Summaries of Wildlife Research Findings

2011

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

SUMMARIES OF WILDLIFE RESEARCH FINDINGS
2011

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TABLE OF CONTENTS

Farmland Wildlife Research Group

Forest Wildlife Research Group

Wildlife Health Program

Biometrics Unit and Surveys

Wetland Wildlife Research Group

Farmland Wildlife Populations and Research Group

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HUNTER PERCEPTIONS AND ACCEPTANCE OF ALTERNATIVE DEER MANAGEMENT REGULATIONS[1]

Louis Cornicelli, David C. Fulton, Marrett D. Grund, and John Fieberg

ABSTRACT

Wildlife managers are often confronted with a policy paradox where a majority of the public support's an outcome, but there is no agreement on specific management strategies to achieve this outcome. Previous research has also reported a link between regulatory acceptance, hunter satisfaction, and hunter participation rates. Thus, human dimensions research aimed at understanding hunter motivations and behavior is needed for effective management. In 2005, we surveyed Minnesota (USA) deer hunters (n = 6,000; 59% response) to evaluate attitudes regarding alternative deer (*Odocoileus virginianus*) harvest regulations. We also conducted a series of forced choice experiments in which respondents were asked to select an option from a list of representative regulations that might be adopted to achieve a particular deer management goal. Specifically, we modeled 5 deer population scenarios ranging from low populations with high buck-harvest rates to populations 50% over goal density. Our results indicate that hunters preferred different regulations depending on the population scenario, but generally preferred antler-point restrictions and disliked limiting buck licenses through a lottery. We also found consistency among scenarios, in that a small percentage of respondents indicated they would not hunt if regulations were changed. The results from this study should help wildlife managers design deer harvest regulations that are both acceptable to hunters and achieve management objectives.

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[1] Wildlife Society Bulletin 35:323-329

ESTABLISHMENT OF FORBS IN EXISTING GRASS STANDS

Kurt Haroldson and Molly Tranel

SUMMARY OF FINDINGS

Interseeding native forbs into reconstructed grasslands could restore plant species diversity and improve wildlife habitat. Survival of forbs interseeded directly into existing vegetation may be enhanced by management treatments that reduce competition from established grasses. We evaluated the effects of two mowing and two herbicide treatments on diversity and abundance of forbs interseeded into established grasslands on 15 sites in southern Minnesota. Each site was burned and interseeded in fall 2009 or spring 2010, and two mowing treatments (once or twice per season) and two grass-selective herbicide treatments (high and low rate) were applied during the 2010 growing season. One year following treatments, 24 (83%) of the 29 native, seeded forbs were observed in the study plots, with no significant difference in seeded species abundance among treatments. Additional vegetation surveys will be conducted on all sites in the study in summers 2012-2013 to determine the extent of forb establishment and persistence.

INTRODUCTION

Minnesota Department of Natural Resources (MNDNR) wildlife managers indicated a need for more information on establishing and maintaining an abundance and diversity of forbs in grasslands (Tranel 2007). 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).

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 and McCain et al. 2010). Temporarily suppressing dominant grasses may increase light, moisture, and nutrient availability to seedling forbs, ultimately increasing forb abundance and diversity (Schmitt-McCain 2008 and McCain et al. 2010). Williams et al. (2007) found that frequent mowing of grasslands in the first growing season after interseeding increased forb emergence and reduced forb mortality. Additionally, Hitchmough and Paraskevopoulou (2008) found that forb density, biomass, and richness were greater in meadows where a grass herbicide was used.

In this study, the effects of two mowing and two herbicide treatments on diversity and abundance of forbs interseeded into established grasslands in southern Minnesota were investigated.

METHODS

Study sites (n=15) were distributed throughout the southern portion of Minnesota’s prairie/farmland region on state and federally owned wildlife areas. Each site was ≥4 ha and characterized by relatively uniform soils, hydrology, and vegetative composition. All sites were dominated by relatively uniform stands of native grasses with few forbs, most of which were non-native, such as sweet clover (*Melitotus alba, M. officinalis*).

Eight sites were burned in October-November 2009 and frost interseeded during December 2009 and March 2010, whereas seven sites were burned and interseeded during

April and May 2010. The same 30-species mix of seed was broadcast seeded at all sites at a rate of 239 pure live seeds/m^2. Seed used on spring-burned sites was cold-moist stratified for 3-5 weeks in wet sand to stimulate germination during spring 2010 and seed used on fall-burned sites was not.

Treatments

Sites were divided into 10 plots of approximately equal size and randomly assigned each of four treatments and the control. Each site received all treatments to account for variability among sites, and each treatment was replicated twice at each site. The following treatments, designed to suppress grass competition, were applied during the first growing season after interseeding (2010) while the forbs were becoming established:

- Mow 1: mowed once to a height of 10-15 cm when vegetation reached 25-35 cm in height.
- Mow 2: mowed twice to a height of 10-15 cm when vegetation reached 25-35 cm in height.
- Herbicide Low: applied grass herbicide Clethodim (Select Max®) at 108 mL/ha (9 oz/A) when vegetation reached 10-15 cm.
- Herbicide High: applied grass herbicide Clethodim (Select Max®) at 215 mL/ha (18 oz/A) when vegetation reached 10-15 cm.

Sampling Methods

Between 25 July 2011 and 27 September 2011, 20 sampling points randomly distributed within each study plot were located using a Global Positioning System receiver. Presence of seeded forbs was estimated in a 76 x 31 cm^2 quadrat at each sampling point. In addition, observers estimated litter depth and percent cover (Daubenmire 1959) of native grasses, exotic grasses, native forbs, exotic forbs, bare ground, and duff within each sampling quadrat. Percent cover was estimated within 6 classes: 0-5%, 5-25%, 25-50%, 50-75%, 75-95%, and 95-100%. Visual obstruction readings (VOR; Robel et al. 1970) were recorded in the 4 cardinal directions at the 5th and 20th quadrats in each plot.

RESULTS

One year following treatments, 24 (83%) of the 29 native, seeded forbs were observed in the study plots (Table 1). Black-eyed Susan (*Rudbeckia hirta*) was the most common seeded forb species (forming 40% of all seeded forb observations), followed by wild bergamot (*Monarda fistulosa*, 16%), golden Alexander (*Zizia aurea*, 10%), common milkweed (*Asclepias syriaca,* 8%), and yellow coneflower (*Ratibida pinnata*, 7%). Differences in seeded forb abundance were not significant among treatments and the control ($P > 0.05$; Table 1).

Native grasses formed the greatest component of canopy cover, averaging 48% cover across all treatments (Table 2). Big bluestem (*Andropogon gerardi*) tended to dominate the study plots, occurring in 82% of the quadrats regardless of treatment ($P > 0.05$). Cover of native grasses was slightly less in the Mow 2 treatment than the Mow 1 treatment. In contrast, cover of exotic grasses was slightly greater in the Mow 2 treatment than other treatments except Herbicide Low (Table 2). Treatments did not significantly affect cover of native forbs or exotic forbs (Table 2).

DISCUSSION

Although the mowing and herbicide treatments were effective in suppressing grasses during the first growing season after application (Tranel 2009), the grasses had recovered by 2011. Most of the seeded forb species became established in low numbers, but we detected no benefit of treatments in supporting greater forb establishment 1 year after interseeding. Williams et al. (2007) also observed similarly abundant seeded forbs in mowed and control treatments at the end of the second growing season, but seeded forbs were twice as abundant in mowed treatments by the beginning of year 5. Hitchmough and Paraskevopoulou (2008) found that, in treatments where grass was suppressed with a graminoid herbicide, sown forb density was higher in the second and third years after treatment and forb richness was greater three years after treatment. Additional vegetation surveys will be conducted on all sites in the study in summers 2012-2013 to determine the extent of forb establishment and persistence.

MANAGEMENT IMPLICATIONS

The use of the pre-emergent grass selective herbicide Clethodim (Select Max) at 108 mL/ha (9 oz/A) and 215 mL/ha (18 oz/A) was effective at suppressing well established native and exotic grasses at the pilot site (Tranel 2009). Growth of grass was stunted but grass mortality was not observed even at the high application rate at any of the study sites. Because this herbicide is fairly inexpensive and requires only one application in a growing season, it could prove to be a cost effective alternative to repeated mowing in areas where grass suppression is desired.

ACKNOWLEDGEMENTS

This project was funded by the Minnesota Department of Natural Resources. The Minnesota Department of Natural Resources and U.S. Fish and Wildlife Service managers provided study sites and equipment, and labor for treating the sites. J. Zajac suggested the idea behind this study. J. Fieburg and J. Giudice provided valuable advice and assistance on the study design and analysis. J. Swanson, A. Krenz, H. Rauenhorst, and K. Zajak conducted most of the field work. M. Grund provided comments on an earlier draft of this report.

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Table 1. Frequency of seeded forb species by treatment type on 15 study sites during 2011 (1 year post treatment). Maximum possible frequency was 3,000 (15 sites x 5 treatments x 2 replicates x 20 quadrats).

Seeded Forb	Control		Mow 1		Mow 2		Herbicide Low		Herbicide High		Sum	% of Total
Alumroot	0	0	0	0	0	0	0	0	0	2	2	0.12
Aster, Heath	2	1	0	8	13	1	0	7	9	0	41	2.39
Aster, New England	1	1	0	1	0	1	0	1	1	0	6	0.35
Aster, Sky Blue	0	1	0	0	0	0	0	0	0	0	1	0.06
Bergamot, Wild	28	29	25	22	29	30	22	35	37	26	283	16.47
Black Eyed Susan	68	59	54	74	81	59	61	92	68	75	691	40.22
Blazingstar, Prairie	0	0	1	0	0	0	1	0	0	0	2	0.12
Blazingstar, Rough	0	0	0	0	0	0	0	0	0	0	0	0.00
Canada Milk Vetch	6	3	5	2	4	6	7	5	5	7	50	2.91
Closed Bottle Gentain	0	0	0	0	0	1	0	0	0	0	1	0.06
Coneflower, N. L. Purple	0	1	0	2	1	7	1	0	2	1	15	0.87
Coneflower, Yellow	11	10	13	8	17	19	7	7	14	18	124	7.22
Culver's Root	0	0	0	0	0	0	0	0	0	0	0	0.00
False Sunflower	0	1	1	3	1	2	0	0	1	3	12	0.70
G. Alexander, Heart Leaf	0	1	0	0	0	0	0	0	1	1	3	0.17
Golden Alexander	16	15	21	27	22	14	2	20	23	13	173	10.07
Goldenrod, Stiff	1	3	0	3	1	0	0	3	0	3	14	0.81
Leadplant	0	0	0	0	0	0	0	0	0	0	0	0.00
Maximilian Sunflower	0	0	0	0	0	0	0	0	0	2	2	0.12
Milkweed, Common	18	17	11	8	11	19	17	9	14	13	137	7.97
Partridge Pea	0	0	0	0	1	0	1	2	0	3	7	0.41
Prairie Cinquefoil	10	3	7	7	5	6	4	4	10	9	65	3.78
Prairie Clover, Purple	1	0	2	2	1	0	2	1	1	1	11	0.64
Prairie Clover, White	0	0	1	1	0	0	0	1	1	2	6	0.35
Prairie Coreopsis	0	0	0	0	0	0	0	0	0	0	0	0.00
Prairie Onion	0	0	0	0	0	0	0	0	0	0	0	0.00
Showy Tick Trefoil	0	0	1	0	1	0	0	0	1	0	3	0.17
Vervain, Blue	9	2	2	9	3	8	2	2	3	5	45	2.62
Vervain, Hoary	2	0	3	3	3	1	2	2	6	2	24	1.40
Sum	173	147	147	180	194	174	129	191	197	186	1718	100.00

Table 2. Comparison of estimated percent cover of native grasses, exotic grasses, native forbs, and exotic forbs on 15 study sites during 2011 (1 year post treatment).

