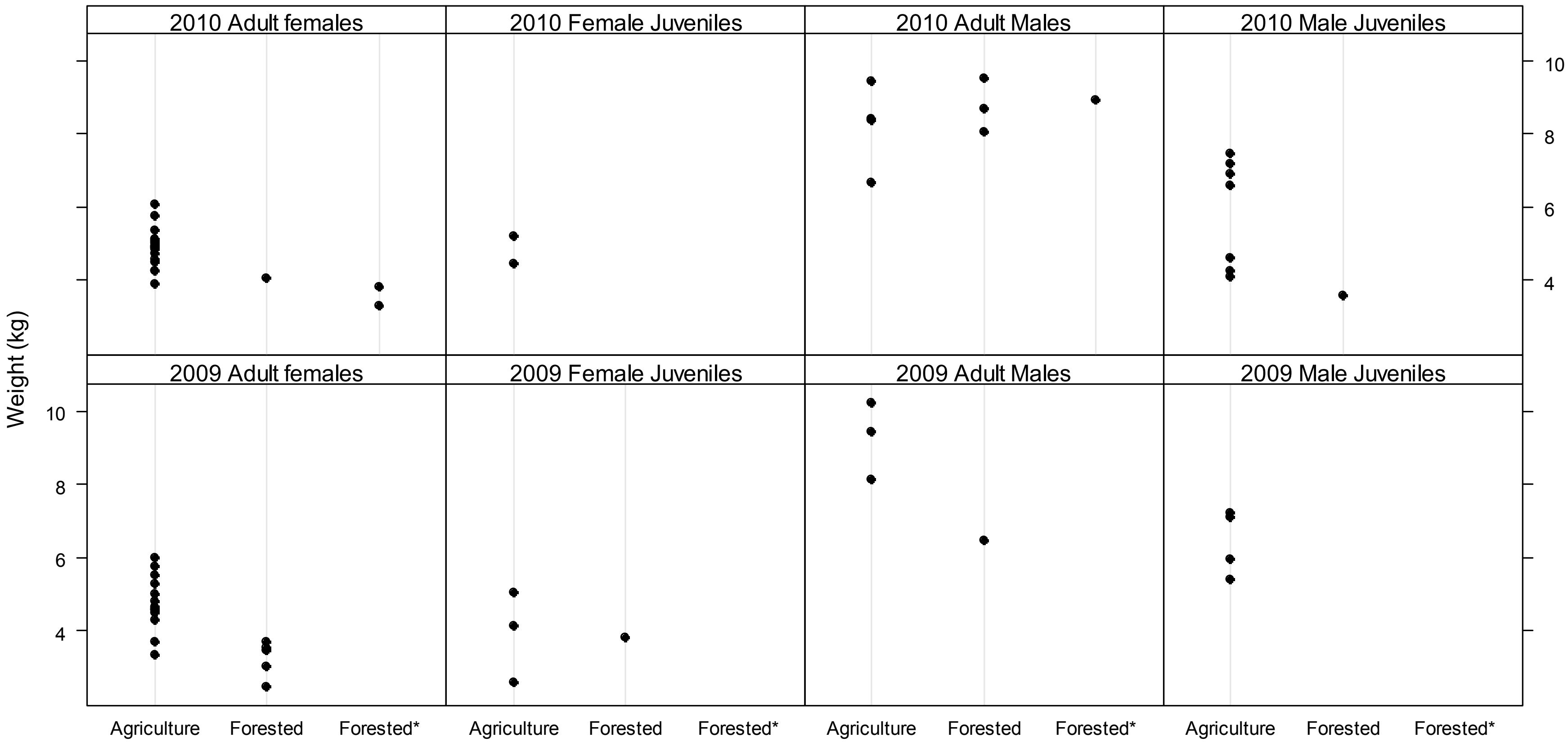


Figure 2. Body weight for 70 wild turkeys collected in forested habitat (*n* = 15) and agricultural habitat (*n* = 55) for winter food habits project on the northern fringe of wild turkey range in Minnesota, 2009 – 2010. Note: in 2010, 3 turkeys sampled in forested habitats had access to a high energy food (these birds are listed as “Forested*” in the above figure).

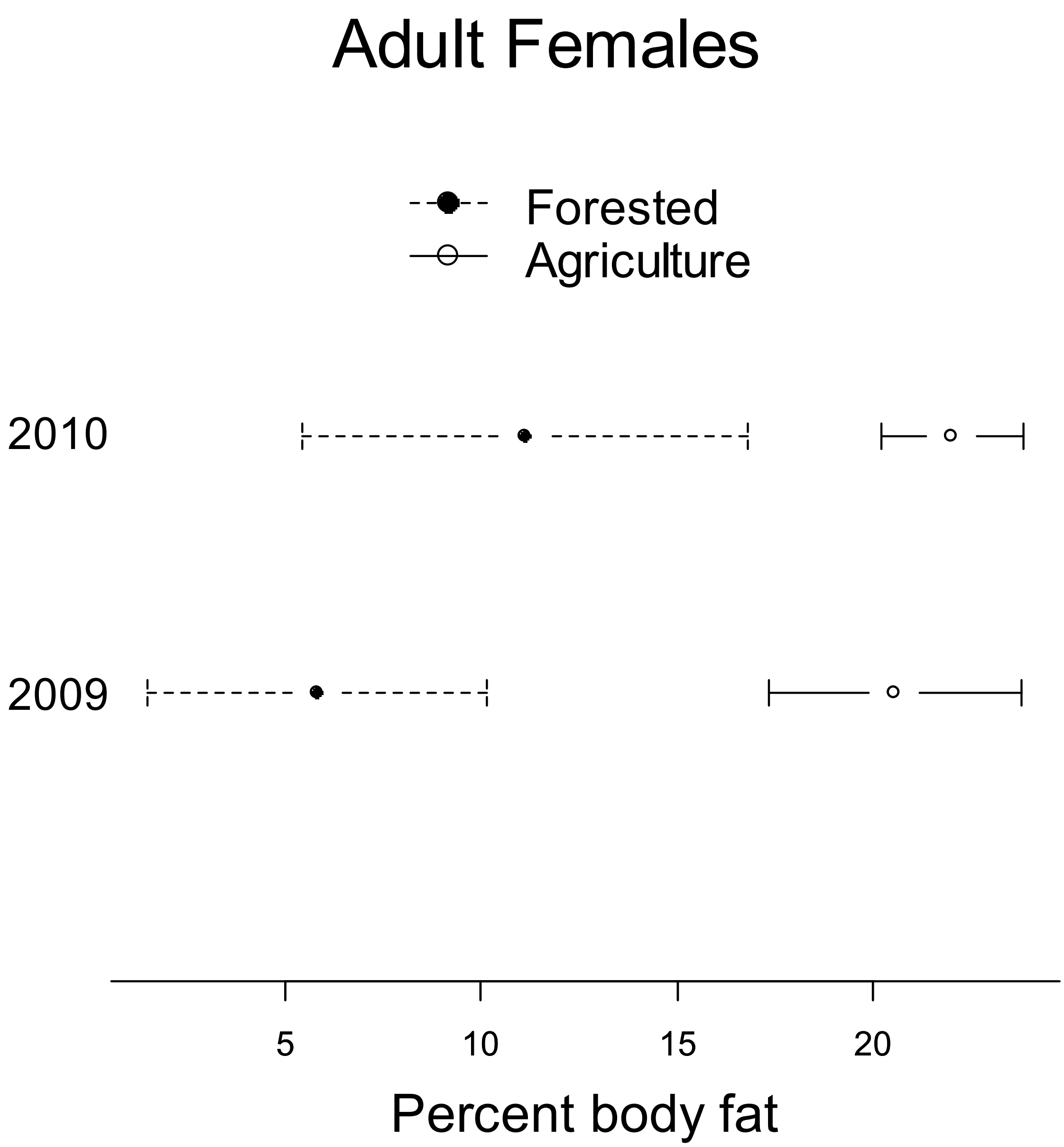


Figure 3. Mean estimated body fat and 95% confidence intervals for adult female wild turkeys in forested habitats (*n* = 8; 5, 2009 and 3, 2010) and agricultural habitats (*n* = 32; 14, 2009 and 18, 2010) collected as part of the winter wild turkey food habits project on the northern fringe of wild turkey range, Minnesota. Note: in 2010, 2 adult females sampled in forested habitats had access to a high energy food.

MOVEMENTS, HABITAT SELECTION, ASSOCIATIONS, AND SURVIVAL OF GIANT CANADA GOOSE BROODS IN CENTRAL TENNESSEE[1]

Eric M. Dunton and Daniel L. Combs

ABSTRACT

The brood-rearing period in giant Canada geese (*Branta canadensis maxima*) is one of the least-studied areas of goose ecology. We monitored 32 broods in Putnam County, Tennessee, from the time of hatching through fledging (i.e., when the goslings gained the ability to fly) and from fledging until broods left the brood-rearing areas during the spring and summer of 2003. We conducted a fixedkernel, home-range analysis for each brood using the Animal Movement Extension in ArcView® 3.3 GIS (ESRI, Redlands, Calif.) software and calculated 95% and 50% utilization distributions (UD) for each brood. We classified 25 broods as sedentary (8 ha 95% UD), three as shifters (84 ha 95% UD), two as wanderers (110 ha 95%UD); two were unclassified because of low sample-size. We measured 5 habitat variables (i.e., percentage of water, percentage of pasture, percentage of development, number of ponds, and distance to nearest unused pond) within a 14.5-ha buffer at nesting locations. We used linear regression, using multi-model selection, information theoretic analysis, to determine which, if any, habitat variables influenced home-range size at a landscape level. The null model was the best information-theoretic model, and the global model was not significant, indicating that landscape level habitat variables selected in this study cannot be used to predict home-range size in the Upper Cumberland region goose flock. We analyzed associations between broods, using a coefficient of association of at least 0.50, and determined association areas by overlaying individual home ranges. Overall gosling survival ($\hat{S}$) during the brood-rearing period was 0.84 (95% CL = 0.78, 0.92), using a staggered-entry Kaplan-Meier survival curve. We believe that abundance of quality forage and pond habitat, high survivorship, and a lack of movement corridors (i.e., rivers, lakes, and reservoirs) were responsible for the relatively small home ranges of geese in the Upper Cumberland region. Associations formed during brood rearing may reduce predation risks and serve as a template for lifelong social bonds with family members and unrelated geese that are reared in the same locations.

[1]Human-Wildlife Interactions 4(2): Fall 2010 *in press*

BULLET FRAGMENTATION AND LEAD DEPOSITION IN WHITE-TAILED DEER AND DOMESTIC SHEEP[1]

Marrett Grund, Lou Cornicelli, Leah T. Carlson, and Erika Butler

ABSTRACT

In February 2008, a private physician in North Dakota radiographed hunter-harvested venison and found that 60 of 100 packages contained metal fragments. This discovery had implications for public-funded venison donation programs and prompted several Midwest states to examine their programs. Approximately 500,000 deer hunters harvest >200,000 deer annually in Minnesota and the state has a similar donation program as the program operated in North Dakota. Therefore, we analyzed fragmentation patterns and lead deposition in carcasses of 8 white-tailed deer (*Odocoileus virginianus*) and 72 domestic sheep (*Ovis aries*). Five different bullet types were fired from centerfire rifles, and we also fired projectiles from a shotgun and blackpowder muzzleloader. We then described fragmentation patterns and lead deposition among treatment groups. Centerfire bullets designed to expand quickly upon impacting the animal had bullet fragments and lead deposited throughout the entire abdominal cavity of carcasses. We also used 2 types of centerfire bullets that were purportedly designed to resist fragmentation. One of these bullet types had fragmentation patterns and lead deposition rates similar to the rapid expanding bullets, the other bullet type resisted fragmentation, and no lead was detected in muscle tissue that we sampled. We determined that a centerfire bullet made from copper resisted fragmentation, and of course did not deposit any lead in muscle tissues. Projectiles fired from the shotgun and blackpowder muzzleloader did deposit lead into carcasses but did not fragment as much as bullets fired from centerfire rifles. Our study suggests that rinsing the abdominal cavity may spread the lead contaminant to other areas of the carcass thereby worsening the contamination situation. We frame conclusions based on our interpretation of limited data but suggest hunters who use centerfire rifles and are concerned about lead exposure should purchase 1 of 2 bullet types that resist fragmentation.