	Native Grasses			Exotic Grasses			Native Forbs			Exotic Forbs		
Treatment	Mean	SD	95% CI	Mean	SD	95% CI	Mean	SD	95% CI	Mean	SD	95% CI
Control	49.08	27.81	46.85-51.31	31.19	33.08	28.54-33.84	21.62	31.97	19.06-24.18	21.25	30.89	18.78-23.72
Mow 1	50.49	27.43	48.30-52.68	33.21	33.45	30.53-35.89	21.48	31.45	18.96-24.00	19.27	26.75	17.13-21.41
Mow 2	45.62	29.40	43.27-47.97	39.35	35.07	36.54-42.16	21.26	32.30	18.68-23.84	20.78	28.77	18.48-23.08
Herbicide high	48.11	27.32	45.92-50.30	31.11	33.26	28.45-33.77	24.98	31.98	22.42-27.54	18.19	24.41	16.24-20.14
Herbicide low	47.63	27.72	45.41-49.85	36.42	35.07	33.61-39.23	22.37	32.23	19.79-24.95	18.40	28.58	16.11-20.69
All	48.12			34.04			22.34			19.58		

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ECOLOGY AND POPULATION DYNAMICS OF BLACK BEARS IN MINNESOTA

David L. Garshelis, Karen V. Noyce, and Mark A. Ditmer[1]

SUMMARY OF FINDINGS

During April 2011–March 2012, we monitored 38 radiocollared black bears (*Ursus americanus)* at 4 study sites representing contrasting portions of the bear's geographic range in Minnesota: Voyageurs National Park (VNP, northern extreme), Chippewa National Forest (CNF; central), Camp Ripley (southern fringe), and a site at the northwestern (NW) edge of the range. Most of the focus of this study has been in the NW site in recent years. Hunting has been the primary source of mortality in all areas, but with a concerted effort to discourage hunters from shooting collared bears, and by clearly marking bears with large ear tags, only 1 collared bear was known to be killed by a Minnesota bear hunter in fall 2011. However, the radiocollared sample was diminished by other human-caused mortalities and radiocollar failures. Reproduction was highest in the NW study site. Data from Global Positioning System (GPS)-radiocollars indicated that males in the NW made significant use of crop fields (corn and sunflowers) from August to October. By contrast, females in this area rarely used crops, but instead spent much of their time in aspen woodlands and shrublands. Thus, the high reproductive output of females is not due to the crops, but to an abundance of natural food in the generally small woodlots that are scattered across this area (on both public and private lands). Analysis of stable isotopes in hair samples were useful in determining the reliance of individual bears on different key foods, especially distinguishing those that fed largely on corn or sunflowers, the 2 main crops consumed by bears in this area. Continuation of this work will aim to predict the extent to which bears can continue expanding westward.

INTRODUCTION

The Minnesota Department of Natural Resources (MNDNR) initiated research on black bears in 1981, spurred by concerns of low population size and over-harvest. This occurred commensurate with a management program designed to restrict the harvest via a quota on bear hunting licenses. For the first 10 years, the bear study was limited to the Chippewa National Forest (CNF), near the geographic center of the Minnesota bear range (Figure 1). Later, we started satellite bear projects in other study sites with different habitat conditions. Each of these began as a graduate student project, supported in part by the MNDNR. After completion of these student projects, we continued studies of bears at Camp Ripley Military Reserve, near the southern fringe of the Minnesota bear range, and in Voyageurs National Park (VNP), on the Canadian border (Figure 1).

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

In 2007 we initiated work in a fourth study site at the northwestern edge of the Minnesota bear range (henceforth NW; Figure 1). This area differs from the other 3 areas in a number of respects: (1) it is largely agricultural (including crop fields, like corn and sunflowers, that bears consume), (2) most of the land, including various small woodlots, is privately-owned, with some

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larger blocks of forest contained within MDNR Wildlife Management Areas (WMAs) and a National Wildlife Refuge (NWR); (3) the bear range in this area appears to be expanding and bear numbers have been increasing, whereas most other parts of the bear range are stable or declining in bear numbers; and (4) hunting pressure in this area is unregulated (it is within the no-quota zone, so there is no restriction on numbers of hunting licenses, and each hunter is allowed to kill 2 bears).

OBJECTIVES

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

METHODS

We attached radiocollars with breakaway and/or expandable devices to bears either when they were captured during the summer or when they were handled as yearlings in the den with their radiocollared mother. We trapped bears this year only in the NW study site, using barrel traps baited with raw bacon. Traps were checked daily, mainly by local volunteer residents. We anesthetized captured bears with Telazol and fitted them with GPS-collars, programmed to collect locations at designated intervals (varying from 6 hr to 20 min, depending on time of year). We used both GPS “pods” (Telemetry Solutions, Concord, CA) that were bolted onto standard VHF collars, and also GPS-Iridium collars (Vectronic Aerospace, Berlin, Germany). The latter collars uploaded location data to an Iridium satellite, which was then transmitted to us daily by email. The location data stored in the pods were retrievable only by physically connecting the pod to a computer; typically, we took pods off bears when they were denning.

During December–March, we visited all radio-instrumented bears once or twice at their den site. We immobilized bears in dens with an intramuscular injection of Telazol, administered with a jab stick or Dan-Inject dart gun. Bears were then removed from the den for processing. We measured lengths and girths, body weight, and biolelectrical impedance (to calculate percent body fat), and took blood and hair samples. We changed or refit the collar, and attached a first collar on some yearlings. All collared bears were given brightly-colored, cattle-size ear tags (7x6 cm; Dalton Ltd., UK) that would be plainly visible to hunters. Additionally, collaborators from the University of Minnesota (Dr. Paul Iaizzo) and Medtronic, Inc. (Dr. Tim Laske) measured heart condition with a 12-lead EKG and ultrasound on a select sample of bears, and implanted (subcutaneously) a miniature heart monitoring device (developed for humans: Reveal®, Medtronic, Inc, Minneapolis, MN) to record heart rate, body temperature, and activity throughout the year. Bears were returned to their dens after processing.

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

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

We plotted GPS locations downloaded from collars on bears in the NW study site. We used a Geographic Information System (GIS) overlay to categorize the covertypes of GPS locations, including types of crop fields.

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

RESULTS AND DISCUSSION

Radiocollaring and Monitoring

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

Mortality

Legal hunting has been the dominant cause of mortality among radiocollared bears from all study sites: since 1981, 77% of mortalities that we observed were due, or likely due to hunting (Table 2). In earlier years of this study, hunters were encouraged to treat collared bears as they would any other bear so that the mortality rate of collared bears would be representative of the population at large. With fewer collared bears left in the study, and the focus shifted to reproduction and habitat use rather than mortality, beginning in 2001 we sought to protect the remaining sample of bears by asking hunters not to shoot them and marking them with easily-visible ear tags (Figure 2).

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

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

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

Reproduction

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

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

Habitat and Crop Use by NW Bears

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

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

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

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

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

FUTURE DIRECTIONS

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

ACKNOWLEDGMENTS

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

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

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

[a] Due to premature failure of breakaway link.

[b] Due to radiocollar failure, unreported kill, or long-distance movement.

[c] Likely shot before denning.

Table 2. Causes of mortality of radiocollared black bears ≥1 year old in 4 Minnesota study sites, 1981–2011. Bears did not necessarily die in the area where they usually lived (e.g., hunting was not permitted within Camp Ripley or VNP, but bears were killed by hunters when they traveled outside these areas).

	CNF	Camp Ripley	VNP	NW	All combined
Shot by hunter	223	11	15	12	261
Likely shot by hunter[a]	8	1	0	4	13
Shot as nuisance	22	2	1	3	28
Vehicle collision	12	8	1	2	23
Other human-caused death	9	1	0	0	10
Natural mortality	7	3	4	0	14
Died from unknown causes	4	2	0	3	9
Total deaths	**285**	**28**	**21**	**24**	**358**

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

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

Year	Litters checked	No. of cubs	Mean cubs/litter	% Male cubs	Mortality after 1 yr[a]
1982	4	12	3.0	67%	25%
1983	7	17	2.4	65%	15%
1984	6	16	2.7	80%	0%
1985	9	22	2.4	38%	31%
1986	11	27	2.5	48%	17%
1987	5	15	3.0	40%	8%
1988	15	37	2.5	65%	10%
1989	9	22	2.4	59%	0%
1990	10	23	2.3	52%	20%
1991	8	20	2.5	45%	25%
1992	10	25	2.5	48%	25%
1993	9	23	2.6	57%	19%
1994	7	17	2.4	41%	29%
1995	13	38	2.9	47%	14%
1996	5	12	2.4	25%	25%
1997	9	27	3.0	48%	23%
1998	2	6	3.0	67%	0%
1999	7	15	2.1	47%	9%
2000	2	6	3.0	50%	17%
2001	5	17	3.4	76%	15%
2002	0	0	—	—	—
2003	4	9	2.3	22%	0%
2004	5	13	2.6	46%	33%
2005	6	18	3.0	33%	28%
2006	2	6	3.0	83%	33%
2007	2	6	3.0	67%	17%
2008	1	3	3.0	100%	33%
2009	1	3	3.0	33%	33%
2010	1	4	4.0	100%	50%
2011	1	4	4.0	25%	50%
2012	1	3	3.0	67%	
Overall	**177**	**466**	**2.6**	**52%**	**19%**

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

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

Year	Litters checked	No. of cubs	Mean cubs/litter	% Male cubs	Mortality after 1 yr[a]
1992	1	3	3.0	67%	0%
1993	3	7	2.3	57%	43%
1994	1	1	1.0	100%	—
1995	1	2	2.0	50%	0%
1996	0	0	—	—	—
1997	1	3	3.0	100%	33%
1998	0	0	—	—	—
1999	2	5	2.5	60%	20%
2000	1	2	2.0	0%	0%
2001	1	3	3.0	0%	33%
2002	0	0	—	—	—
2003	3	8	2.7	63%	33%
2004	1	2	2.0	50%	—
2005	3	6	2.0	33%	33%
2006	2	5	2.5	60%	—
2007	3	7	2.3	43%	0%
2008	2	5	2.5	60%	0%
2009	3	7	2.3	29%	29%
2010	2	4	2.0	75%	25%
2011	3	8	2.7	50%	25%
2012	1	2	2.0	100%	
Overall	**34**	**80**	**2.4**	**53%**	**22%**

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

Table 5. Black bear cubs examined in dens in Voyageurs National Park during March, 1999–2012. All adult collared females were killed by hunters in fall 2007, so no reproductive data were obtained during 2008–2009.

Year	Litters checked	No. of cubs	Mean cubs/litter	% Male cubs	Mortality after 1 yr[a]
1999	5	8	1.6	63%	20%
2000	2	5	2.5	60%	80%
2001	3	4	1.3	50%	75%
2002	0		—	—	—
2003	5	13	2.6	54%	8%
2004	0		—	—	—
2005	5	13	2.6	46%	20%
2006	1	2	2.0	50%	0%
2007	3	9	3.0	44%	—
2008	0				
2009	0				
2010	1	2	2.0	50%	0%
2011	1	2	2.0	0%	0%
2012	1	2	2.0	0%	
Overall	**27**	**60**	**2.2**	**48%**	**26%**

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

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

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

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

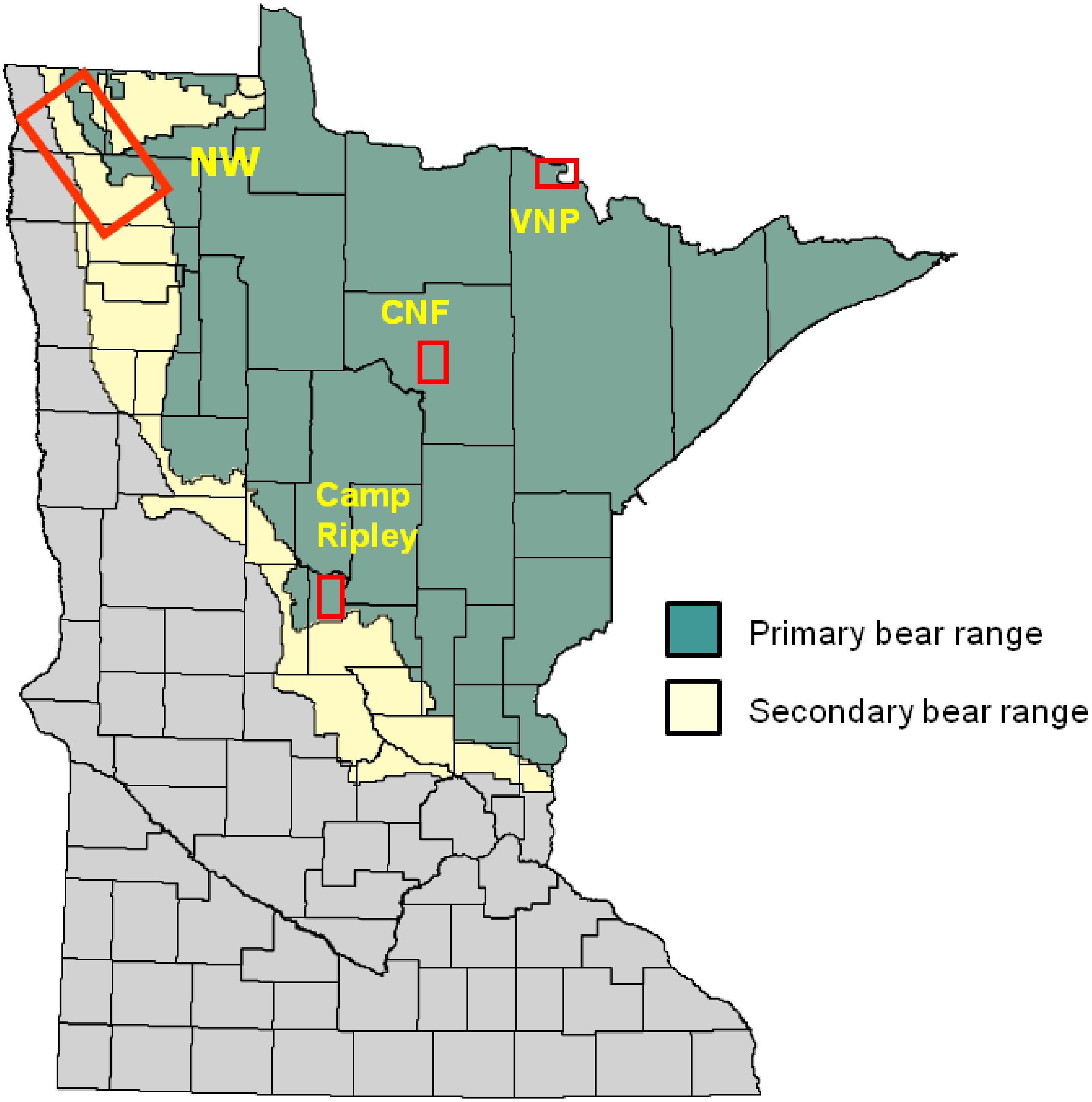

Figure 1. Location of 4 study sites within Minnesota's bear range: CNF (Chippewa National Forest, central bear range; 1981–2012); VNP (Voyageurs National Park, northern fringe of range; 1997–2012); Camp Ripley Military Reserve (near southern edge of range; 1991–2012); NW (northwestern fringe of range;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