[1]Human-Wildlife Interactions 4(2): Fall 2010 *in press*

EVALUATION OF EARLY ANTLERLESS SEASONS IN THE NORTH METRO AND IN NORTHWEST MINNESOTA

Marrett Grund and Lou Cornicelli

SUMMARY OF FINDINGS

We conducted a study to determine if including an early antlerless season into the deer management hunting framework would increase the harvest of antlerless deer and reduce deer densities in northwest Minnesota and in the north metro region. We monitored deer harvest data, hunter success rates, and conducted aerial surveys to determine if populations were reduced. Based on these trend indicators and surveys, we concluded that the early antlerless season was very effective at reducing deer numbers in the northwest region, but was ineffective at reducing deer numbers in the north metro region. Consequently, more aggressive management options need to be employed in the north metro region and additional research is warranted to explain these differences.

INTRODUCTION

Over the past 70 years, white-tailed deer (*Odocoileus virginianus*) management has changed from focusing on augmenting population growth through habitat protection, hunting regulations, and predator control, to concern about how best to limit deer densities and the consequent impacts of deer on society and forest ecosystems. In fact, managing overabundant deer has emerged as one of the most significant natural resource management challenges over the past 2 decades for many state wildlife agencies in the midwestern and eastern United States (Warren 1997).

The deer program currently used in Minnesota is based on a framework that was created in the early 1970s (Cornicelli 2009). Essentially, the seasonal framework allowed for an unlimited number of hunters to hunt each year, but allow the population to grow as well. This was accomplished by setting the annual bag limit at 1 and providing hunters with a license that allowed the harvest of antlered deer only. A limited number of antlerless permits were offered to hunters to harvest a prescribed quota of antlerless deer. This deer management system worked well throughout the 1970s and 1980s, when the management goal was to increase deer numbers throughout the state.

Modifications to the seasonal framework began in the 1990s and 2000s as the management goal shifted from population growth to attempting to stabilize or reduce deer numbers. The bag limit was increased so that hunters could harvest up to 5 antlerless deer. Survey data collected in 2004 suggested that >70% of hunters taking antlerless deer were only harvesting 1 deer per year. Consequently, additional changes in the bag limit would likely not be effective, because a low percentage of hunters were utilizing the maximum bag limit. Starting in 2005, an early antlerless season was added to 7 deer permit areas (DPAs) in northwest Minnesota and in the north metro region (a region including and around Minneapolis and St. Paul) to evaluate its potential to increase antlerless harvest and reduce deer densities. This study examines harvest data that occurred in these 7 DPAs from 2005-2009.

OBJECTIVES

1) Evaluate the early antlerless season by examining hunter harvest data;
2) Document trends in hunter success rates; and
3) Evaluate the early antlerless season by using population estimates derived from aerial surveys and population modeling.

STUDY AREAS

The northwestern Minnesota study area included 4 DPAs, which encompassed approximately 5,600km^2 (2,250mi^2). The study area in the north metro was 3,800km^2 (1,465mi^2). The northwest study area can be considered mostly flat terrain with a relatively low percentage of woody cover (13%). The north metro is a mosaic of woodlots which comprise 25% of the landscape. Winter severity indices (Lenarz 2009) are higher in the northwest than in the north metro (Grund 2001). However, buck harvest trends suggest that winters have been relatively mild along the transition zone of Minnesota since 1997 (Grund 2009, Lenarz 2009).

METHODS

Seasonal Framework

Minnesota Department of Natural Resources (MNDNR) offered an unlimited number of early antlerless licenses at 25% of the cost of a regular firearms license. The early antlerless license could only be used to harvest an antlerless deer during the early antlerless season. The early antlerless season occurred during the second weekend of October, which was typically 3 weeks before the regular firearms season. The bag limit during the early antlerless season was 2 antlerless deer. A hunter could harvest another 5 antlerless deer during other hunting seasons in these DPAs: the regular firearms season was a 9-day season that started on the Saturday closest to 6 November, the muzzleloader season was a 16-day season that began the Saturday after Thanksgiving, and the archery season began in mid-September and ended 31 December. In all seasons except for the early antlerless season, there were an unlimited number of hunting licenses available to hunters, which allowed hunters to take a deer of either sex. An unlimited number of antlerless-only licenses were available, but an individual hunter could only purchase up to 5 of these licenses. Only 1 antlered buck could be harvested per hunter during 1 hunting year, which encompassed all hunting seasons.

Harvest Data

Successful hunters were required to register deer at a designated registration station within 24 hours of the close of the hunting season. Hunters were required to report their deer as an adult male, adult female, fawn male, or fawn female. Registration station operators were not required to inspect deer or verify that registration information provided by the hunter was correct. We used the percentage of females in the harvest as an index to the harvest pressure on antlerless deer (Roseberry and Woolf 1991, Grund 2001) and we discuss numerical trends in harvest across years for each permit area from 2005-2009.

We measured hunter effort based on total hunter numbers. Hunters were asked by store clerks which DPA they intended to hunt when they purchased their hunting license. We assumed that effort per hunter was constant across years and changes in catch-per-unit-effort (i.e., hunter success rates) reflected changes in deer densities (Roseberry and Woolf 1991).

Population Monitoring and Modeling

Aerial surveys were conducted during the first year (winter 2006) and last year (winter 2010) of this project to assess population change over the 5 years. Methods for aerial surveys can be reviewed in Haroldson (2008).

Population modeling was conducted using an accounting-type model similar to POP2 (Bartholow 1986). The model we used was a management model that incorporated numeric harvest values of adult males, adult females, fawn males, and fawn females, and simulated non-hunting survival rates based on literature reviews and previous field studies (Grund 2001). Reproductive values were derived from fetus surveys conducted on car-killed female deer. We ran simulations with the entire harvest that occurred and we removed deer harvested during the early antlerless season to assess where each population might be in 2010 had the early antlerless season not been held for 5 consecutive years.

RESULTS AND DISCUSSION

Harvests

Excluding DPA 210, the average female harvest increased 30% in each northwestern DPA during the first year of the early antlerless season (Table 1). The boundaries for DPA 210 changed significantly in 2005, therefore, a comparison between years cannot be made for that DPA. The average antlered male harvest increased only 3% in each northwestern DPA during the first year of the early antlerless season. In contrast to the 30% increase in female harvest observed in northwestern study areas, the average female harvest only increased 7% in each north metro DPA during 2005 whereas the average buck harvest decreased by 5% the same year. The results from the first year suggested that the early antlerless season may be more effective in the northwestern DPAs than in the north metro DPAs due to the substantial increase in females harvested (30% vs. 7%).

The percentage of females in the harvest averaged almost 53% in the northwest DPAs from 2005-2009 (Table 2). In comparison, the percentage of females in the harvest was only 47% in the north metro DPAs during the same years. The average percentage of females in the harvest from 2003-2004 for the northwestern and north metro DPAs was 46% and 45%, respectively. Thus, the early season increased the percentage of females in the harvest by 7% whereas it only increased the percentage by 2% in the north metro.

The antlered male harvest (Table 1) and the total antlered male harvest (Figure 1) declined >12% over the 5 years in the northwest but was unchanged in the north metro (Table 1). However, hunter success rates on antlered males were stable in DPA 210, which may reflect a deer density that did not remarkably change over the course of 5 years (Table 3). Most likely, hunter success rates on antlered males will continue to decline in DPAs 256 and 257 as they have the prior 2 years if the early antlerless season were continued in those areas. Between year variability is apparent in hunter success in the north metro study areas, but there is not a clear downward trend indicating that success is worsening due to fewer deer being on the landscape.

Population Monitoring and Modeling

In northwest Minnesota, deer densities declined on average about 6% per year in each permit area based on aerial survey counts. Over the course of 5 years, each northwest deer population had an approximate 32% reduction in aerial counts (Figure 2). In contrast, deer counts in the north metro increased approximately 28% over the 5-year time period (Figure 3). However, it is noteworthy that the counts in DPA 236 were reduced >10% over the 5-year period (Figure 3). These results agree with the aforementioned harvest results, but suggest that 2 populations (DPAs 225 and 227) actually increased under these aggressive management strategies. Certainly, the study was continued too long in DPA 256 because the management goal was only to reduce deer densities by 33% in that DPA.

In northwest Minnesota, population modeling we conducted suggests deer densities declined each year under early antlerless seasons but would have increased under regular firearms seasons (Figure 4). Thus, the additive harvest was critical to managing the population according to its population objective. In the north metro study area; however, the harvest should be deemed additive (Grund 2007) but insufficient to meet its population management objective because the population was not reduced even with the early antlerless harvests (Figure 5). It is noteworthy that the population would have increased >50% in the north metro had EA seasons not been used over the 5-year season.

Management Implications

Harvest management strategies in the northwest study areas should become more conservative as many of those areas were brought below the management goal of 33%. However, harvest management strategies on antlerless deer in the north metro study areas should be liberalized so that survey and harvest data indicate that populations are being managed according to goal.

Preliminary results from the alternative deer management study indicate including an antler-point restriction regulation to the seasonal framework would increase the antlerless harvest by about 12% and that earn-a-buck regulations would increase the antlerless harvest by >50% during the first year they are implemented in Minnesota (Grund 2007). Preliminary modeling would suggest using earn-a-buck for 1 year then using antler-point restrictions as an approach to maintain high harvest rates on antlerless deer in the north metro study area.

ACKNOWLEDGEMENTS

We acknowledge all the wildlife managers that assisted us throughout this study, particularly R. Naplin, D. Pauly, T. Wolfe, D. Nelson, B. Lueth, and B. Welsh. This project would not happen without their continuous support and advice. We also appreciate the assistance from J. Loegering (UMN-Crookston) and W. Faber (Central Lakes College-Brainerd) and all the hundreds of students from the Universities of Minnesota-Twin Cities, University of Minnesota-Crookston, Minnesota State University-Mankato, St. Cloud State University, and Central Lakes College we were able to obtain to help us collect data for this project. Neither of us has seen so much data collected in order to make changes to a deer management program, their assistance was crucial to our success. We also thank M. DonCarlos and R. Kimmel for providing guidance throughout this project.

LITERATURE CITED

Bartholow, J. 1986. POP-II system documentation. Fossil Creek Software, Fort Collins, Colorado, USA.

Cornicelli, L. J. 2009. Integrating social considerations into managing white-tailed deer in Minnesota. Dissertation. University of Minnesota, Minnesota, Twin Cities, USA.

Grund, M. D. 2001. Options for monitoring white-tailed deer populations in Minnesota. Dissertation, Southern Illinois University, Carbondale, Illinois, USA.

Grund, M. D. 2009. Monitoring population trends of white-tailed deer in Minnesota's farmland/transition zone – 2009. pages 19 – 28 *in* Dexter, M. H. editor. Status of wildlife populations , fall 2009. Unpublished report, Division of Fish and Wildlife, Minnesota Department of Natural Resources, St. Paul, Minnesota, USA.

Grund, M. D. 2007. Evaluating the effects and early antlerless –only deer hunting season has on antlerless harvests. pages 36-42 *in* DonCarlos. M., W. R. O. Kimmel. J. S. Lawrence, and M. S. Lenarz. Summaries of Wildlife Research Findings – 2007. Unpublished Report, Division of Fish and Wildlife, Minnesota Department of Natural Resources, St. Paul, Minnesota, USA.

Haroldson, B. H. 2008. Estimating deer density using aerial quadrat surveys. Pages 1-7, *in* M. Doncarlos, R. O. Kimmel, J. S. Lawrence, and M. S. Lenarz, editors. Summaries of Wildlife Research Findings, 2008. Division of Fish and Wildlife, Minnesota Department of Natural Resources, St. Paul, Minnesota, USA.

Lenarz, M. S. 2009. Population trends of white-tailed deer in the forest zone – 2009. Pages 101-111 *in* M. H. Dexter, editor. Status of wildlife populations, fall 2009. Division of Fish and Wildlife, Minnesota Department of Natural Resources, St. Paul, Minnesota, USA.

Roseberry, J. L, and A. Woolf. 1991. A comparative evaluation of techniques for analyzing white-tailed deer harvest data. Wildlife Monographs 117.

Warren, R. J. 1997. The challenge of deer overabundance in the 21st century. Wildlife Society Bulletin 25:213-214.

Table 1. Harvest, by deer permit area and sex, of white-tailed deer in the north metro and northwestern Minnesota early antlerless season study areas, 2000-2009. The early antlerless season began in 2005.