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

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

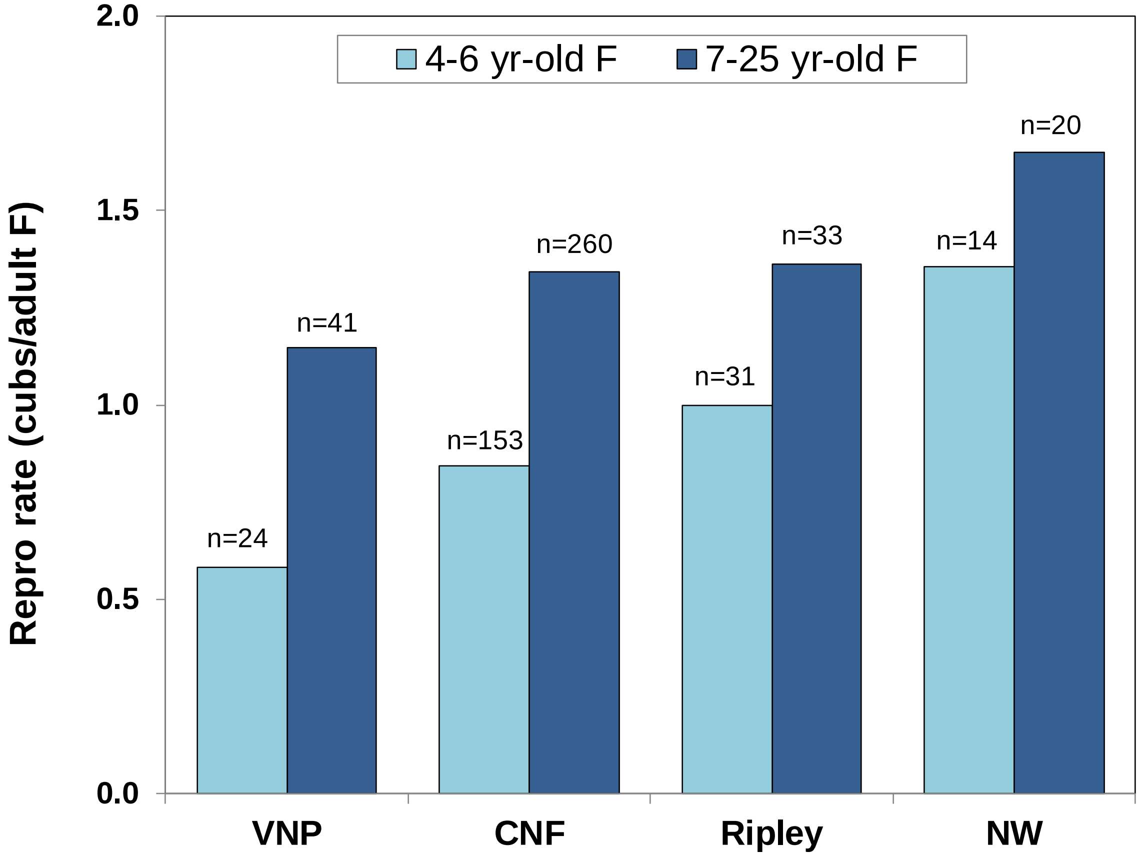

Figure 4. Reproductive rates of radiocollared bears within 4 study sites (see Figure 1) through March 2012. Sample sizes refer to the number of female bear-years of monitoring in each area for each age group. Data include only litters that survived 1 year (even if some cubs in the litter died).

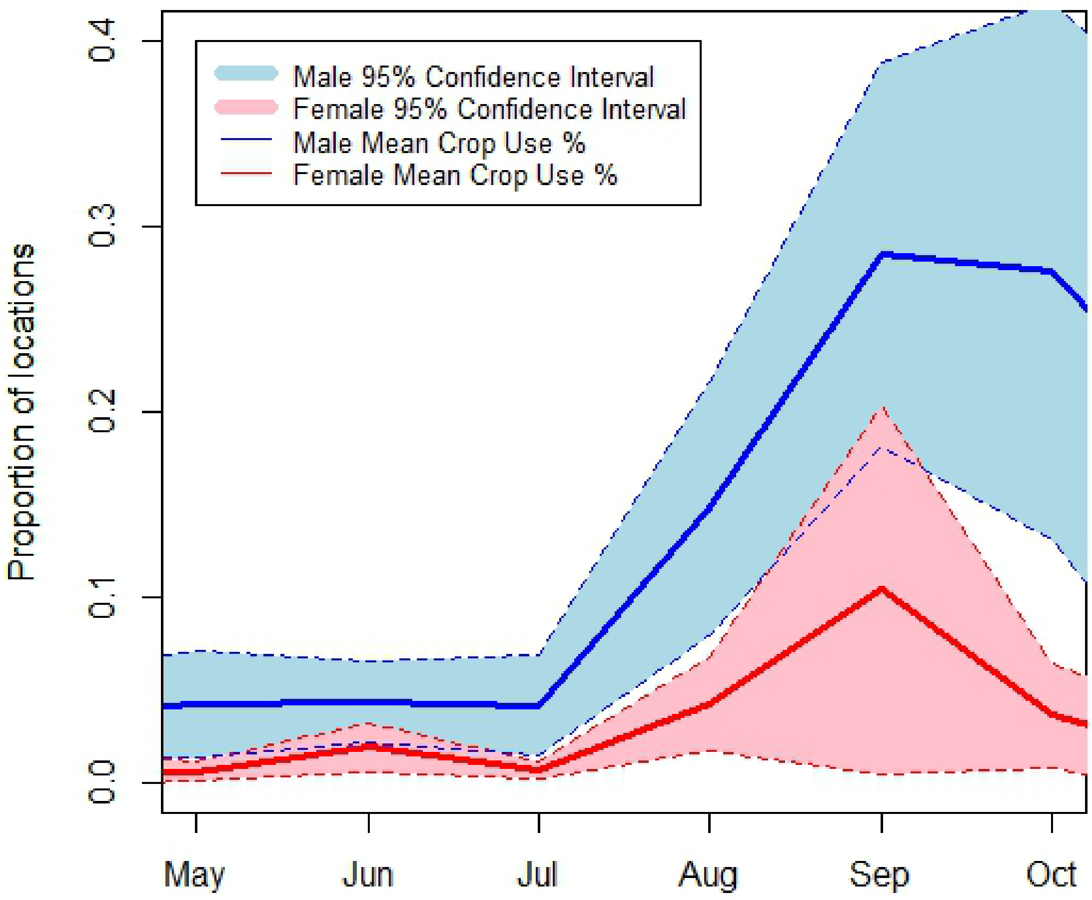

Figure 5. Proportion of GPS locations of radiocollared male and female black bears in NW Minnesota that were in crop fields each month, 2007–2011.

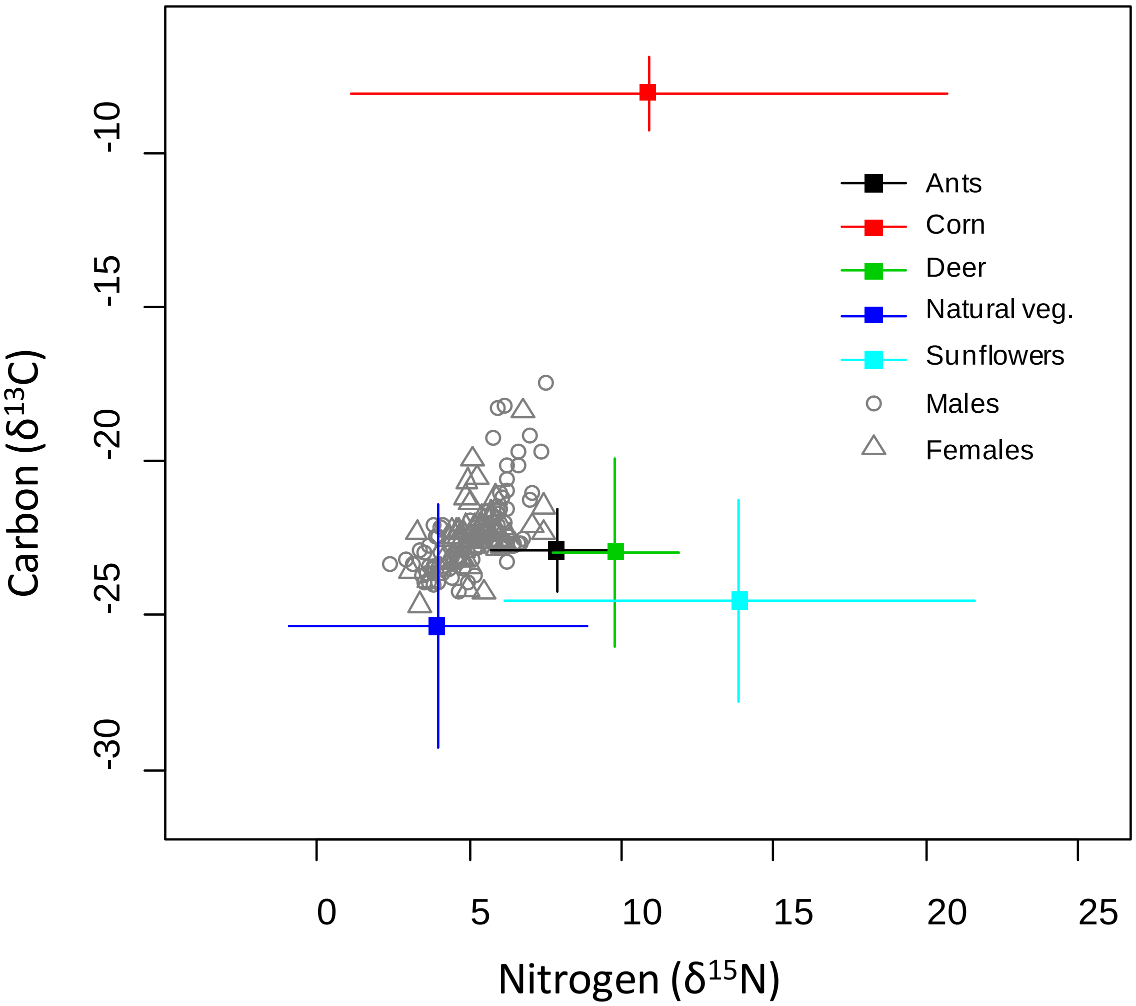

Figure 6. Stable isotope signatures obtained from hair samples of collared and hunter-killed black bears in NW Minnesota, 2007–2011 (*n* = 141) compared to mean isotope signatures (and 95%CI) of 5 types of bear foods that separated out using stable isotopes of C and N. Full hair samples of bears were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

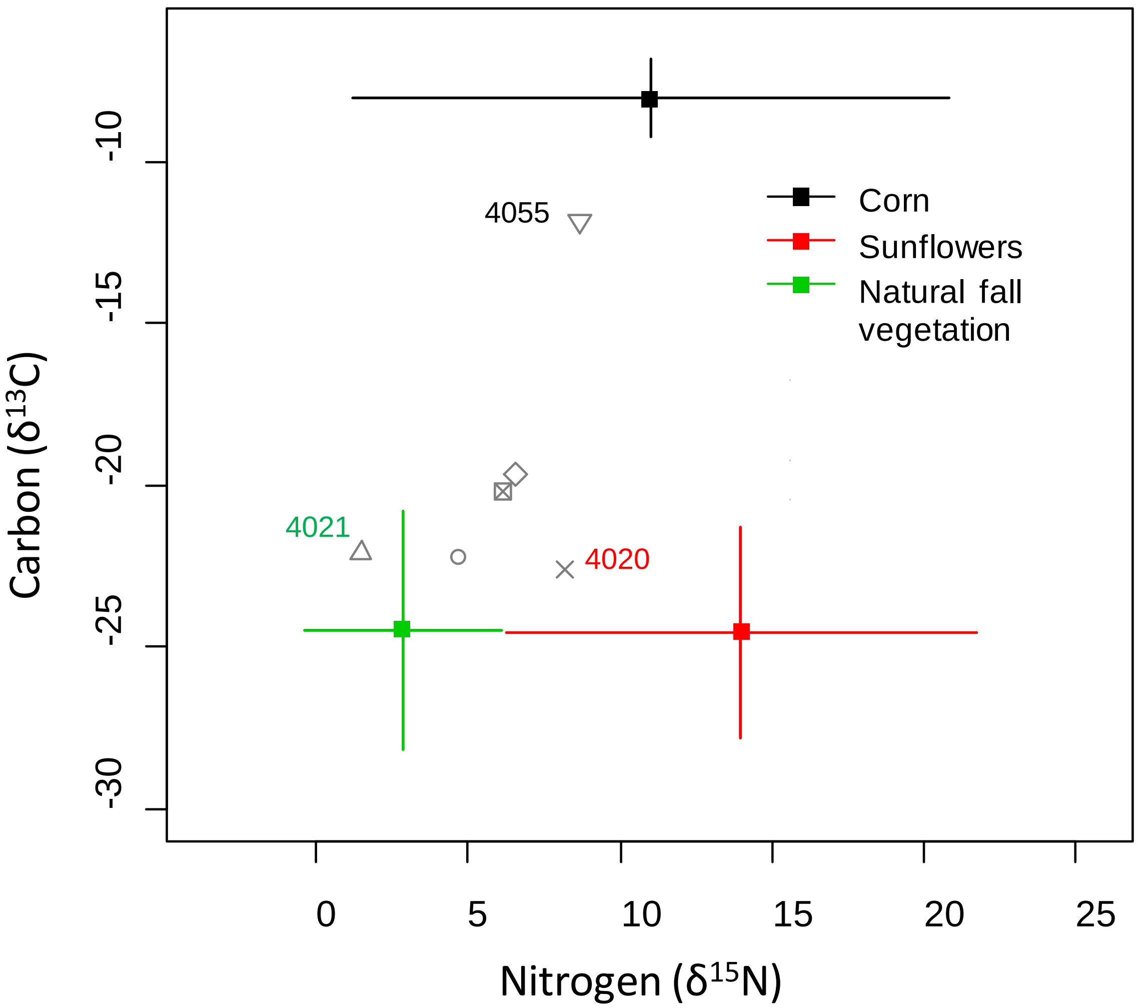

Figure 7. Stable isotope signatures of 6 GPS-collared bears in NW Minnesota during fall (based on only the proximal third of hair samples plucked in winter) matched against mean isotope signatures (and 95%CI) of 3 types of fall foods. Identification numbers of 3 individuals are highlighted who were located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

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

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

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

SUMMARY OF FINDINGS

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

INTRODUCTION

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

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

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

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

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

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

OBJECTIVES

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

METHODS

Study Area

Our study included 4 winter range sites located along the southeastern boundary of the Chippewa National Forest in north-central Minnesota, USA ($46^o52'$-$47^o15'$N and $93^o45'$-$94^o07'$W). The Willow (Wil), Inguadona (Ing), Shingle Mill (Shi), and Dirty Nose (Dir) sites were 20, 24, 23, and 13 km^2, respectively. The uplands were dominated by deciduous and mixed deciduous-conifer stands, whereas northern white cedar (*Thuja occidentalis*), black spruce (*Picea mariana*), balsam fir, and tamarack (*Larix laricina*) were most apparent on the lowlands (Doenier et al. 1997). The winter diet of deer on the 4 sites was highly diverse (about 36 browse species), but beaked hazel (*Corylus cornuta*), mountain maple (*Acer spicatum*), and red-osier dogwood (*Cornus sericea*) accounted for 82 and 89% of species browsed during mild and severe winters, respectively (DelGiudice et al. 2010).

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

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

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

Deer Capture, Handling, and Monitoring

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

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

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

Habitat Composition of Sites

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

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

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

Data Analyses

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

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

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

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

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

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

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

RESULTS

VHF Data

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

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

GPS Data

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

DISCUSSION

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

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

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

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

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

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

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

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

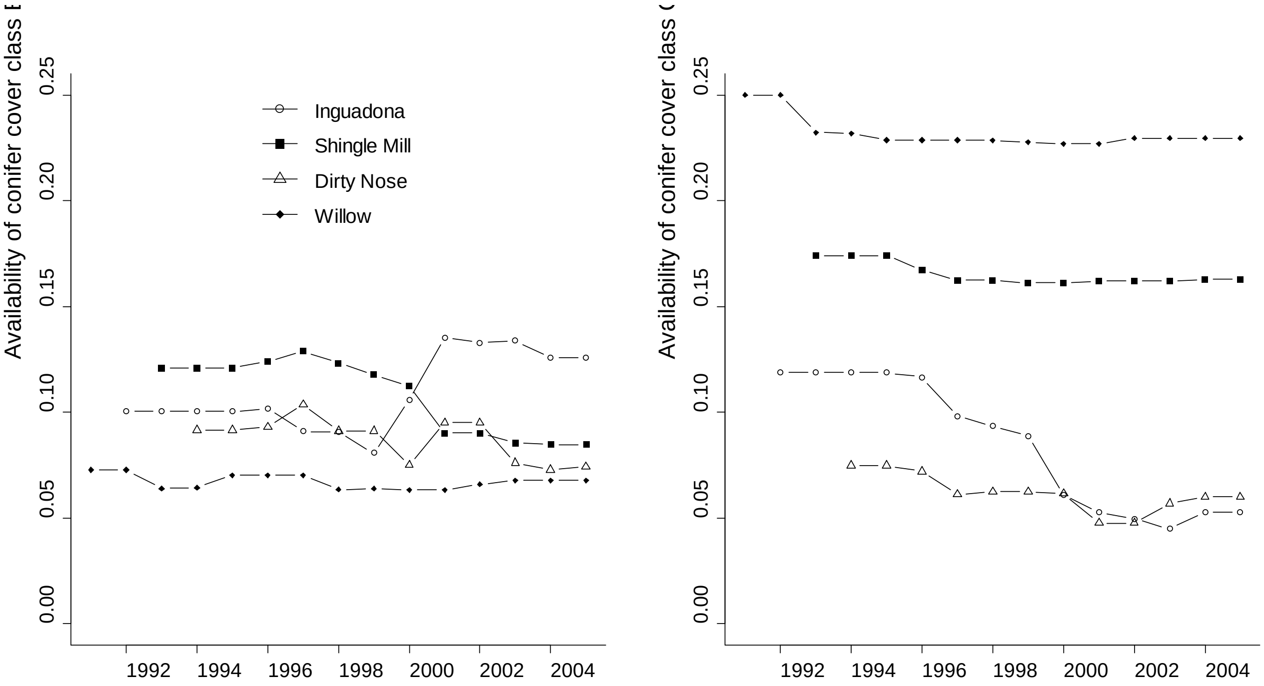

Figure 1. Proportional availability (versus time) of conifer cover classes B (40% ≤ *x* < 70% canopy closure, left panel) and C (≥70% canopy closure, right panel) for each of 4 study sites, north-central Minnesota, 1991-2005. First-year baseline was dependent on the year the site was incorporated into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

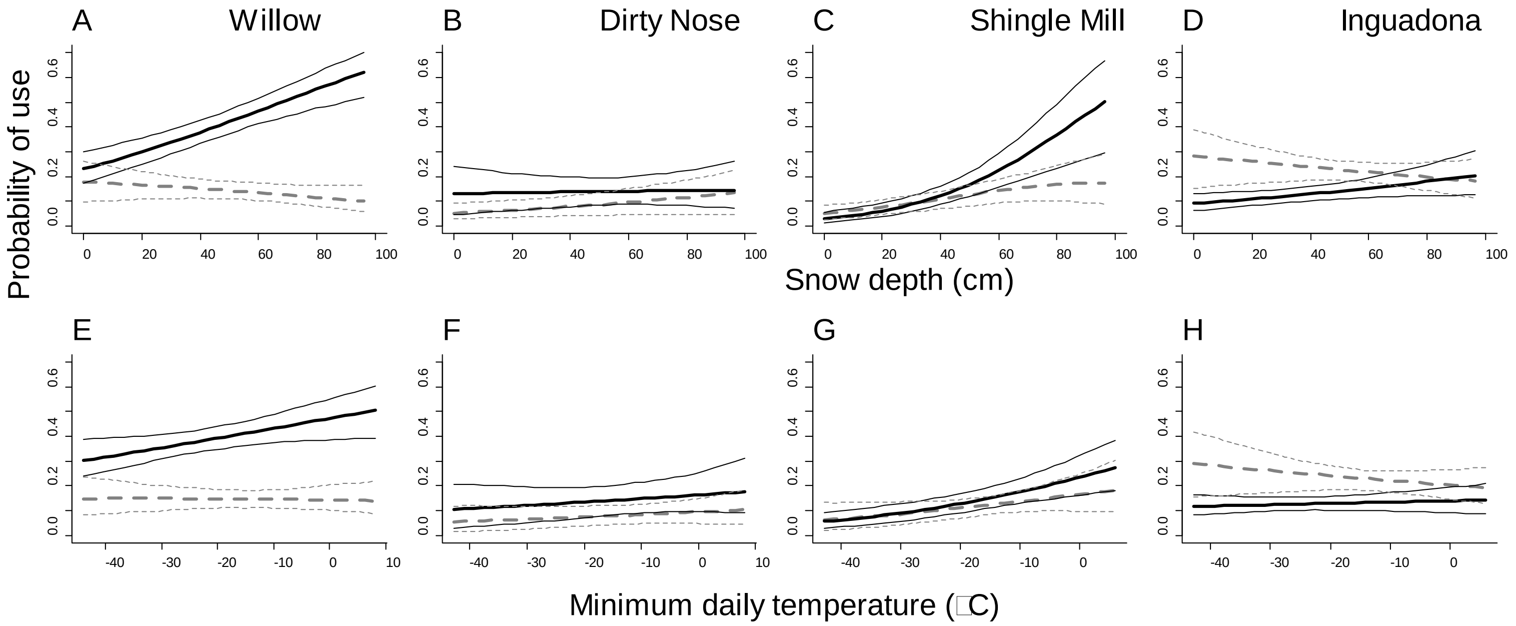

Figure 2. Model-based predicted probabilities of adult (≥1.5 years old), female white-tailed deer using conifer class C (≥70% canopy closure, dotted gray line) and conifer class B (40% ≤ *x* < 70% canopy closure, solid black line) during daytime hours (i.e., 0730-1700 hr) as a function of snow depth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

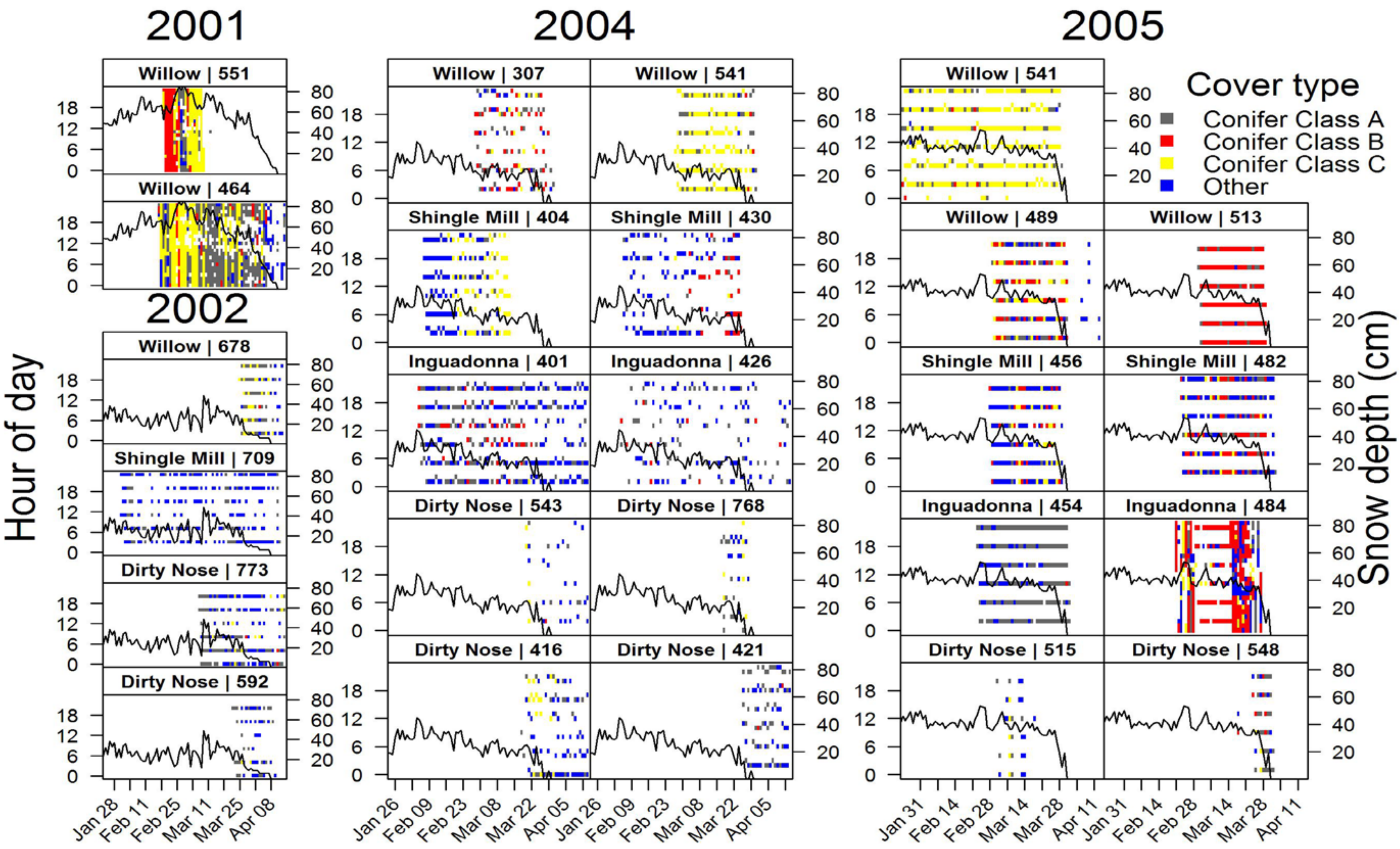

Figure 3. Date-time plots illustrating use of conifer classes A (< 40% canopy closure), B (40% ≤ *x* < 70% canopy closure), C (≥70% canopy closure) and “other” (open and hardwood types) by adult (≥1.5 years old), female white-tailed deer monitored using GPS collars collecting locations hourly or every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