	Northwest								Northmetro					
	209		210		256		257		225		227		236	
Year	M	F	M	F	M	F	M	F	M	F	M	F	M	F
2000	534	364	801	523	633	421	612	471	2077	1501	1347	799	1365	782
2001	634	421	905	589	748	721	729	728	2175	1780	1488	1006	1374	949
2002	639	435	922	666	642	514	607	499	2039	1539	1392	909	1364	1093
2003	720	611	1035	843	792	745	735	656	2422	2113	1611	1302	1513	1385
2004	795	611	966	862	752	720	729	607	2081	1576	1480	1119	1314	1153
2005	850	851	1571	1398	783	846	708	793	1943	1722	1310	1208	1348	1202
2006	834	959	1565	1558	820	1094	669	727	2229	2176	1502	1404	1242	1217
2007	870	979	1485	1556	842	909	633	726	2002	1886	1478	1467	1387	1275
2008	709	819	1466	1436	680	681	566	582	1773	1575	1261	1062	1174	1015
2009	778	808	1433	1365	598	668	521	629	2045	1728	1429	1249	1198	1032

Table 2. Percentage of female deer in total deer harvest for each deer permit area in the north metro and northwestern Minnesota early antlerless season study areas, 2000-2009.

	Northwest				Northmetro		
Year	209	210	256	257	225	227	236
2000	41	40	40	43	42	37	36
2001	40	39	49	50	45	40	41
2002	41	42	44	45	43	40	44
2003	46	45	48	47	47	45	48
2004	43	47	49	45	43	43	47
2005	50	47	52	53	47	48	47
2006	53	50	57	52	49	48	49
2007	53	51	52	53	49	50	48
2008	54	49	50	51	47	46	46
2009	51	49	53	55	46	47	46

Table 3. Hunter success rates on antlered male white-tailed deer for each deer permit area in the north metro and northwestern Minnesota early antlerless season study areas, 2005-2009.

	Northwest				Northmetro		
Year	209	210	256	257	225	227	236
2005	23	26	23	24	20	20	29
2006	26	28	25	28	25	26	28
2007	28	26	27	27	24	25	32
2008	23	26	22	23	20	23	28
2009	25	26	20	21	22	24	29

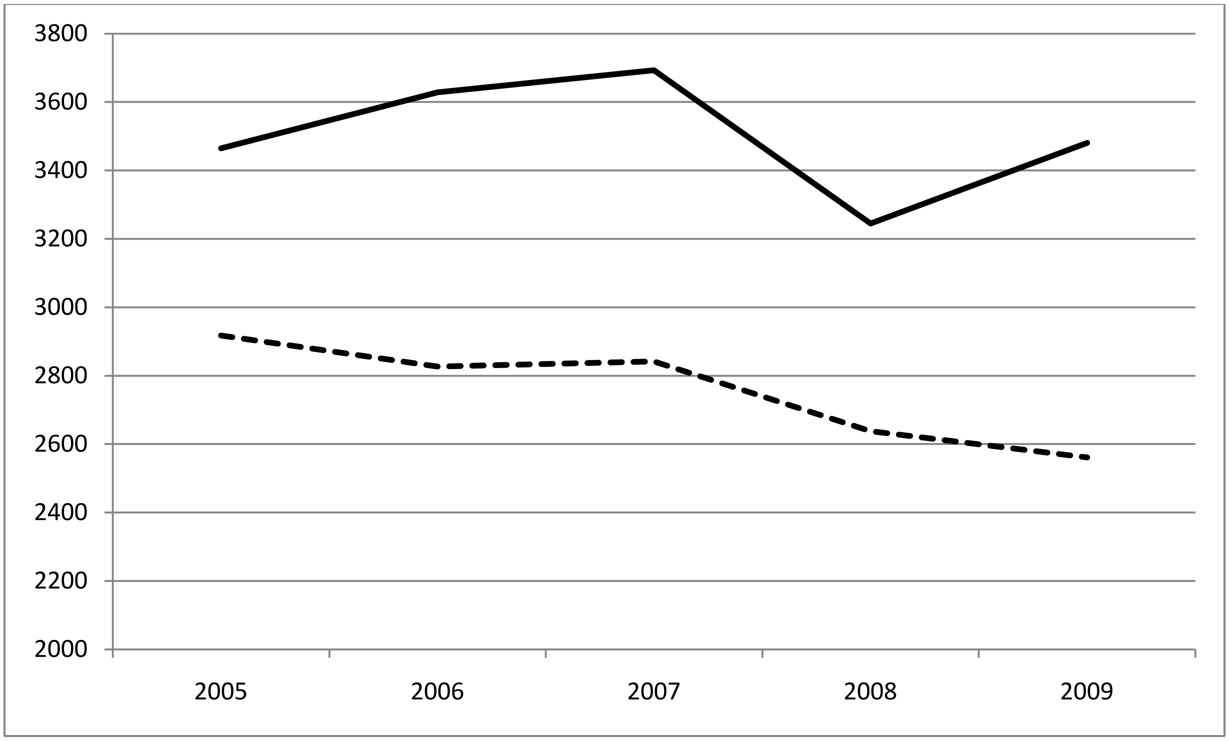


Figure 1. Number of antlered males harvested in early antlerless study areas in northwestern Minnesota (dashed line) and in the north metro (solid line), Minnesota.

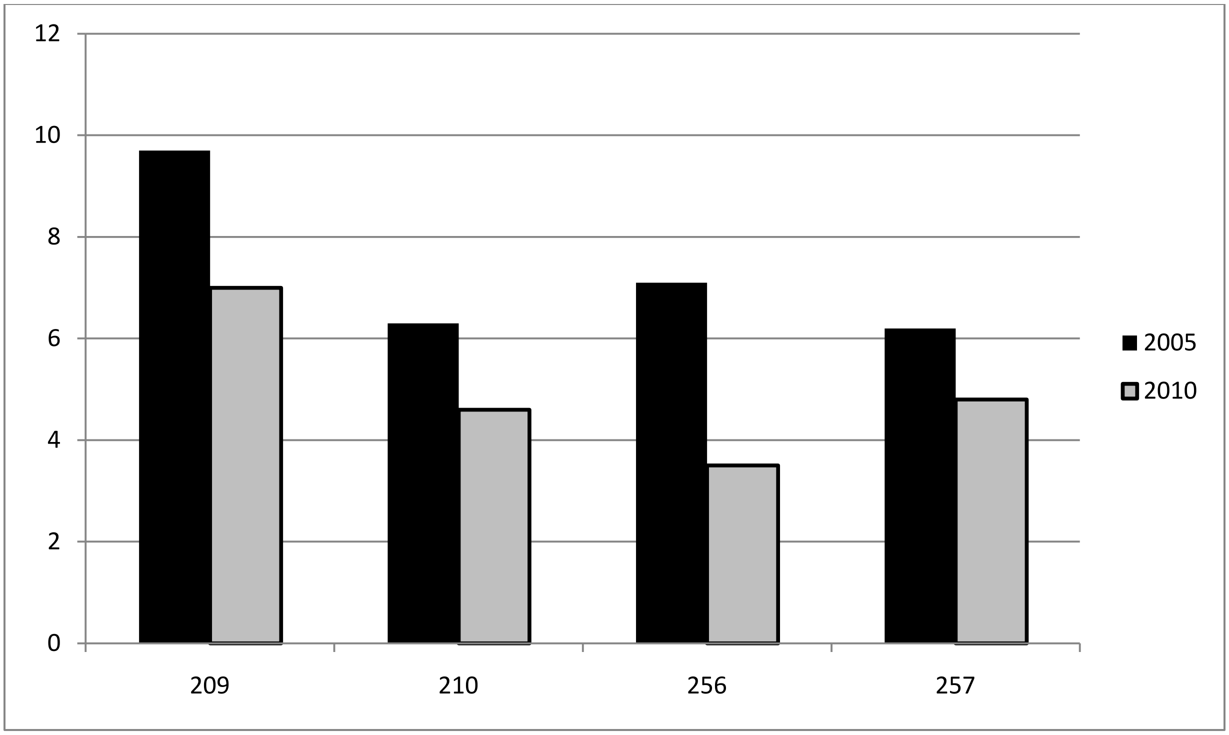


Figure 2. Aerial count results in early antlerless deer permit areas in northwest Minnesota, 2005 and 2010.

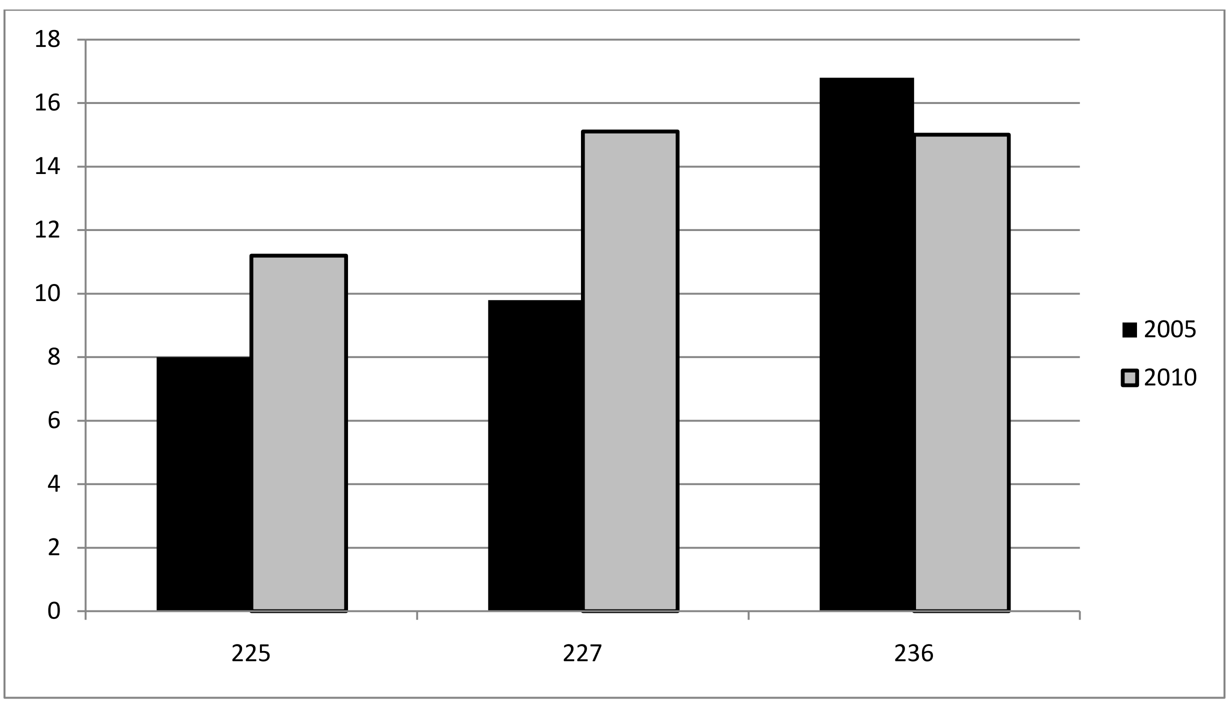


Figure 3. Aerial count results in early antlerless deer permit areas in the north metro, Minnesota, 2005 and 2010.

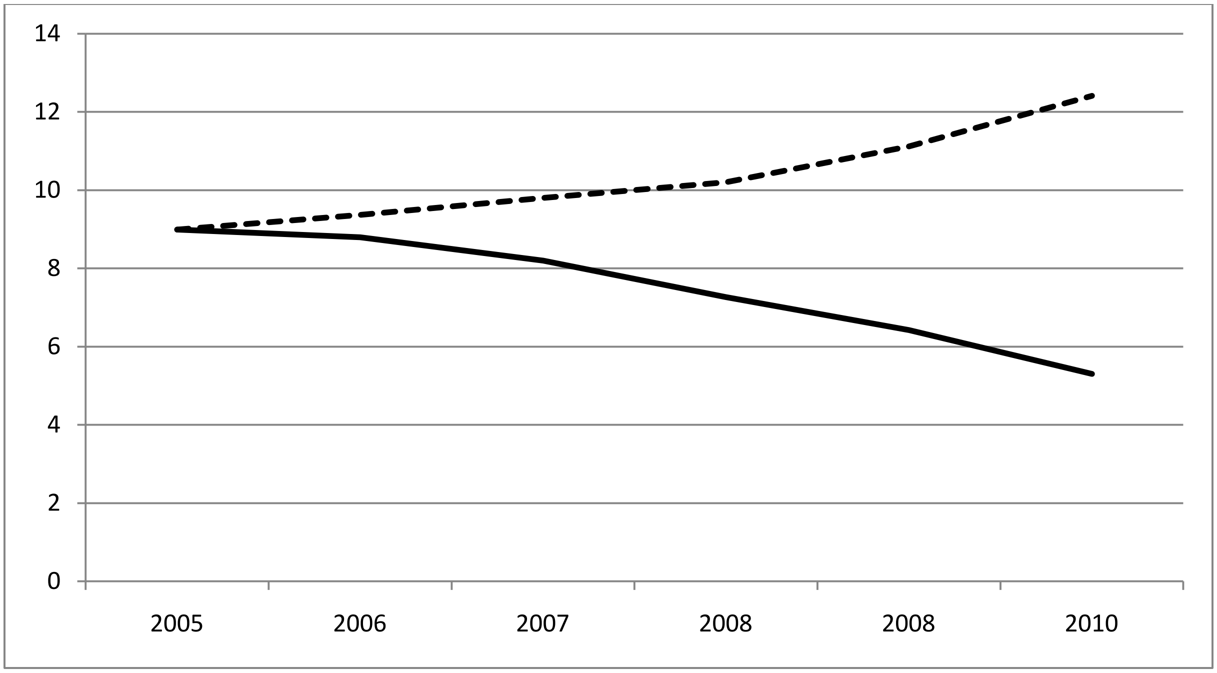


Figure 4. Modeling results depicting what actually occurred through modeling harvest data including harvests from the early antlerless season (solid line) and what would have occurred had the early antlerless season had not been held (dashed line) in northwest Minnesota deer permit areas, 2005-2010.

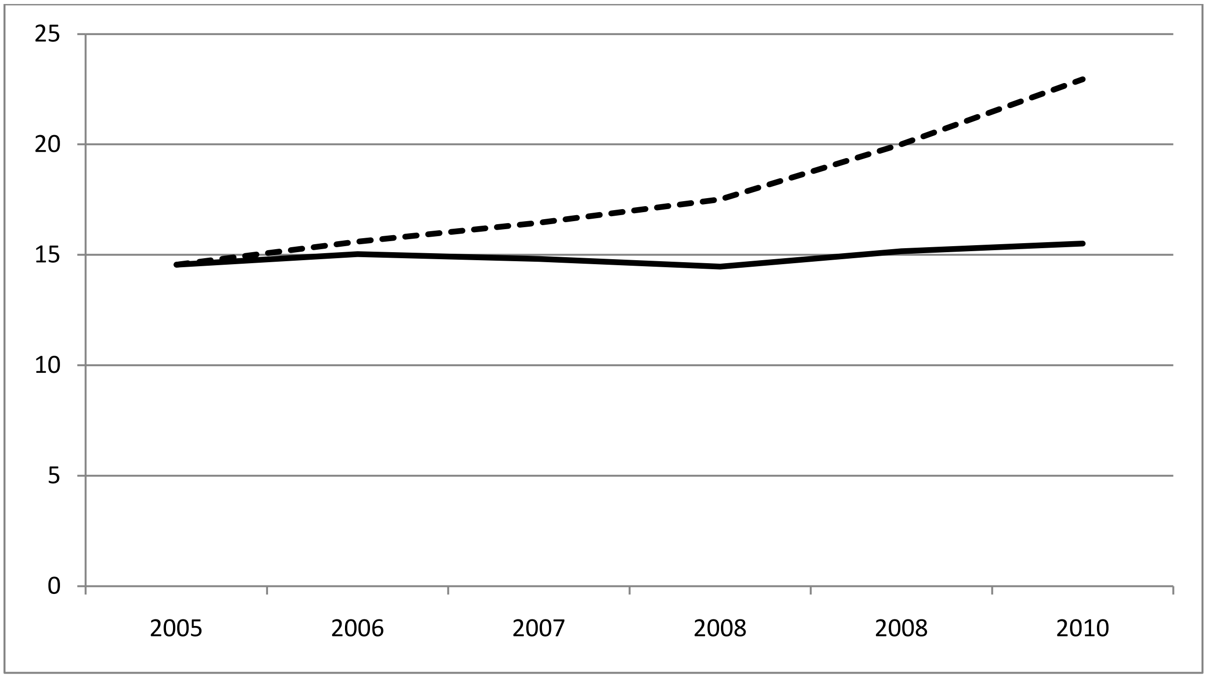


Figure 5. Modeling results depicting what actually occurred through modeling harvest data including harvests from the early antlerless season (solid line) and what would have occurred had the early antlerless season had not been held (dashed line) in north metro Minnesota deer permit areas, 2005-2010.

SURVIVAL ANALYSIS AND COMPUTER SIMULATIONS OF LETHAL AND CONTRACEPTIVE MANAGEMENT STRATEGIES FOR URBAN DEER[1]

Marrett Grund

ABSTRACT

I monitored survival of 34 female white-tailed deer (*Odocoileus virginianus*) in Bloomington, Minnesota, between October 1996 and December 1999. Twenty deer died: 19 were killed by vehicles, and 1 was killed in a deer removal program conducted by an adjacent suburb. Summer survival was high and varied little across the 3 years of study (range = 0.93–0.95), fall survival ranged from 0.84–1.00, and winter survival was generally high during the 3 years of study except during a severe winter (range = 0.72–0.95). I calculated population growth rates (λ) from Leslie matrix projections using these survival estimates and productivity data collected from roadkilled female deer in the Twin Cities metropolitan area. When winter survival was high (0.94), my model simulations indicated the Bloomington deer population increased by 21% when no deer management program was in place. When a low winter survival rate (0.72) was modeled, the population decreased by 7% even when no deer management program was implemented. I modeled the impact contraception may have on population growth and concluded that treating >50% of adult females was necessary to stabilize population growth and treating all females was necessary to decrease population growth under high winter survival conditions. I concluded that removal programs are more effective than immunocontraception programs because survival contributes more to population growth rates in deer populations than fecundity. I recommend removing 20% and 40% of adult females in the population to cause the population to stabilize or to reduce deer numbers, respectively. I recommend managers collect deer-vehicle collision data because these data potentially represent the most accurate and easily obtainable life history component of an urban deer herd.

[1]Human-Wildlife Interactions 4(2): Fall 2010 *in press*

RESEARCH PROPOSAL: CONTROLLING ENCROACHMENT OF WOODY VEGETATION IN GRASSLANDS

Kurt Haroldson

SUMMARY OF FINDINGS

Expansion of woody vegetation has become one of the greatest threats to grassland and prairie landscapes. The purpose of this study is to compare the effectiveness of various combinations of burning, mechanical, and herbicide treatments for reducing abundance of woody vegetation in grasslands in the prairie-transition zone of Minnesota. Because woody plants have developed strategies to recover from periodic disturbance, we will apply burning, mowing, and herbicide treatments repeatedly over 3 years in an attempt to deplete root reserves and ultimately kill woody plants on treated sites. We will assess the effectiveness of treatments on woody vegetation by measuring the change in canopy cover and stem density in response to each treatment. To evaluate potential unintended effects of treatments on herbaceous vegetation, we will estimate cover and frequency of grasses and forbs twice annually during each growing season. The results of this study will help guide managers in identifying the most effective approaches to maintaining high quality prairie and grasslands.

JUSTIFICATION

Grassland management is one of the most important activities of wildlife managers in Minnesota's prairie and transition zones. Restoring and maintaining grassland/prairie habitats are necessary for successful management of grassland wildlife, including waterfowl (Minnesota Department of Natural Resources [MNDNR] 2006a), sharp-tailed grouse (*Tympanuchus phasianellus*; Berg 1997), prairie chickens (*Tympanuchus cupido*; Svedarsky et al. 1997), pheasants (*Phasianus colchicus*; MNDNR 2005), and grassland songbirds (MNDNR 2006b).

Historically, the dominant threat to grasslands in the U.S. was conversion to agriculture (Samson and Knopf 1994). Although agricultural conversions continue today, expansion of woody vegetation has become one of the greatest threats to grasslands (Heisler et al. 2003, Briggs et al. 2005). Prior to European settlement, expansion of woody vegetation was constrained by frequent fire and low abundance of woody vegetation. During the past 150 years, however, fire suppression and deliberate planting of trees and shrubs for windbreaks and shelterbelts have resulted in a relatively uniform distribution of woody seed sources throughout the prairie-transition zones of Minnesota. Furthermore, human and lightning-caused fires before European settlement occurred during spring, summer, and fall (Bragg 1982, Higgins 1984, McCain and Elzinga 1994), but grassland/prairie management today emphasizes spring burning. Thus, proximity to woody seed sources and season of burning have changed since European settlement.

Heisler et al. (2003) reported that expansion of woody vegetation in a tallgrass prairie of eastern Kansas was constrained by annual spring burning, but not by spring burning on a return interval ≥4 years. Spring burning acted as a pruning mechanism for aboveground shoots of woody vegetation, but post-fire increases in light and nitrogen stimulated vigorous resprouting and growth of woody vegetation (McCarron and Knapp 2003). Thus, spring fire may be required annually to constrain expansion of some woody species once established in grasslands, but spring fire alone may not be sufficient to eliminate co-dominance of woody vegetation (Heisler et al. 2003, McCarron and Knapp 2003, Briggs et al. 2005).

Carbohydrate reserves in plants vary seasonally following a cycle of depletion and restoration related to the growth cycle of the plant (Miller 2000). Mortality rates of woody vegetation can be enhanced by repeated prescribed burning during low carbohydrate periods. Hardwoods in the understory of conifer forests were effectively controlled in Minnesota (Buckman 1964), Colorado (Harrington 1989), and the Southeast (Hodgkins 1958, Waldrop and

Lloyd 1991) by repeated prescribed burning during summer when carbohydrate reserves were low. In grassland environments, however, researchers have expended comparatively little effort to investigate effects of growing-season fire on woody and herbaceous plants. Adams et al. (1982) eliminated dogwood (*Cornus drummondii*), green ash (*Fraxinus pennsylvanica*), and cottonwood (*Populus deltoides*) from an Oklahoma grassland with a summer burn. Growing season fire controlled woody vegetation and enhanced desirable forbs in a Tennessee grassland better than dormant season fire, herbicides, or summer mowing (Gruchy et al. 2006). Howe (1995) found that summer fires in a floodplain grassland in Wisconsin delayed the progression to dominance of large, late-flowering C_4 (warm season) grasses (e.g., big bluestem [*Andropogon gerardii*]) and allowed early flowering species, which virtually disappeared in spring-burned or unburned plots, to persist or even prosper. However, 2 cycles of summer fire over 3 years favored a mixture of C_3 (cool season) and C_4 grasses, including the aggressive C_3 reed canary grass (*Phalaris arundinacea*), which was planted in the study plots (Howe 2000).

In a survey of grassland information needs, MNDNR wildlife managers requested help in finding solutions for managing woody encroachment of grasslands (Tranel 2008). The purpose of this study is to compare the effectiveness of various combinations of burning, mechanical, and herbicide treatments for reducing abundance of woody vegetation in grasslands in the prairie-transition zone of Minnesota. Our objectives are to: (1) measure the change in density and cover of woody vegetation in response to treatments; (2) describe relative responses of C_3 versus C_4 grasses and forbs to treatments and seasonal timing of treatments; and (3) identify factors that influence the response of woody and herbaceous vegetation to treatments.

STUDY AREA

This study will be conducted on grassland sites in the prairie-transition zone of Minnesota that have experienced recent encroachment by woody vegetation. Sites suitable for this study should have been protected from grazing and fire for at least 1 year prior to application of study treatments, and encroaching woody vegetation should be relatively young (largest trees ≤ 4 inches in diameter). Sites may include properties managed as Wildlife Management Areas, Waterfowl Production Areas, National Wildlife Refuges, The Nature Conservancy Reserves, or private lands (e.g., Conservation Reserve Program lands). The location and number of sites will be determined based on pilot study results, conformity of sites to criteria described above, willingness of managers to fully participate in the study design, and available budget.

METHODS

Because woody plants have developed strategies to recover from periodic disturbance (Miller 2000), repeated applications of burning, mowing, and herbicide treatments will likely be required over time to deplete root reserves and ultimately kill woody plants on study sites. Therefore, this study will be conducted over 5 years (2011-2015). A pilot study during 2010 will be used to evaluate sampling techniques, determine the number of study sites needed, and select suitable study sites. Pre-treatment vegetation surveys will be conducted during 2011. Treatments (Table 1) will be applied during 2012-2014. The number of treatments used and their assignment to study sites will depend on results of the pilot study. We will attempt to use the pilot (pre-treatment) data to define strata (or blocking factors) within which treatments will be randomly assigned. When forming strata, we will consider factors such as size and density of woody cover patches, species composition, site hydrology, and geographic region.

We will attempt to apply seasonal treatments based on phenology of woody vegetation rather than absolute date. Mowing and herbicide treatments will be applied only to those portions of plots that contain woody vegetation. Vegetation cut during mowing treatments will not be harvested or otherwise removed from study sites. Because variability in quality of burns may confound effects of burn season, we will estimate fire conditions associated with each burn.