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

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

SUMMARY OF FINDINGS

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

INTRODUCTION

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

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

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

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

OBJECTIVES

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

STUDY AREA

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

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

METHODS

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

Data Analyses

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

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

RESULTS AND DISCUSSION

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

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

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

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

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

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

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

ACKNOWLEDGMENTS

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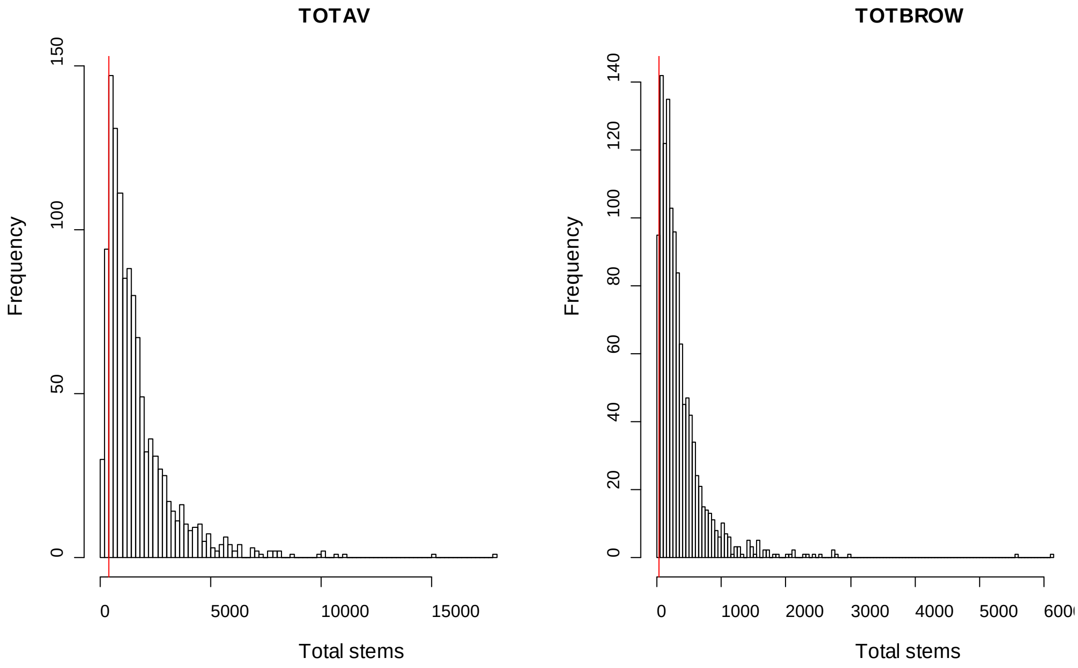


Figure 1. Density histograms of total browse stems available (left, TOTAV) and used (right, TOTBROW) by white-tailed deer on 4 study sites, north-central Minnesota, mid-January to March 1992-2005. Browse availability and use were not monitored during winter 1998-1999.

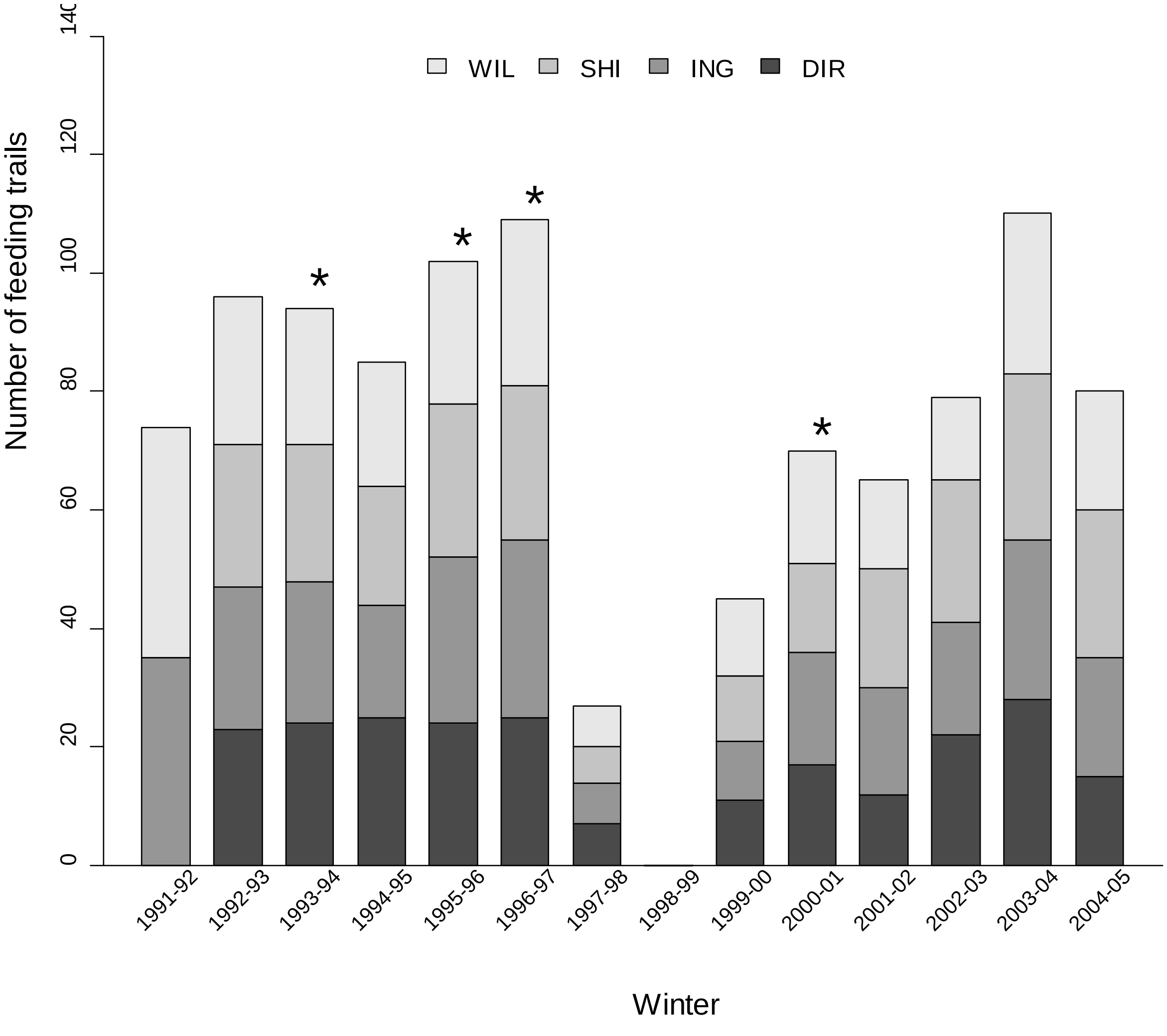


Figure 2. Distribution of total sample sizes (feeding trails with ≥400 browse stems available and ≥30 stems browsed) among years and 4 sites (WIL, SHI, ING, DIR) in a study of winter food habits of white-tailed deer, north-central Minnesota, winters 1991-1992 to 2004-2005. Browse availability and use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

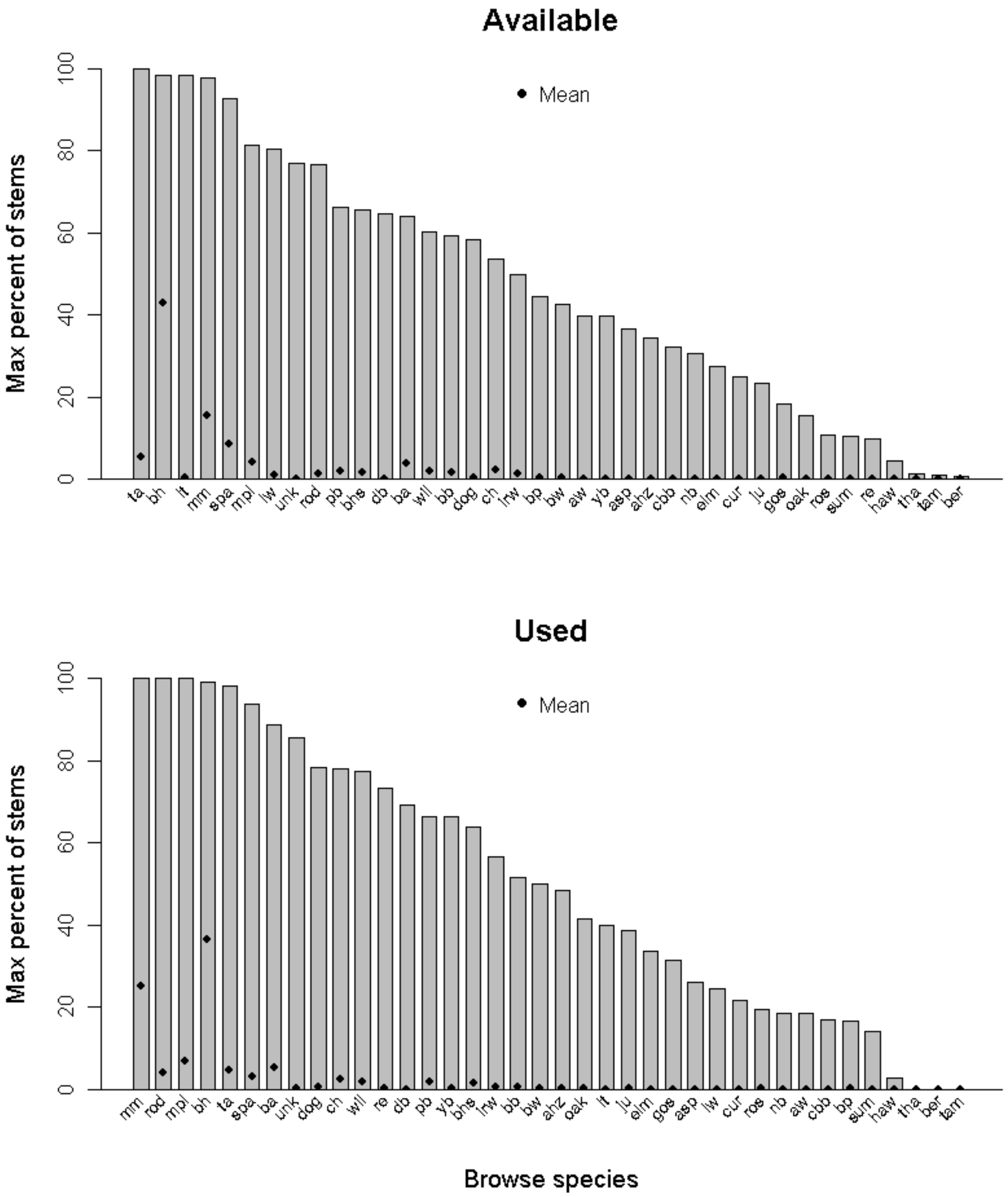


Figure 3. Mean and maximum percent of browse stems (by species) available and used by white-tailed deer on 4 sites (WIL, SHI, ING, DIR), north-central Minnesota, winters 1991-1992 to 2004-2005. Browse availability and use were not monitored during winter 1998-1999. (ta = trembling aspen, bh = beaked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