Prior to application of burn treatments, we will estimate fuel load using dry mass of clippings of live herbaceous vegetation and dead litter. On the day of a burn, we will record weather conditions (e.g., air temperature, wind speed, humidity) and estimate fuel moisture and fire intensity (Johnson 1992). The proportion of above-ground standing vegetation and litter that is consumed by fire will be estimated after each prescribed burn. We will attempt to distinguish portions of plots burned by head fires versus back fires and flank fires.

Vegetation surveys will be conducted twice annually during the growing season (late spring and late summer) during 2011-2015. Canopy cover and density of woody vegetation will be estimated by line-intercept methods (Canfield 1941) using permanent transects established through patches of woody vegetation. In addition, a sample of individual woody plants located along transects will be selected for detailed study. Species, diameter at breast height, and life status (alive versus dead) will be recorded for selected trees during each sampling period. Similarly, species, clump diameter (measured along cardinal axes), and life status will be recorded for selected shrubs during each sampling period.

Cover and frequency of herbaceous plant species will be measured along permanent transects, half located in patches containing woody vegetation (the same transects used for woody vegetation sampling) and half located in herbaceous-only vegetation. Each transect will contain circular 10 m^2 plots distributed at 10-m intervals. During each sampling period, canopy cover of each plant species will be estimated using a modified Daubenmire scale (Daubenmire 1959, Abrams and Hulbert 1987). Frequency of plant species will be estimated as the proportion of plots containing a species.

Because initial canopy cover, frequency, and density estimates will vary among sites, we will calculate change in values from 2011. When seasonal estimates differ, we will use the larger of the 2 values for subsequent analyses. Effects of treatments on changes in vegetation canopy, frequency, and density will be tested with analysis of variance.

ACKNOWLEDGEMENTS

I thank G. Hoch and many natural resource managers for framing the early concepts for this project. M. Tranel, E. Dunton, R. Kimmel, J. Giudice, and J. Fieberg provided comments on earlier drafts of this proposal.

LITERATURE CITED

Abrams, M. D., and L. C. Hulbert. 1987. Effect of topographic position and fire on species composition in tallgrass prairie in northeast Kansas. American Midland Naturalist 117: 442-445.

Adams, D. E., R. C. Anderson, and S. L. Collins. 1982. Differential response of woody and herbaceous species to summer and winter burning in an Oklahoma grassland. Southwestern Naturalist 27: 55-61.

Berg, W. E. 1997. The sharp-tailed grouse in Minnesota. Wildlife Report 10, Minnesota Department of Natural Resources, St. Paul, Minnesota.

Bragg, T. B. 1982. Seasonal variations in fuel and fuel consumption by fires in a bluestem prairie. Ecology 63: 7-11.

Briggs, J. M., A. K. Knapp, J. M. Blair, J. L. Heisler, G. A. Hoch, M. S. Lett, and J. K. McCarron. 2005. An ecosystem in transition: causes and consequences of the conversion of mesic grassland to shrubland. BioScience 55: 243-254.

Buckman, R. E. 1964. Effects of prescribed burning on hazel in Minnesota. Ecology 45: 626-629.

Canfield, R. H. 1941. Application of the line intercept method in sampling range vegetation. Journal of Forestry 39: 388-394.

Daubenmire, R. 1959. A canopy-coverage method of vegetation analysis. Northwest Science 33: 43-66.

Gruchy, J. P., C. A. Harper, and M. J. Gray. 2006. Methods for controlling woody invasion into CRP fields in Tennessee. Pages 315-321 *in* S. B. Cederbaum, B. C. Faircloth, T. M. Terhune, J. J. Thompson, and J. P. Carroll, editors. Gamebird 2006: Quail VI and Perdix XII. Warner School of Forestry and Natural Resources, Athens, Georgia, USA.

Harrington, M. G. 1989. Gambel oak root carbohydrate response to spring, summer, and fall prescribed burning. Journal of Range Management 42: 504-507.

Heisler, J. L., J. M. Briggs, and A. K. Knapp. 2003. Long-term patterns of shrub expansion in a C_4-dominated grassland: fire frequency and the dynamics of shrub cover and abundance. American Journal of Botany 90: 423-428.

Higgins, K. F. 1984. Lightning fires in North Dakota grasslands and in pine-savanna lands of South Dakota and Montana. Journal of Range Management 37: 100-103.

Hodgkins, E. J. 1958. Effects of fire on undergrowth vegetation in upland southern pine forests. Ecology 39: 36-46.

Howe, H. F. 1995. Succession and fire season in experimental prairie plantings. Ecology 76: 1917-1925.

Howe, H. F. 2000. Grass response to seasonal burns in experimental plantings. Journal of Range Management 53: 437-441.

Johnson, E. A. 1992. Fire and vegetation dynamics. Cambridge University Press, Cambridge, United Kingdom.

McCain, W. E., and S. L. Elzinga. 1994. The occurrence of prairie and forest fires in Illinois and other Midwestern states, 1679 to 1854. Erigenia 13: 79-90.

McCarron, J. K., and A. K. Knapp. 2003. C_3 shrub expansion in a C_4 grassland: positive post-fire responses in resources and shoot growth. American Journal of Botany 90: 1496-1501.

Miller, M. 2000. Chapter 2: fire autecology. Pages 9-34 *in* J. K. Brown and J. K. Smith, editors. Wildland fire in ecosystems: effects of fire on flora. General Technical Report RMRS-GTR-42-vol. 2. U.S. Department of Agriculture, Forest Service, Rocky Mountain Research Station, Ogden, Utah, USA.

Minnesota Department of Natural Resources. 2005. Long range plan for the ring-necked pheasant in Minnesota. Division of Fish and Wildlife, Minnesota Department of Natural Resources, St. Paul, Minnesota.

Minnesota Department of Natural Resources. 2006a. Long range duck recovery plan. Division of Fish and Wildlife, Minnesota Department of Natural Resources, St. Paul, Minnesota.

Minnesota Department of Natural Resources. 2006b. Tomorrow's habitat for the wild and rare: an action plan for Minnesota Wildlife. Comprehensive wildlife conservation strategy, Division of Ecological Services, Minnesota Department of Natural Resources, St. Paul, Minnesota.

Sampson, F., and F. Knopf. 1994. Prairie conservation in North America. BioScience 44:418-421.

Svedarsky, W. D., T. J. Wolfe, and J. E. Toepfer. 1997. The greater prairie chicken in Minnesota. Wildlife Report 11, Minnesota Department of Natural Resources, St. Paul, Minnesota.

Tranel, M. A. 2008. Management-focused research needs of Minnesota's wildlife managers-grassland management activities. Pages 91-95 *in* M.W. DonCarlos et al. (eds.), Summaries of Wildlife Research Findings 2007. Minnesota Department of Natural Resources, Division of Fish and Wildlife, Wildlife Populations and Research Unit. St. Paul, Minnesota.

Waldrop, T. A., and F. T. Lloyd. 1991. Forty years of prescribed burning on the Santee fire plots: effects on overstory and midstory vegetation. Pages 45-50 *in* S. C. Nodvin and T. A. Waldrop, editors. Fire and the environment: ecological and cultural perspectives. General Technical Report SE-69, U.S. Department of Agriculture, Forest Service, Southeastern forest Experiment Station, Asheville, North Carolina, USA.

Table 1. Examples of burning, mowing, and herbicide treatments being considered for application during 2012-2014 to control woody vegetation on grasslands in the prairie-transition zone of Minnesota.

Treatment	2012	2013	2014
1	Spring burn	Rest	Rest
2	Spring burn	Spring burn	Spring burn
3	Spring burn, summer mow	Rest	Spring burn, summer mow
4	Summer mow	Summer mow	Summer mow
5	Summer burn	Summer mow	Summer burn
6	Summer mow	Summer herbicide[a]	Summer mow

[a]Folier application of Garlon 3A.

SURVIVAL OF WILD TURKEY HENS TRANSPLANTED BEYOND THEIR CURRENT DISTRIBUTION IN MINNESOTA[1]

Chad J. Parent, Brett J. Goodwin, and Eric M. Dunton

ABSTRACT

The current distribution of eastern wild turkeys in Minnesota extends well beyond their ancestral range. Severe winter conditions are believed to prevent turkeys from persisting in areas beyond their current northern distribution. Biologists and wildlife managers understand that turkeys are physiologically capable of surviving outside of their current distribution if food is available during winter. However, winter severity influences the availability of food. We transplanted radioed female wild turkeys to northwestern (Red Lake and Pennington Counties) Minnesota during winter in 2006 and 2007 to investigate the viability of turkeys north of their current distribution. These areas were located approximately 55 km north of the current distribution of turkeys in Minnesota, and represent one of the northern most transplants in North America. We estimated winter (1 January – 31 March) and annual survival probabilities of female wild turkeys using the Kaplan-Meier method, compared winter conditions to historical climate data to evaluate winter severity, and identified the cause of mortality during winter. Winter in 2006 was average (i.e., winter conditions similar to climate averages) at our study areas and survival estimates were 0.300 (SE = 0.077). Survival estimates increased to 0.820 (SE = 0.075) in 2007 following a mild winter (i.e., higher temperature and less snow relative to climate averages). Survival estimates in 2006 were low, but consistent with survivorship estimates previously observed in central Minnesota. Survival estimates in 2007 were among the highest observed in Minnesota. We identified the cause of mortality in 61% of female turkeys (50% due to predation, 9%, due to starvation, and 2% due to vehicle collision), while a lack of evidence precluded us from identifying the cause in the remaining mortality events. Turkeys are capable of persisting in northwestern Minnesota and winter is not a limiting factor. However, in extreme northern regions, localized periods of severe winter conditions appear to influence survival.

[1]Proceedings of the National Wild Turkey Symposium 10. *in press*

EFFECTS OF SUPPLEMENTAL FOOD AND EXPERIENCE ON WINTER SURVIVAL OF TRANSPLANTED WILD TURKEYS[1]

Marco Restani, Richard O. Kimmel, John R. Fieberg, and Sharon L. Goetz

ABSTRACT

Wildlife biologists have provided supplemental food during winter to improve post-release survival of Wild Turkeys (*Meleagris gallopavo*) transplanted north of their ancestral range in Minnesota. We evaluated the effectiveness of this action by monitoring overwinter and annual survival of 140 transplanted turkeys on three supplemental food and three control study areas in 2004 and 2005. Both winters of study were mild relative to historic snowfall levels and temperature. Patterns of mortality during winter were consistent across years with most mortalities occurring on control study sites. Turkeys that had been released in the prior year and survived until January of the current year had little mortality, regardless of supplemental food. The relative risk of death estimated from proportional hazards models for turkeys at supplemental food sites relative to those at control sites during winter was 5.0 in 2004 and 9.7 in 2005. Estimates of relative risk for newly released relative to experienced turkeys during winter were 9.4 in 2004 and 12.6 in 2005. Site-to-site variability in risk decreased during the non-winter period with treatment and control sites having more similar risk levels. Ninety-one turkeys died and mammalian predation was the most common cause of known mortality.

[1]The Wilson Journal of Ornithology 121(2):366–377, 2009

ESTABLISHMENT AND MAINTENANCE OF FORBS IN EXISTING GRASS STANDS- PILOT SEASON UPDATE

Molly Tranel

SUMMARY OF FINDINGS

Managers requested more information on establishing and maintaining an abundance and diversity of forbs in grasslands. Survival of forbs interseeded directly into existing vegetation may be enhanced by management treatments that reduce competition with established grasses. In this study, I will investigate the effects of 2 mowing and 2 herbicide treatments on diversity and abundance of forbs interseeded into established grasslands in the farmland region of Minnesota, and I will monitor insect response to interseeding treatments. I selected 17 study sites, each ≥4 ha and characterized by similar soils and a relatively uniform stand of forb-deficient native grass. I will apply 2 mowing treatments (once or twice per season) and 2 grass-selective herbicide treatments (high and low rate) during the 2010 growing season while interseeded forbs are becoming established. During 2011-2013, I will compare species richness and structural characteristics of vegetation among treatment and control plots at each study site. I will also estimate insect abundance and diversity on all sites as well as some additional native prairie sites.

INTRODUCTION

In a survey on grassland information needs (Tranel 2008), 82% of Minnesota Department of Natural Resources (MNDNR) wildlife managers indicated a need for information on maintaining plant species diversity in restored grasslands. In particular, managers wanted more information on establishing and maintaining an abundance and diversity of forbs in grasslands. A diversity of forbs in grasslands provides the heterogeneous vegetation structure needed by some bird species for nesting and brood rearing (Volkert 1992, Sample and Mossman 1997). Forbs also provide habitat for invertebrates, an essential food for grassland birds and their broods (Buchanan et al. 2006). Insect abundance in chick diets has been positively correlated with growth rates and survival in gallinaceous birds such as grouse (Park et al. 2001, Huwer et al. 2008), gray partridge (*Perdix perdix;* Sotherton and Robertson 1990), and pheasants (*Phasianus cholchicus*; Hill 1985). Broods of gallinaceous birds such as prairie chickens (*Tympanuchus cupido*) move directly from nests to brood habitat (Svedarsky 1979), and habitats with high forb abundance were preferred (Jones 1963, Drobney and Sparrow 1977).

The forb component on many restored grasslands has been lost or greatly reduced. Managers interested in increasing the diversity and quality of forb-deficient grasslands are faced with the costly option of completely eliminating the existing vegetation and planting into bare ground, or attempting to interseed forbs directly into existing vegetation. Management techniques that reduce competition from established grasses may provide an opportunity for forbs to become established in existing grasslands (Collins et al. 1998). Temporarily suppressing dominant grasses may increase light, moisture, and nutrient availability to seedling forbs, ultimately increasing forb abundance and diversity (Schmitt-McCain 2008). Williams et al. (2007) found that frequent mowing of grasslands in the first growing season after interseeding increased forb emergence and reduced forb mortality. Similarly, Hitchmough and Paraskevopoulou (2008) found that forb density, biomass, and richness were greater in meadows where a grass herbicide was used.

In this study, I will investigate the effects of 2 mowing and 2 herbicide treatments on diversity and abundance of forbs interseeded into established grasslands in southern Minnesota. In addition, I will monitor insect abundance in response to interseeding treatments.

Finally, I will track the cost of implementing each management technique and conduct a cost-benefit analysis.

STUDY AREA

I selected 17 study sites distributed throughout the southern portion of Minnesota's prairie/farmland region (Figure 1), including 16 sites on state-owned Wildlife Management Areas (WMA) and 1 site on a federally owned Waterfowl Production Area (WPA). Each site was ≥4 ha and characterized by similar soils, hydrology, and vegetative composition. All sites were dominated by relatively uniform stands of native grasses with no or few forbs, although invasive grasses were present at most sites. I evaluated feasibility of treatments and potential for identifying forb seedlings on a separate pilot site, Wood Lake WMA.

METHODS

Although I intended to prepare each site for dormant-season interseeding by burning in fall 2009, an unusually wet October did not allow for burning at 9 of the 17 sites (Figure 1). As a result, 8 sites were burned in fall 2009 and frost interseeded during late fall and winter, and the remaining 9 sites were burned and interseeded during spring 2010. Seed used on spring-burned sites was cold-moist stratified for 3-5 weeks in a mixture of wet sand to stimulate germination during spring 2010 at all sites.

Treatments

After each site was prepared and seeded, I divided them into 10 plots of approximately equal size. I randomly assigned each of 4 treatments and the control to 2 of the 10 plots (i.e., each of the 4 treatments and control will be replicated twice within each site). The following treatments, which are designed to suppress grass competition, will be applied during the 2010 growing season while the forbs are becoming established:

- Mow to a height of 10-15 cm once when vegetation reaches 25-35 cm in height.
- Mow to a height of 10-15 cm twice when vegetation reaches 25-35 cm in height.
- Apply grass herbicide Clethodim (Select Max) at 108 mL/ha (9 oz/A) when vegetation reaches 10-15 cm.
- Apply grass herbicide Clethodim (Select Max) at 215 mL/ha (18 oz/A) when vegetation reaches 10-15 cm.

Vegetation Sampling

Prior to burning and interseeding, all sites were surveyed by a botanist in summer 2009 to determine species already present and general condition of each site. This also allowed for field testing of the vegetation survey protocol. Four transects 50 meters in length were randomly located within each study plot and recorded using a Global Positioning System unit. We estimated percent cover of live vegetation (Daubenmire 1959) every 5 m and litter depth every 10 meters. We recorded visual obstruction readings (VOR; Robel et al. 1970) in the 4 cardinal directions at the beginning and the end of each transect. Species richness was estimated by counting the number of species present in each sampling frame.

Insect Sampling

I will estimate insect abundance and diversity at each site as a separate part of this study. Protocols for collection of insects will be determined in a pilot study during the summer of 2010. Insect sampling will begin in full during the summer of 2011, one year after management treatments (and disturbance to insects) have ended.

Timeline

Summer 2010: apply management treatments, begin insect pilot sampling
Summer 2011: monitor vegetation and insects (sample June and September)
Summer 2012: monitor vegetation and insects (sample June and September)
Summer 2013: monitor vegetation and insects (sample June and September)

RESULTS

The Wood Lake pilot site was frost interseeded in January 2009. Three of the 4 treatments (mow once, herbicide at low rate, and herbicide at high rate) were applied at the site during the summer of 2009. Due to staffing limitations, herbicide treatments were applied when the grass was taller (31 cm) than prescribed (10-15 cm). One month after treatments were applied, average VOR was shorter in treated plots than in control plots (Figure 2), indicating that the prescribed treatments were effective in suppressing growth of grass. VOR was similar for low and high herbicide treatments. Black-eyed susan (*Rudbeckia hirta*) and goldenrod (*Solidago spp.*) seedlings were observed in most of the treatment plots. Also observed occasionally in some of the plots were partridge pea (*Chamaechrista fasciculata*), and purple prairie clover (*Dalea purpurea*) seedlings.

ACKNOWLEDGEMENTS

I thank U.S. Fish and Wildlife Service and MNDNR managers for providing the land, equipment, and labor for this project. I thank G. Brand and J. Swanson for assisting with field work. K. Haroldson and R. Kimmel provided comments on an earlier draft of this report.

LITERATURE CITED

Buchanan, G. M., M. C. Grant, R. A. Sanderson, and J. W. Pearce-Higgins. 2006. The contribution of invertebrate taxa to moorland bird diets and the potential implications of land-use management. Ibis 148:615-628.

Collins, S. L., A. K. Knapp, J. M. Briggs, J. M. Blair, and E. M. Steinauer. 1998. Modulation of diversity by grazing and mowing in native tallgrass prairie. Science 280:745-747.

Daubenmire, R.F. 1959. Canopy coverage method of vegetation analysis. Northwest Science 33:43-64.

Drobney, R. D., and R. D. Sparrow. 1977. Land use relationships and movements of greater prairie chickens in Missouri. Transactions of the Missouri Academy of Science 10-11: 146-160.

Hill, D. A. 1985. The feeding ecology and survival of pheasant chicks on arable farmland. Journal of Applied Ecology 22:645-654.

Hitchmough, J. and A. Paraskevopoulou. 2008. Influence of grass suppression and sowing rate on the establishment and persistence of forb dominated urban meadows. Urban Ecosyst 11:33-44.

Huwer, S. L., D. R. Anderson, T. E. Remington, and G. C. White. 2008. Using human-imprinted chicks to evaluate the importance of forbs to sage grouse. Journal of Wildlife Management 72: 1622-1627.

Jones, R. E. 1963. Identification and analysis of lesser and greater prairie chicken habitat. Journal of Wildlife Management 27:757-778.

Park, K. J., P. A. Robertson, S. T. Campbell, R. Foster, and Z. M. Russell, D. Newborn, and P. J. Hudson. 2001. The role of invertebrates in the diet, growth and survival of red grouse (L*agopus lagopus scoticus*) chicks. Journal of Zoology 254:137-145.

Robel, R.J., J.N. Briggs, A.D. Dayton, and L.C. Hulbert. 1970. Relationships between visual obstruction measurements and weight of grassland vegetation. Journal of Range Management 23:295-297.

Sample, D. W. and M. J. Mossman. 1997. Managing habitat for grassland birds, a guide for Wisconsin. Wisconsin Department of Natural Resources, Bureau of Integrated Science Services, Madison, USA.

Schmitt-McCain, K. N. 2008. Limitations to plant diversity and productivity in restored tallgrass prairie. Thesis. Kansas State University, Manhattan, USA.

Sotherton, NW, and PA Robertson. 1990. Indirect impacts of pesticides on the production of wild gamebirds in Britain. Pages 84-102 *in* KE Church, RE Warner, and SJ Brady, editors. Perdix V: gray partridge and ring-necked pheasant workshop. Kansas Department of Wildlife and Parks, Emporia, Kansas, USA.

Svedarsky, W. D. 1979. Spring and summer ecology of female greater prairie chickens in northwestern Minnesota. Dissertation, University of North Dakota, Grand Forks, USA.

Tranel, MA. 2008. Research Proposal: establishment and maintenance of forbs in existing grass stands. *In* MW DonCarlos, RO Kimmel, JS Lawrence, and MS Lenarz, editors. Summaries of Wildlife Research Findings. Minnesota Department of Natural Resources, St. Paul, MN.

Tranel, MA. 2009. Management focused research needs of Minnesota's wildlife managers-grassland management activities. *In* MW DonCarlos, RO Kimmel, JS Lawrence, and MS Lenarz, editors. Summaries of Wildlife Research Findings. Minnesota Department of Natural Resources, St. Paul, MN.

Volkert, W. K. 1992. Response of grassland birds to a large-scale prairie planting project. The Passenger Pigeon 54: 191-195.

Willliams, DW, LL Jackson, and DD Smith. 2007. Effects of frequent mowing on survival and persistence of forbs seeded into a species-poor grassland. Restoration Ecology 15:24-33.

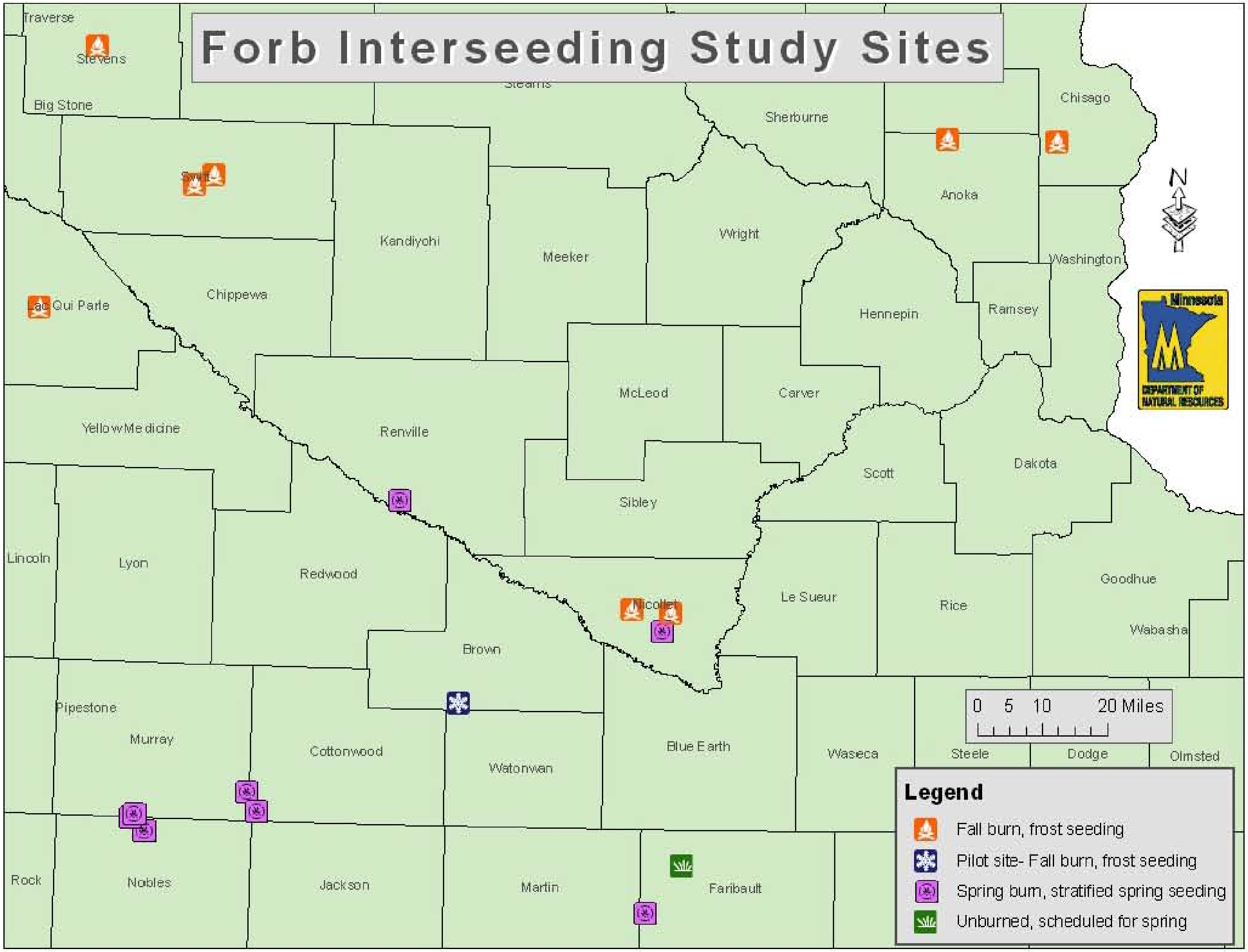


Figure 1. Locations of study sites for forb interseeding study, categorized by season of interseeding, southern Minnesota, 2009.