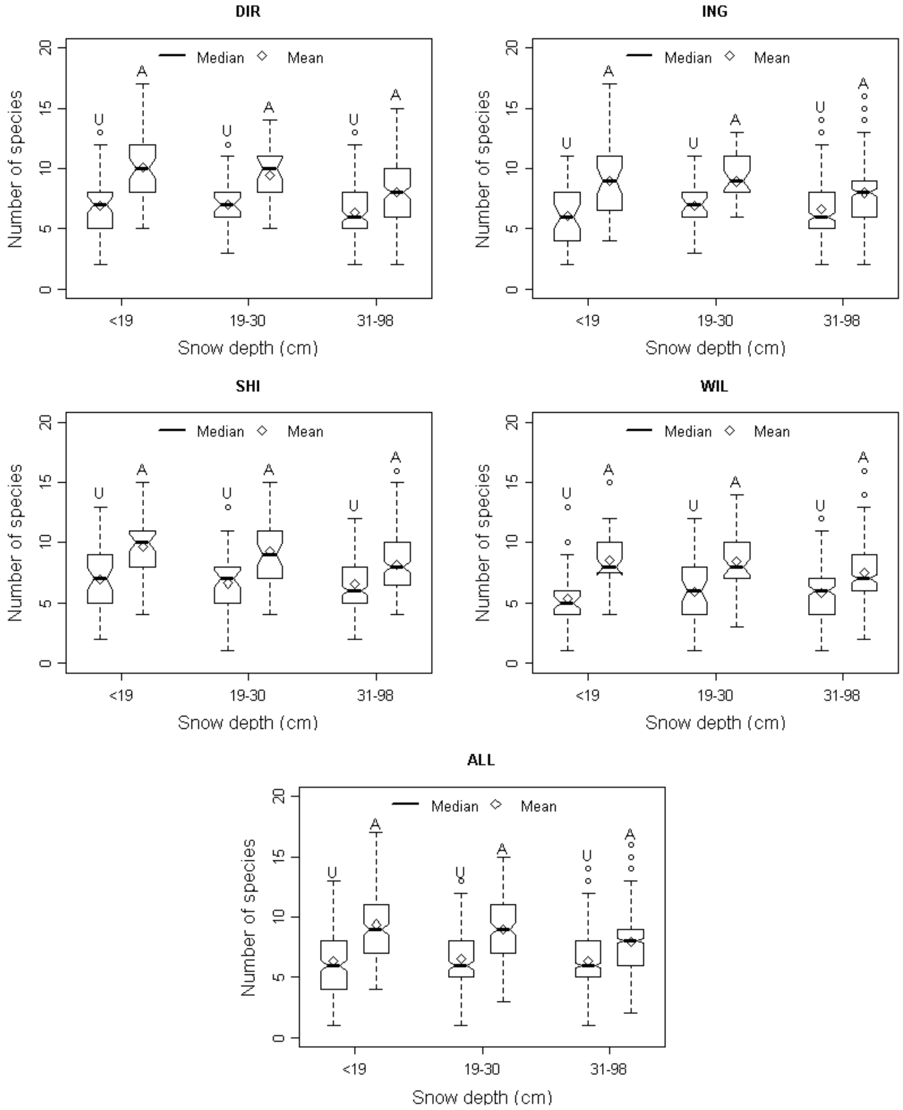


Figure 4. Mean and median number of browse species used (U) and available (A) along feeding trails of white-tailed deer when snow cover was shallow (< 19 cm), moderate (19-30 cm), and deep (31-98 cm) on 4 sites (WIL, SHI, ING, DIR), north-central Minnesota, winters 1991-1992 to 2004-2005. Browse availability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

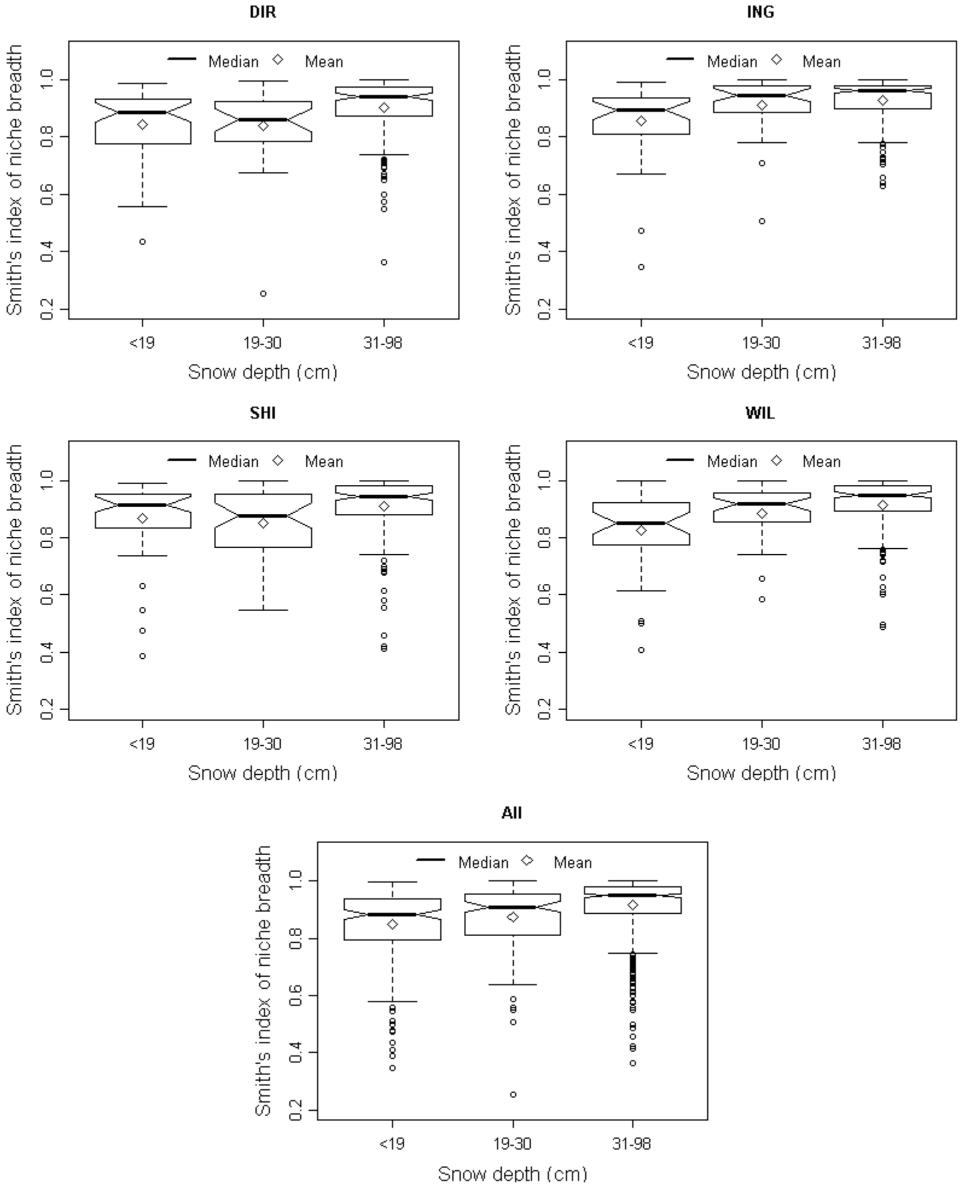


Figure 5. Median and mean niche breadth (Smith’s index of heterogeneity, 1 = max diversity) of winter browse diet of white-tailed deer when snow cover was shallow (< 19 cm), moderate (19-30 cm), and deep (31-98 cm) on 4 sites (WIL, SHI, ING, DIR), north-central Minnesota, winters 1991-1992 to 2004-2005.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

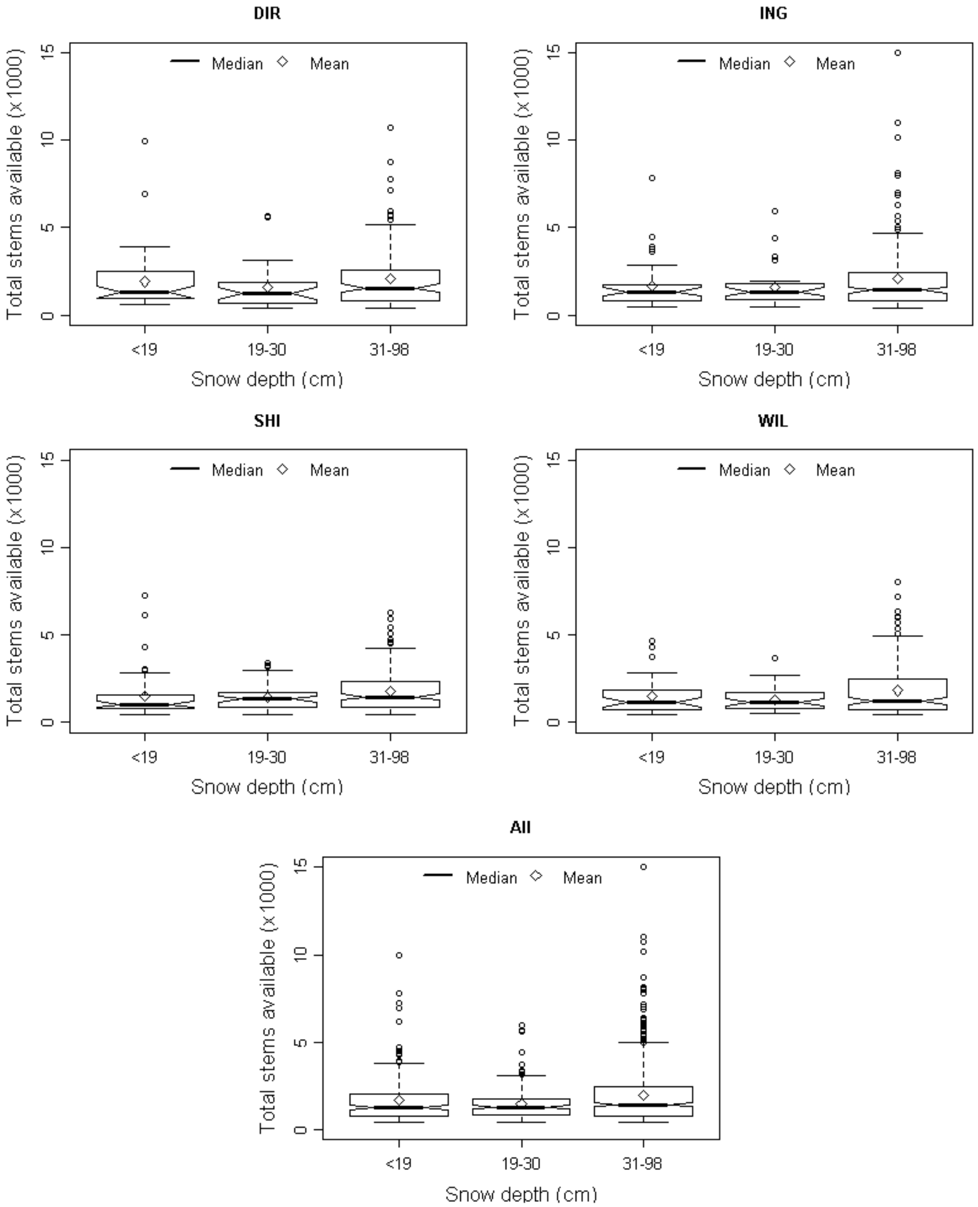
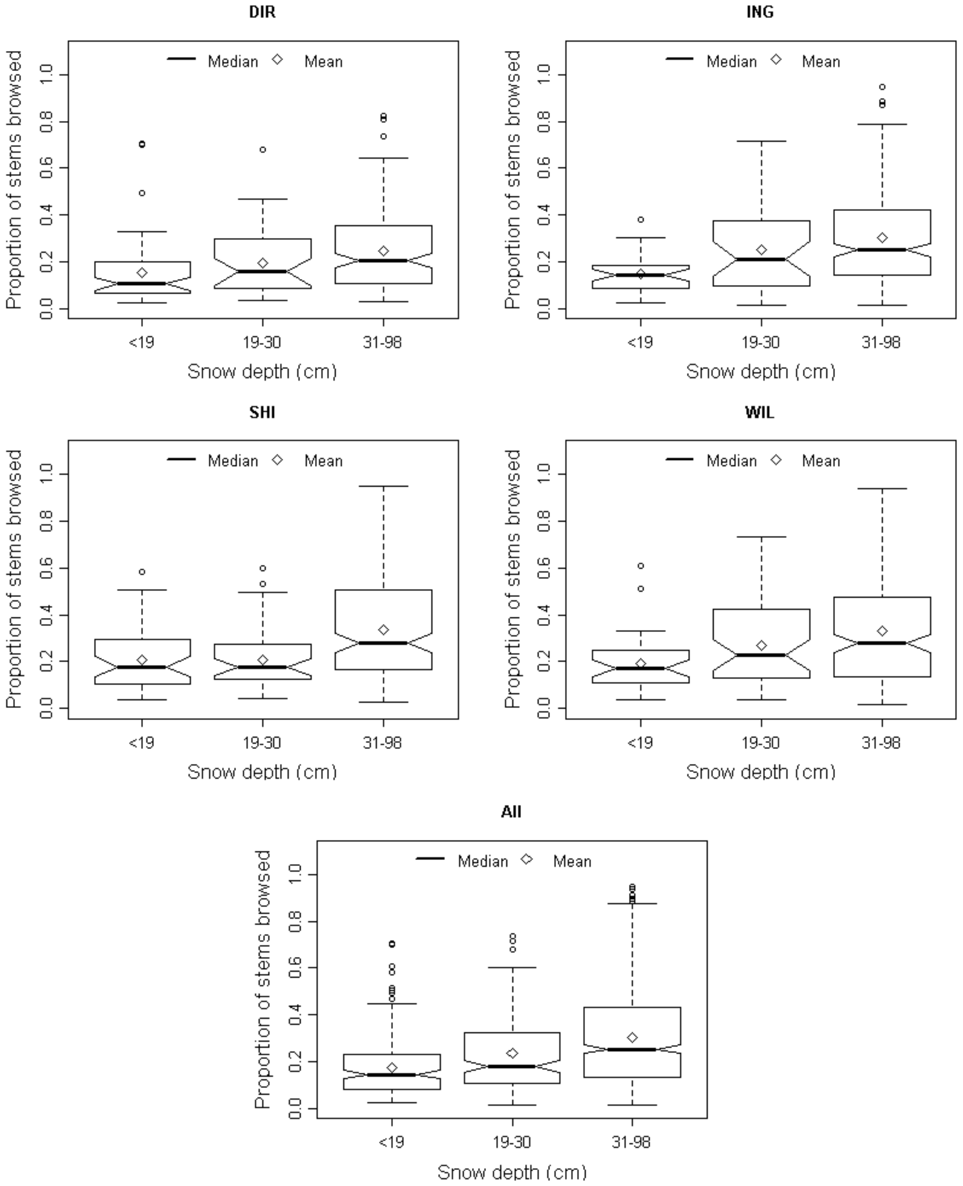