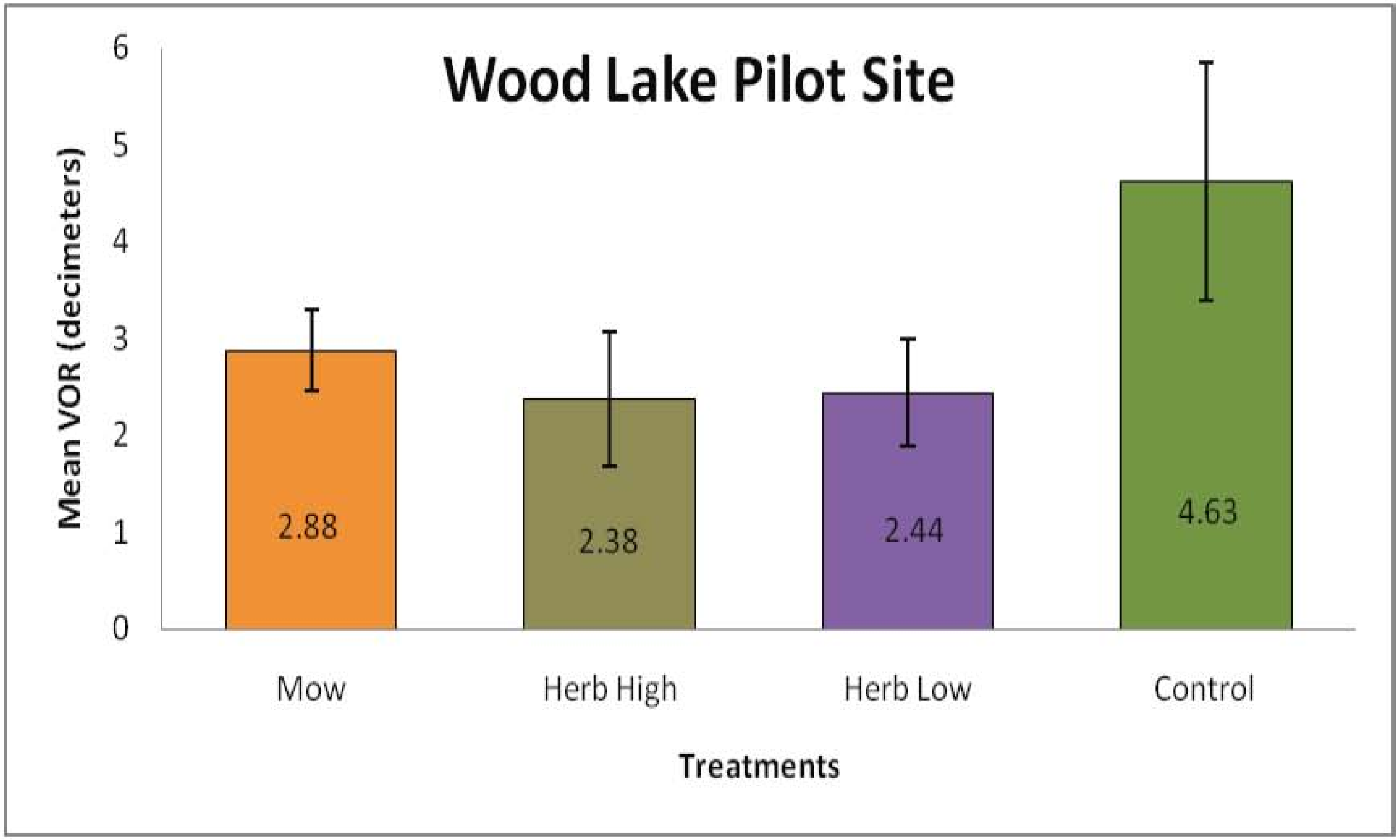


Figure 2. Mean (± SD) VOR by treatment type at the pilot study site, Wood Lake, Minnesota, 2009. Treatments included mow once, herbicide at high rate (215 mL/ha), and herbicide at low rate (108 mL/ha).

RESEARCH PROPOSAL: MONITORING PRAIRIE INVERTEBRATE ABUNDANCE AND DIVERSITY TO INFORM BEST GRASSLAND MANAGEMENT PRACTICES

Molly Tranel and Daren Carlson

INTRODUCTION

Invertebrates play critical functional roles in the prairie community from pollination to serving as essential food sources for grassland birds and other animals. The goal of this project is to evaluate methods to estimate diversity and abundance of invertebrates in grassland habitats, and to use the developed protocol to monitor invertebrate communities in both native prairies and planted grasslands. The project will be conducted on a series of native prairie sites paired with planted grassland sites located primarily in Minnesota's Prairie Parkland Province. The results from this project will provide information to more effectively monitor important components of native prairie and surrogate grasslands, and identify grassland management techniques that maintain or improve prairie and surrogate grassland habitat for prairie Species of Greatest Conservation Need (SGCN) and other wildlife.

JUSTIFICATION

Because many invertebrates are solely associated with native prairie and play critical functional roles, they have been identified as a key animal group for monitoring (Kremen et al. 1993). Fifteen insect species and 8 spider species, including the Red Tailed Prairie Leafhopper (*Aflexia rubranura*), Dakota Skipper (*Hesperia dacotae*), and *Marpissa grata* – a species of jumping spider, are prairie-associated SGCN. Invertebrates are crucial to healthy prairie ecosystems functions such as pollination, nutrient cycling (Arenz and Joern 1996), and decomposition (Whiles and Charlton 2005). Furthermore, invertebrates are an essential food for grassland birds and their broods (Buchanan et al. 2006). Yet, information on prairie invertebrates is sparse.

Recent acceleration of efforts to maintain or restore prairies have accentuated the need for long term data collection, storage, and analysis using a consistent set of monitoring protocols to: 1) detect changes and long-term trends (status and trend monitoring) and 2) evaluate the success of prairie management and restoration activities (effectiveness monitoring). Estimates of invertebrate diversity and abundance are the best measures of habitat quality for prairie invertebrates. However, some invertebrate species with a close functional relationship to prairie plant species may serve as indicators of prairie condition and quality.

The purpose of this project is to develop methods for monitoring the status and trends of invertebrate communities across a range of grassland habitats from high quality prairies to planted grasslands, and for monitoring the effectiveness of management treatments intended to maintain or improve quality of grassland habitats. Our objectives are to: 1) evaluate the effectiveness of 3 invertebrate sampling methods (i.e., pit traps vs. sweep nets vs. vacuum sampler) for estimating invertebrate diversity and abundance; and 2) identify invertebrate taxa that may serve as indicators for trend and effectiveness monitoring of grassland habitats. This proposal expands on 2 studies currently in progress. The first is a study on vegetation and bird diversity on high-quality prairie sites in western Minnesota. The second is a study evaluating methods for establishing and maintaining forbs in existing species-poor grasslands (Tranel 2009).

STUDY AREA

Study sites will be chosen from high quality prairie sites and planted grasslands in Anoka, Chisago, Stevens, Pope, Swift, Lac Qui Parle, Chippewa, Redwood, Renville, Brown, Nicollet, Blue Earth, Faribault, Nobles, Murray, and Cottonwood Counties (Figure 1). Sites will be paired to include a native prairie and a restored grassland site. Paired sites will be chosen so that they are geographically close and have similar soils, topography, and plant communities (e.g. dry, mesic, or wet prairie). Most sites will be located in the Prairie Parkland Province, but there may also be some sites in the Eastern Broadleaf Forest Province of Minnesota.

METHODS

During the first year of this study we will evaluate sampling methods that best estimate invertebrate diversity and abundance in both native prairies and planted grasslands. Successful methods will be those that effectively sample the greatest number of arthropods across functional groups for the least time and monetary cost. Invertebrate sampling methods will include pit traps, sweep nets, and vacuum sampling. We will preserve and identify collected invertebrates to the order and, if possible, family. We will identify a subsample of targeted taxonomic groups (e.g., ground beetles, leafhoppers) to species to determine if they could serve as indicator taxa. We will evaluate the utility of insect extractor devices (Molano-Flores 2002) to sort invertebrate samples after collection. During the first year of this study, we will also track time and monetary costs associated with collecting and processing the invertebrate samples. Based on spatial and temporal variability, we will estimate the number of samples needed per site and season to monitor invertebrate communities in native prairies and planted grasslands.

During the second and third years of this study, we will monitor invertebrates in selected prairies and planted grasslands, based on the methods developed in the first year of this study, to assess short-term trends and effects of different management techniques. We will use monitoring results to guide the development of a long-term monitoring program for invertebrate communities in grassland habitats in Minnesota.

Timeline

- Spring and Summer 2010: (*Pilot study phase*)

  Conduct literature review, determine sampling methodology, hire field staff, and begin field testing sampling methods

- Summer 2011: (*Full study phase*)

  Collect samples at all field sites, process/sort/identify invertebrate samples, send subset of samples to expert for further identification

- Summer 2012:

  Collect samples at all field sites, process/sort/identify invertebrate samples, send subset of samples to expert for further identification

ACKNOWLEDGEMENTS

We thank U.S. Fish and Wildlife Service and MNDNR managers for providing the land, equipment, and labor for this project. W. Crowell, J. Norris, K. Haroldson and R. Kimmel provided comments on an earlier draft of this report. This project was made possible through a grant from the Federal State Wildlife Grant program.

LITERATURE CITED

Arenz, CL and A Joern. 1996. Prairie legacies-invertebrates. *In* Samson, FB and FL Knopf, editors. Prairie conservation: preserving North America's most endangered ecosystem. Island Press, Washington DC.

Buchanan, GM, MC Grant, RA Sanderson, and JW Pearce-Higgins. 2006. The contribution of invertebrate taxa to moorland bird diets and the potential implications of land-use management. Ibis 148(4):615-628.

Kremen, C, RK Colwell, TL Erwing, DD Murphy, RF Noss, and MA Sanjayan. 1993. Terrestrial arthropod assemblages: Their use in conservation planning. Conservation Biology 7(4):796-808.

Minnesota Dept. of Natural Resources (MNDNR) 2006. Tomorrow's Habitat for the Wild and Rare: An Action Plan for Minnesota Wildlife, Comprehensive Wildlife Conservation Strategy. Division of Ecological Services, Minnesota Dept. of Natural Resources.

Molano-Flores, B. 2002. Critical Trends Assessment Program Monitoring Protocols. Illinois Natural History Survey, Office of the Chief, Technical Report 2002-2, Champaign, IL. 38 pp, + Figures, Tables, and Appendix.

Tranel, MA. 2009. Research Proposal: establishment and maintenance of forbs in existing grass stands. *In* MW DonCarlos, RO Kimmel, JS Lawrence, and MS Lenarz, editors. Summaries of Wildlife Research Findings. Minnesota Department of Natural Resources, St. Paul, MN.

Whiles, MR and RE Charlton. 2005. The ecological significance of tallgrass prairie arthropods. Annual Review of Entomology 51:387-412.