Figure 6. Median and mean abundance of browse stems available along feeding trails of white-tailed deer when snow cover was shallow (< 19 cm), moderate (19-30 cm), and deep (31-98 cm) on 4 sites (WIL, SHI, ING, DIR), north-central Minnesota, winters 1991-1992 to 2004-2005. Browse availability and use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



DIR

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

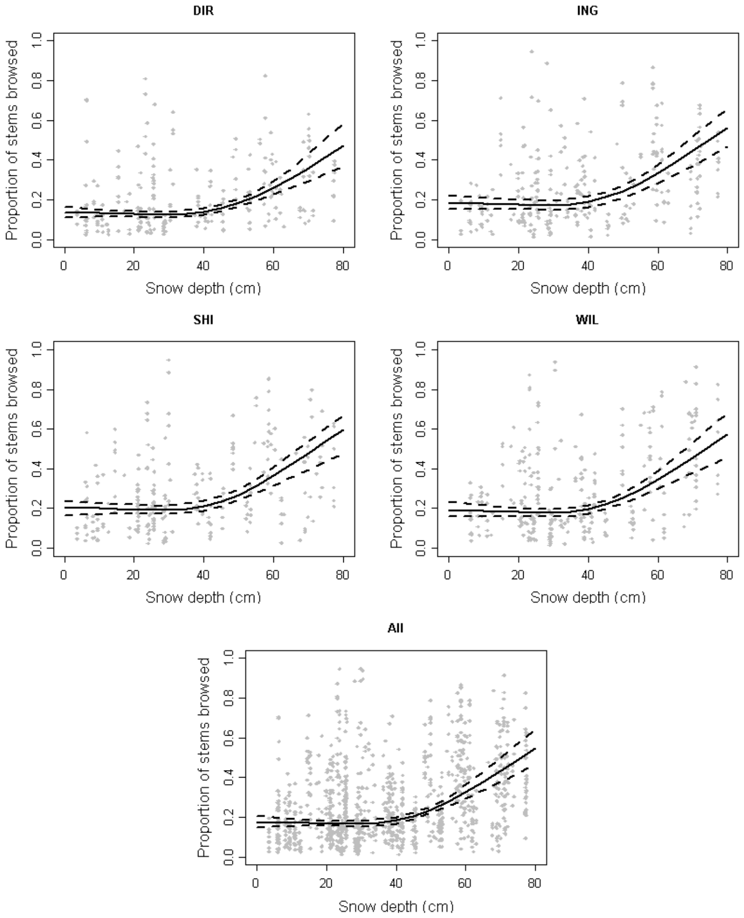

Figure 8. Regression relationship of the proportion of total browse stems used by white-tailed deer versus snow depth on 4 sites (WIL, SHI, ING, DIR), north-central Minnesota, winters 1991-1992 to 2004-2005. Browse availability and use were not monitored during winter 1998-1999. All regression coefficients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

REPRODUCTIVE ECOLOGY OF FISHER AND MARTEN IN MINNESOTA

John Erb, Pam Coy, and Barry Sampson

SUMMARY OF FINDINGS

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

INTRODUCTION

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

Legally, fisher and marten were unprotected in Minnesota prior to 1917, after which harvest season length restrictions were implemented. These protections were removed in the mid-1920s, and remained so until all harvest was prohibited in 1929. Seasons remained closed until 1977 for fisher and 1985 for marten, when limited harvests were reinstated. While harvest is legal in approximately the northern 50% of the state, most marten harvest occurs in counties bordering Canada, particularly in northeast and north-central Minnesota. Fisher harvest occurs in most of the northern 50% of the state, though harvest is comparatively low in extreme northeast Minnesota (Lake and Cook counties), and spatially variable, though generally increasing, in the Red River Valley (western Minnesota) and the highly fragmented transitional forests in central Minnesota. Peak harvest levels have been near 4,000 and 3,500 for marten and fisher, respectively. However, due to apparent multi-year population declines for both species, harvest seasons from 2007 - 2011 were reduced from 16 days to 9 days, and starting in 2010, the fisher harvest limit was reduced from 5 to 2. During this 5-year period, harvests have averaged ~2,100 and 1,400 for marten and fisher, respectively.

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

Martes pregnancy rate and litter size data are generally quantified from 1 of 4 methods: counts of corpora lutea (CL) in ovaries; counts of blastocysts (BC) in uteri; placental scar (PS) counts; or direct observation of litter size (Gilbert 1987, Mead 1994). Assuming both species are induced ovulators (but see Cherepak and Connor 1992, Frost et al. 1997), CL counts should accurately reflect copulation and ovulation rates, but all CL persist even if only 1 ovum is fertilized. Blastocyst counts reflect the number of fertilized ova, but not all BC may implant in the uterus and develop, and BC are often destroyed in poorly preserved carcasses. Hence, these 2 measures may not only overestimate litter size for parous females, but may also overestimate parturition rate (i.e., females may ovulate, 1 or more ova become fertilized, yet they fail to ultimately den and give birth). Placental scars, formed last in the reproductive process, would seem the most reliable carcass-based estimate of parturition rate and litter size. However, several authors (Gilbert 1987, Payne 1982, Strickland and Douglas 1987) have suggested that PS may not always persist long enough in mustelids to be detected during the harvest season when carcasses are easily collected, and PS can persist in some species even if fetuses are resorbed (Conaway 1955), and may be affected by observer variability (Johnson et al. 1995). Nevertheless, PS have been reliably used in the past (e.g., Coulter 1966, Crowley et al. 1990), though others have noted that reliable results may only be obtainable when doing microscopic analysis of fresh and properly preserved/prepared uteri (Mead 1994, Frost et al. 1999).

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

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

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

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

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

STUDY AREA

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

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

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

METHODS

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

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

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

During the pilot year, we deployed several radio-collar designs on fisher, including an ATS M1585 zip-tie collar (~43 g), an ATS M1930 collar (~38 g), and a Lotek SMRC-3 collar (~ 61 g; deployed on adult males only). Since the pilot year, we have primarily deployed ATS M1940 (~43 g) or Sirtrack TVC-162 collars (~45 g) on fisher. The majority of martens have been fitted with Holohil MI-2 collars (~31 g). We retrofitted each collar with a temperature data logger to provide ancillary information on winter activity and spring den attendance patterns, as well as to provide information on time of death for other study objectives.

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

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

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

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

Prey sampling transects have also been established in both study areas. Prey sampling is being conducted primarily to document between-area differences in prey abundance, annual within-area fluctuations in prey, and ultimately to assess whether fisher or marten habitat use, diet, survival, or reproductive success is correlated with prey dynamics. Prey-sampling transects ($n \approx 125$ 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 protocols used on an existing small mammal survey in Minnesota (Aarhus-Ward 2009). During both spring (hare pellet sampling) and fall (small mammal trapping), we will also count the number of red squirrels (*Tamiasciurus hudsonicus*) observed or heard along each transect. Rather than using 10-min point counts (e.g., Mattson and Reinhart 1996, Bayne and Hobson 2000) with our small mammal/hare pellet stations as the sampling points, we will simply record the number of unique squirrels detected along each transect while checking pellet plots and small mammal traps. Information on white-tailed deer and ruffed grouse (*Bonasa umbellus*) populations may be available from existing surveys or population models.

RESULTS AND DISCUSSION

Including the pilot year of the study, a total of 169 martens (80F, 89M) and 80 fishers (44F, 36M) have been radio-collared. Herein we provide a basic summary of data collected to date on den use chronology, den structures, pregnancy status, and litter size. Because tooth aging has not yet been completed for all animals, some of which may be only 1 year of age (i.e., not capable of producing kits), we present results only for animals known to be ≥2 years of age during spring den visits, or those of unknown age but for which we have confirmed parturition at the time of this writing (i.e., until age is known, we do not include animals that we have confirmed to be nulliparous). As of this writing, spring 2012 reproductive status assessment is largely complete for radio-collared fishers, but still ongoing for many martens.

Treating females that were alive during multiple parturition periods (years) as independent units, and excluding females known to be 1 year of age, a total of 31 female martens have been available for monitoring during the kit-rearing season. However, to date we have only confirmed age and reproductive status for 18 female martens, 10 of which we have also confirmed litter size (Table 1). Pooling years, pregnancy rate has been ~43% for 2-year-olds, and ~90% for martens 3 years or older. Because it has been comparatively difficult to inspect marten natal dens with video equipment, we have had to rely more on remote cameras to obtain litter information when kits are moved by the female, or when they are older and more mobile. Hence, many estimates of marten litter size are reported as minimums. Acknowledging this, average size of 10 litters confirmed to date is 2.9 (Table 1). Based on initial data, it appears marten kits are typically born in mid- to late-April. Given the timing of our marten capture (blood-drawing) operations (i.e., mid-December through early February), preliminary results indicate that marten progesterone levels have not sufficiently elevated in pregnant animals at that time to allow us to confirm mid-winter pregnancy status.

A total of 18 marten natal or maternal dens have been located to date (Table 2). Based on 14 dens confirmed prior to June 1 of each year, 64% have been in tree cavities, whereas 36% have been in underground burrows (Table 2). We have confirmed only 5 maternal dens used after June 1, and 4 were in underground burrows situated in rock-laden soils with the remaining one in above- and below-ground structure created from an exposed root mass (Table 2). Most female marten appear to move their kits from their natal den to 1 or more maternal dens in the first 6 weeks following birth.

Similar to marten, we treat female fishers that were alive during multiple parturition periods (years) as independent units. Excluding individuals known to be 1 year of age during the parturition period, we have confirmed both age and reproductive status for 36 female fishers (Table 3). Pooling all female fishers that produced a litter that we were able to count (n=27), average litter size is 2.6 (range = 1–4). Age-specific sample sizes are small (Table 3), but there is some indication that average litter size for 2-year-olds is lower than for older females (~2.1 versus 2.9). There is also some indication that birth rates are lower for 2-year-olds compared to older females. Parturition rate for known 2-year-olds (n=17) is 53% whereas parturition rate for female fishers known to be ≥3 years of age (n=17) is 94% (Table 3).

Based on data collected to date, it appears fisher kits are typically born in mid- to late-March, or ~1 month earlier than marten kits. Perhaps owing to earlier parturition, as well as apparently longer active gestation (Powell et al. 2003), it does appear that fisher progesterone levels are sufficiently elevated in pregnant females at the time of our winter capture operations (i.e., mid-Dec. through mid-March) to allow accurate assessment of mid-winter pregnancy status using seasonal hormone profiles developed in Maine (Frost et al. 1999). Furthermore, although blood has not been collected on all females, all those confirmed pregnant in mid-winter (progesterone) produced litters in spring, suggesting that overwinter disruption of pregnancy is not common.

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

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

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

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

Table 2. Natal and maternal den structures used by radio-collared female marten in Minnesota.