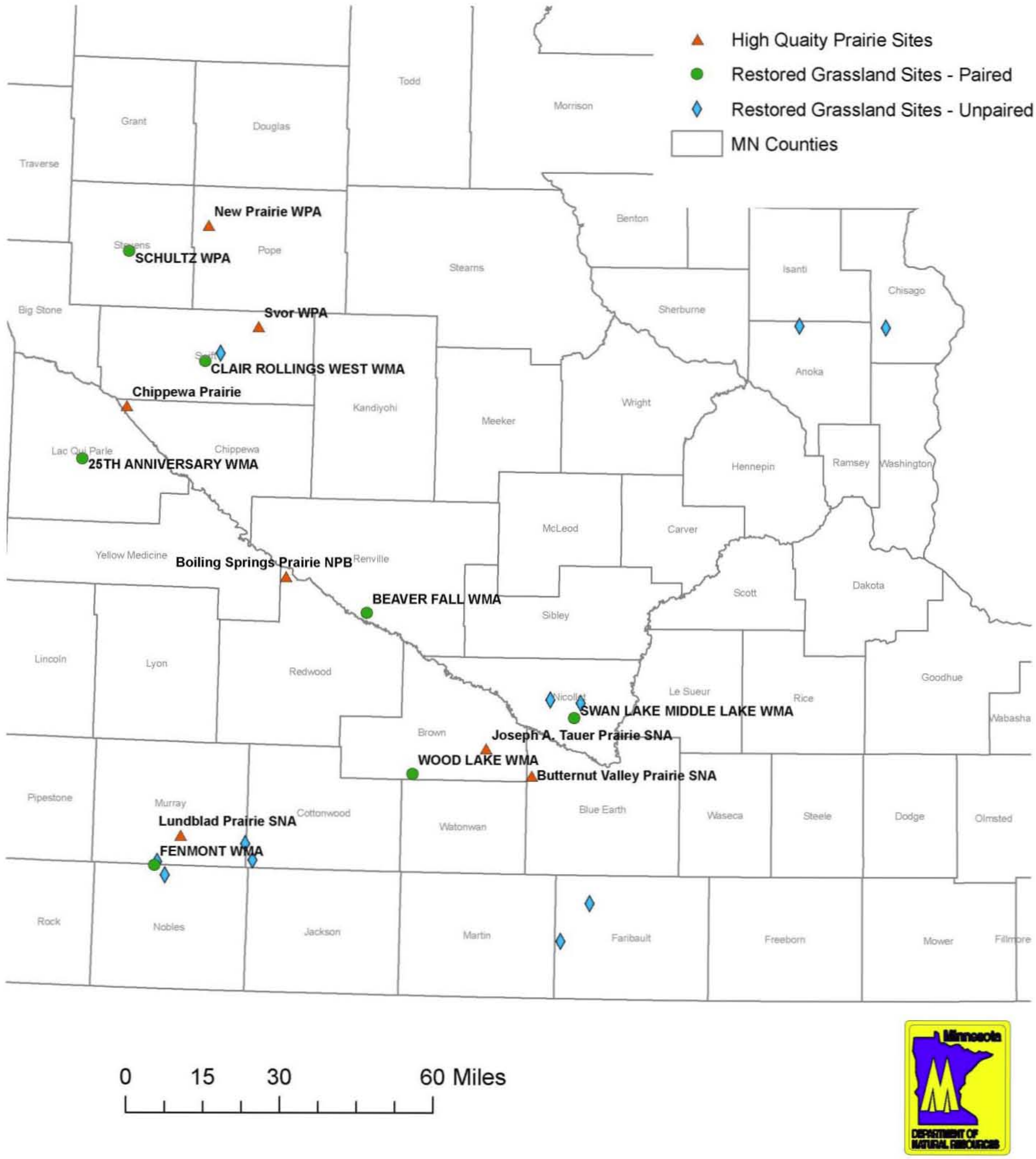


Figure 1. Potential invertebrate monitoring sites. Sites were paired to include 1 restored grassland site (green circle) and 1 high quality prairie site (orange triangle) with similar topographic and plant community characteristics near one another. Unpaired sites (blue diamond) will not be sampled. Southern MN, 2010.

Publications

The following is a list of scientific reports and other publications by personnel in the Wildlife Populations and Research Unit for the approximate period of March 2009 through February 2010.

Some titles by Unit personnel pertain to work done while employed By the MNDNR, while other titles are from work done elsewhere (e.g. as a graduate student, employed by another agency, while on leave of absence, etc.)

An asterisk (*) before an author's name indicates that the report was listed as in press or in review in previous publications of the Summaries of Wildlife Research Findings.

Included under scientific reports are those that haven't been published and those accepted for publication (in press).

Names in bold indicate a MNDNR employee.

Forest Wildlife Populations and Research Group Publications

*Baker, L. R., A. A. Tanimola, O. S. Olubode, and **D. L. Garshelis**. 2009. Distribution and abundance of sacred monkeys in Igboland, southern Nigeria. American Journal of Primatology 71: 71:574–586.

DelGiudice, G. D., B. A. Sampson, M. S. Lenarz, M. W. Schrage, and A. J. Edwards. 2010. Winter body condition of moose (*Alces alces*) in a declining population in northeastern Minnesota. Journal of Wildlife Diseases *Accepted*.

***Garshelis, D.L.** 2009. Family Ursidae (Bears). Pages 448–497 *in* D. E. Wilson and R. A. Mittermeier, editors. Handbook of the Mammals of the World. Vol. 1. Carnivores. Lynx Edicions. Barcelona, Spain. .

Hwang, M-H., **D. L. Garshelis**, Y-H. Wu, and Y. Wang. 2010. Home ranges of Asiatic black bears in the Central Mountains of Taiwan: gauging whether a reserve is big enough. Ursus: in press.

*Kochanny, C. O., **G. D. DelGiudice**, and **J. R. Fieberg**. 2009. Comparing Global Positioning System and Very High Frequency telemetry home ranges of white-tailed deer. Journal of Wildlife Management 73:73: 779-787.

Laske, T. G., H. J. Harlow, **D. L. Garshelis**, and P. Iaizzo. 2010. Extreme respiratory sinus arrhythmia enables hibernating black bear survival — Physiological insights and applications to human medicine. Journal of Cardiovascular Translational Research: in press.

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: in press.

*Liu, F, W. McShea, **D. L. Garshelis**, X. Zhu, D. Wang, J. Gong, and Y. Chen. 2009. Spatial distribution as a measure of conservation needs: an example with Asiatic black bears in south-western China. Diversity and Distributions: 15: 649–659.

Martin, D. J., B. R. McMillan, **J. D. Erb**, T. A. Gorman, and D. P. Walsh. 2010. Diel activity patterns of river otters (*Lontra Canadensis*) in southeastern Minnesota. Journal of Mammalogy Accepted.

*Packer, C., M. Kosmala, H. Cooley, H. Brink, L Pintea, **D. Garshelis**, G. Purchase, M. Strauss, A. Swanson, G. Balme, L. Hunter, and K. Nowell. 2009. Sport hunting, predator control and conservation of large carnivores. PLoSONE 4(6): e5941

Steinmetz, R. and **D. L. Garshelis**. 2010. Estimating ages of bear claw marks in Southeast Asian tropical forests as an aid to population monitoring. Ursus: in press

*In press last year.

<u>Wetland Wildlife Populations and Research Group Publications</u>

Berkman, L, K., M. J. Saltzgiver, E. J. Heist, C. K. Nielsen, **C. L. Roy**, and P. D. Scharine. 2009. Hybridization and polymorphic microsatellite markers for two lagomorph species (Genus Sylvilagus): implications for conservation. Conservation Genetics Resources 1:419-424.

Giudice, J. H., J. R. Fieberg, **M. C. Zicus, D. P. Rave,** and R. G. Wright. 2010. Cost and Precision Functions for Aerial Quadrat Surveys: a Case Study of Ring-Necked Ducks in Minnesota. Journal of Wildlife Management 74:in press.

Hanson, M. A., B. Palik, J. O Church, and A. T. Miller. In Review. Influences of Upland Timber Harvest on Aquatic Invertebrate Communities in Seasonal Ponds: Efficacy of Harvest Buffers. Ecology and management of Wetlands DOI 10.1007/s11273-009-9167-1.

Herwig, B.R., Zimmer, K.D., **Hanson, M.A.**, Konsti, M.L., Younk, J.A., Wright, R.W., Vaughn, S.R., and Haustein, M.D. 2010. Factors influencing fish distributions and community structure in shallow lakes within prairie and prairie-parkland regions of Minnesota, USA. Wetlands DOI 10.1007/s13157-010-0037-7.

Zimmer, K.D., **M.A. Hanson**, B.R. Herwig, and M.L. Konsti. 2009. Thresholds and stability of alternative regimes in shallow prairie-parkland lakes of central North America. Ecosystems 12:843-852.

Farmland Wildlife Populations and Research Group Publications

Carbaugh, J. S., D. L. Combs, and E. M. Dunton. 2010. Nest site selection and nesting ecology of giant Canada geese in central Tennessee. Human-Wildlife Interactions. In Press.

Cornicelli, L, and M. D. Grund. 2011. Don't throw the baby out with the bathwater: Non-random public input data can be useful. Human Dimensions of Wildlife 16: in review.

Drake, J. F., R. O. Kimmel, J. D. Smith, and G. Oejlert. 2009. Conservation reserve program grasslands and ring-necked pheasant abundance in Minnesota. Pages 302-314 in S.B. Cederbaum et al. eds. Gamebird 2006: Quail VI and Perdix XII. 31 May – 4 June, 2006. Warnell School of Forestry and Natural Resources, Athens, GA, USA.

Dunton, E. M., and D. L. Combs. 2010. Movements, habitat selection, associations, and survival of giant Canada goose broods in central Tennessee. Human-Wildlife Interactions. In press.

Fieberg, J., L. Cornicelli, D. C. Fulton, and M. D. Grund. 2010. Design and analysis of simple choice surveys for natural resource management. Journal of Wildlife Management 74: in press.

Grund, M. D., L. Cornicelli, L. T. Carlson, and E. Butler. 2010. Bullet fragmentation and lead deposition in white-tailed deer and domestic sheep. Human-Wildlife Interactions 4: in press.

Grund, M. D. 2010. Survival analysis and computer simulations of lethal and contraceptive management strategies for urban deer. Human-Wildlife Interactions 4: in press.

Mitchell, M. D., and R. O. Kimmel. 2009. Landowner attitudes and perceptions regarding wildlife benefits of the Conservation Reserve Program (CRP) in south central Minnesota. Rural Minnesota Journal 2009. Pages 93-105.

Mitchell, M.D., R.O. Kimmel, and J. Snyders. In review. Reintroduction and range expansion of eastern wild turkeys (*Meleagris gallopavo sylvestrs)* in Minnesota. Geographical Review.

Parent, C. J., B. J. Goodwin, and E. M. Dunton. 2010. Survival of wild turkey hens transplanted beyond their current distribution in Minnesota. Proceedings of the National Wild Turkey Symposium 10. In Press.

Restani, M., R.O. Kimmel, J.R. Fieberg, and S.L. Goetz. 2009. Effects of supplemental food and experience on winter survival of transplanted wild turkeys. Wilson Journal of Ornithology 121(2):366-377.

Tranel, M.A. and R.O. Kimmel. 2009. Impacts of lead ammunition on wildlife, the environment, and human health – a literature review and implications for Minnesota. *In* R,T, Watson, M. Fuller, M. Pokra, and W.G. Hunt (eds.) Ingestion of Spent Lead Ammunition: Implications for Wildlife and Humans. The Peregrine Fund, Boise, Idaho, USA.

Wildlife Biometrics Publications

Fieberg, J., J. Matthiopoulos, M. Hebblewhite, M.S. Boyce, J. L. Frair. In press. Correlation and studies of habitat selection: problem, red herring, or opportunity? *Philosophical Transactions of the Royal Society, Series B*.

Frair, J. L., **J. Fieberg**, M. Hebblewhite, F. Cagnacci, N. DeCesare, and L Pedrotti. In press. Resolving issues of imprecise and habitat-biased locations in ecological analyses using GPS telemetry data. *Philosophical Transactions of the Royal Society, Series B*.

Kie, J. G., Matthiopoulos, J., **Fieberg, J.**, Mitchell, M.S., Powell, R. A., Cagnacci, F., Gaillard, J-M., and P. Moorcroft. In Press. The home-range concept: are traditional estimators still relevant with modern telemetry technology? *Philosophical Transactions of the Royal Society, Series B*.

Fieberg, J. and **G. D. DelGiudice**. In press. Estimating age-specific hazards from wildlife telemetry data. *Journal of Environmental and Ecological Statistics*.

Fieberg, J., Cornicelli, L., Fulton, D. C., and **Grund, M. D**. 2010. Design and analysis of simple choice surveys for natural resource management. *Journal of Wildlife Management* 74:871-879.

Giudice, J. G., J. R. Fieberg, M. C. Zicus, D. P. Rave, and R. G. Wright. 2010. Constructing cost and precision functions for aerial quadrat surveys: a case of ring-necked ducks in Minnesota. *Journal of Wildlife Management* 74:342–349.

Fieberg, J., Rieger, R.H., **Zicus, M. C.,** Schildcrout, J. S. 2009. Regression modelling of correlated data in ecology: subject specific and population averaged response patterns. *Journal of Applied Ecology* 46:1018-1025.

Fieberg, J., and **G. D. DelGiudice**. 2009. What time is it? Choice of time origin and scale in extended proportional hazards models. *Ecology* 90:1687-1697.

Other Publications

DelGiudice, G. D. 2009. "Do wolf tracks and few deer on your fall hunting area mean what you think they mean?" Whitetales. Fall Issue: 12-14.