ID	Year	Confirmed	Den type	Den structure	Den details
M09-254	2010	4/19	natal	tree cavity	15.9" dbh live red maple
M09-237	2010	4/19	natal	tree cavity	16.8" dbh live tamarack
M08-140	2009	4/21	natal	underground burrow	rock-laden soil
M09-280	2010	4/28	natal	underground burrow	rock-laden soil
M08-140	2008	4/30	natal	underground burrow	rock-laden soil
M12-016	2012	5/1	natal	tree cavity	live cedar
M12-014	2012	5/7	natal	tree cavity	live cedar
M09-286	2010	5/7	natal	tree cavity	21.5" dbh live cedar
M12-024	2012	5/9	natal	tree cavity	14.6” dbh black ash
M09-262	2010	5/10	natal	tree cavity	18.8" dbh live cedar
M09-286	2009	5/19	natal	tree cavity	16.1" dbh live cedar
M09-286	2010	5/19	maternal	tree cavity	18.6” dbh live cedar
M09-286	2009	5/22	maternal	tree cavity	20.9" dbh live cedar
M09-254	2010	5/26	maternal	underground burrow	rock-laden soil
M09-286	2010	6/12	maternal	underground burrow	rock-laden soil
M08-140	2009	7/6	maternal	underground burrow	base of snag, rocky soil
M09-286	2009	7/9	maternal	underground burrow	along roots; base of cedar
M09-254	2010	7/12	maternal	underground burrow	rock-laden soil

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

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

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

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

ID	Year	Confirmed	Den type	Den structure	Den details
F10-507	2011	3/9	natal	tree cavity	16.9” dbh aspen snag?
F09-394	2012	3/13	natal	tree cavity	24" dbh live red oak
F08-353	2011	3/16	natal	tree cavity	16.4" dbh live aspen
F10-503	2011	3/16	natal	tree cavity	26.1” dbh aspen snag
F08-353	2012	3/16	natal	tree cavity	live aspen
F09-394	2012	3/22	maternal	tree cavity	live aspen
F08-353	2010	3/24	natal	tree cavity	15.1" dbh live aspen
F09-394	2010	3/26	natal	tree cavity	24.9” dbh live aspen
F10-507	2010	3/26	natal	tree cavity	25.6” dbh live oak
F10-507	2012	3/26	natal	tree cavity	20.3" dbh live aspen
F12-073	2012	3/26	natal	tree cavity	22.2" dbh live aspen
F12-054	2012	3/26	natal	tree cavity	16.7" dbh live aspen
F11-316	2012	3/27	natal	tree cavity	26.2" dbh live aspen
F09-394	2011	3/28	natal	tree cavity	15.8” dbh live aspen
F11-008	2012	3/28	natal	tree cavity	live sugar maple
F10-501	2011	3/30	natal	tree cavity	20.3” dbh live cedar
F11-316	2011	3/31	natal	tree cavity	13.6” dbh aspen snag
F11-052	2011	4/1	natal	tree cavity	23.8" dbh live aspen
F08-375	2009	4/7	natal	tree cavity	21.9” dbh w. pine snag
F08-353	2009	4/8	natal	tree cavity	23.2" dbh live aspen
F09-360	2009	4/8	natal	tree cavity	15.3” dbh aspen snag
F10-503	2011	4/8	maternal	tree cavity	18” dbh live red maple
F09-394	2009	4/9	natal	tree cavity	13.8” dbh aspen snag
F09-394	2010	4/9	maternal	tree cavity	22.1” dbh live aspen
F09-394	2011	4/9	maternal	tree cavity	24” dbh live aspen
F09-461	2010	4/11	natal	tree cavity	18.3" dbh live oak
F08-353	2011	4/11	maternal	tree cavity	19.0" dbh live aspen
F10-507	2011	4/12	maternal	tree cavity	15.1” dbh live oak
F10-507	2012	4/12	maternal	tree cavity	aspen
F10-507	2010	4/13	maternal	tree cavity	22.1” dbh aspen snag
F09-380	2009	4/14	natal	tree cavity	23.6” dbh aspen snag
F11-316	2011	4/14	maternal	tree cavity	16.6” dbh live oak
F09-370	2009	4/15	natal	tree cavity	23.5” dbh aspen snag
F09-394	2009	4/18	maternal	tree cavity	21.5” dbh live aspen
F12-073	2012	4/18	maternal	tree cavity	aspen snag
F09-394	2010	4/20	maternal	tree cavity	26.1” dbh live aspen
F08-353	2010	4/22	maternal	tree cavity	24.3" dbh aspen snag
F10-503	2011	4/25	maternal	tree cavity	23.7” dbh live pine
F09-394	2011	5/4	maternal	tree cavity	19.8” dbh live aspen
F10-507	2011	5/9	maternal	Hollow log off ground	18.3” diam. aspen
F09-461	2010	5/18	maternal	tree cavity	22.3” dbh live aspen
F09-360	2009	5/29	maternal	tree cavity	19.1” dbh live oak
F08-375	2008	6/25	maternal	Hollow log on ground	15.7" diam. sugar maple

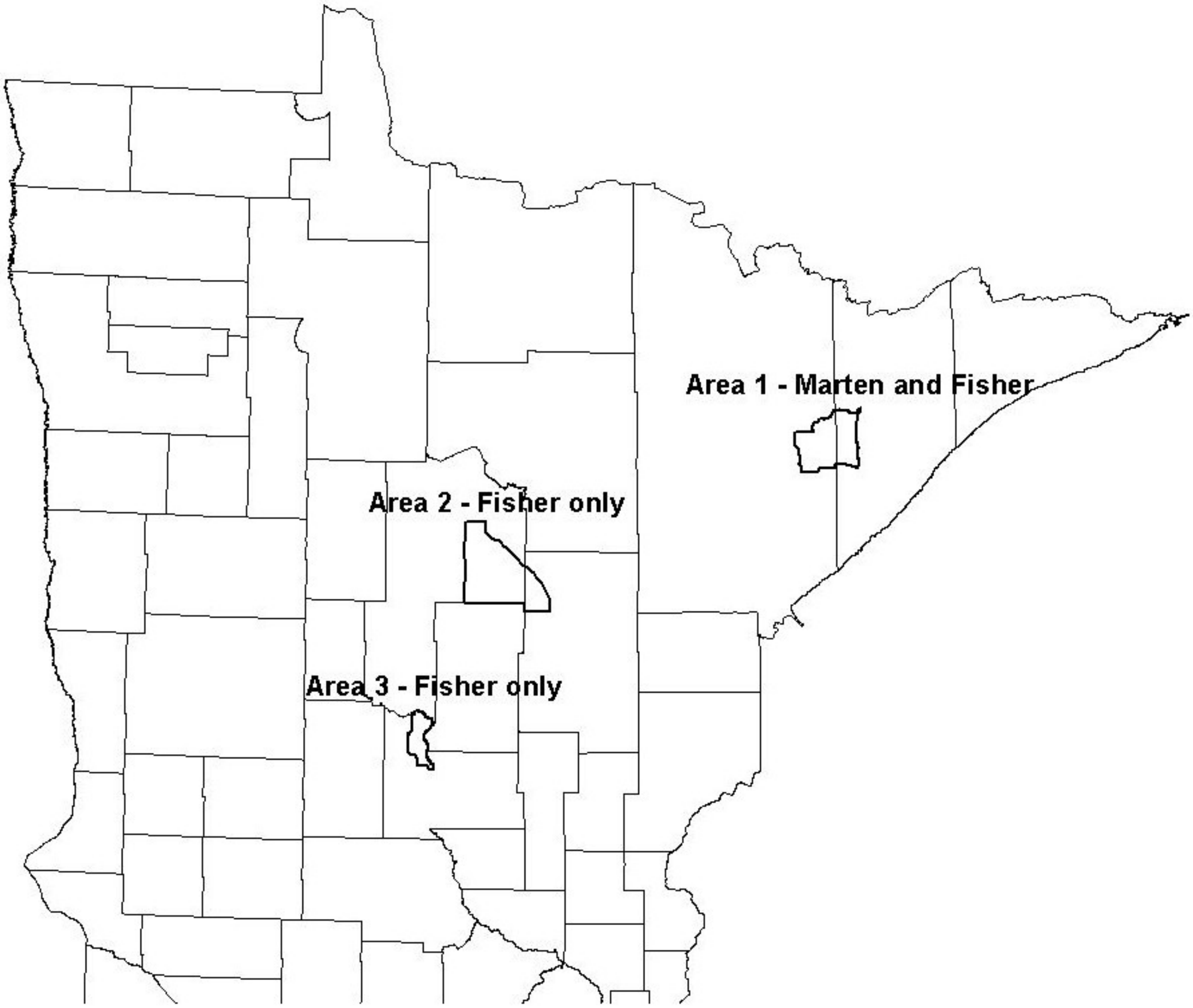

Figure 1. Fisher and marten study areas in Minnesota, 2008-2012.

SURVIVAL AND CAUSES OF MORTALITY FOR FISHER AND MARTEN IN MINNESOTA

John Erb, Barry Sampson, and Pam Coy

SUMMARY OF FINDINGS

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

Of the 80 fishers radio-collared, 23 are still being monitored (14F, 9M), radio contact was lost on 24, and 33 deaths (18F, 15M) have occurred. Of the 33 fisher deaths, 11 (33%) were attributed to humans [2 (1M, 1F) hit by cars and 9 (6M, 3F) trapped], whereas 22 (67%) were attributed to natural causes (17 (5M, 12F) predation deaths and 5 (3M, 2F) from unknown natural causes). Nearly all of the predated fishers were killed in spring. Eleven of the 12 female fisher predation mortalities were attributed to other mammalian carnivores, while 4 of the 5 male fisher predation mortalities were attributed to raptors (all bald eagles). Of greatest significance, 11 of the 12 female fishers killed by predators were adults, and 10 of the 11 adult females were killed while they still had dependent young in natal dens, indirectly resulting in the death of all their offspring. The deaths of these 10 nursing females represent ~28% of the adult female fishers monitored during the kit-rearing season since the study began. We hypothesize that the timing and magnitude of female mortality is a result of increased movement and increased vulnerability at this time of year. However, it remains unclear whether the pattern we have observed to date is consistent with past dynamics, and if not, whether the underlying explanation is related to short-term (e.g., periodic fluctuations in prey) or long-term (e.g., deteriorating habitat quality) changes affecting fisher energetics/activity, or a result of changes in the predator community. What is clear from initial results is that for both species, predation has been the largest single source of mortality.

INTRODUCTION

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

Legally, fisher and marten were unprotected in Minnesota prior to 1917, after which harvest season length restrictions were implemented. These protections were removed in the mid-1920s, and remained so until all harvest was prohibited in 1929. Seasons remained closed until 1977 for fisher and 1985 for marten, when limited harvests were reinstated. While marten harvest is legal in approximately the northern 50% of the state, most harvest occurs in counties bordering Canada, particularly in northeast and north-central Minnesota. Fisher harvest occurs in most of the northern 50% of the state, though harvest is comparatively low in extreme northeast Minnesota (Lake and Cook counties), and lower, though perhaps increasing, in the Red River Valley (western Minnesota) and the highly fragmented transitional forests in central Minnesota. Peak harvest levels have been near 4,000 and 3,500 for marten and fisher, respectively. However, due to apparent multi-year population declines for both species, harvest seasons from 2007-2011 were reduced from 16 days to 9 days, and starting in 2010, the fisher harvest limit was reduced from 5 to 2. During this 5-year period, harvests have averaged ~2,100 and 1,400 for marten and fisher, respectively.

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

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

Natural mortality, particularly via predation, appears more common with martens. Marten survival data is available from Wisconsin (McCann et al. 2010), Maine (Hodgman et al. 1994, 1997), Ontario (Thompson 1994), Oregon (Bull and Heater 2001), British Columbia (Poole et al. 2004), Alaska (Flynn and Schumacher 1995, 2009), Quebec (Potvin and Breton 1997), and Newfoundland (Fredrickson 1990). Although we do not summarize details of these studies here, a couple conclusions are worthwhile. First, when comparing across studies, annual adult non-harvest mortality rates varied from ~ 0.07 – 0.48. Juvenile data was rarely separated, but a few studies pooled ages, and mortality rates also were within the above interval. While this variability may be attributable to both sampling and biological variability, the wide range suggests that it is risky to assume results from any area are applicable elsewhere. Secondly, at least 1 study (Maine; Hodgman et al. 1997) has documented significantly higher natural mortality for females compared to males, and others researchers have postulated this to be common given the typical male–biased harvest, 50:50 sex ratio at birth, and often balanced adult sex ratio (Strickland et al. 1982, Strickland and Douglas 1987). Due to male-biased harvest and our *assumed* sex-related equality in non-harvest mortality, our marten population model previously projected a very female-biased population, contradicting our preliminary capture results and suggesting that our model inputs were overestimating female survival, underestimating male survival, or incorrectly assuming a 50:50 birth sex ratio.

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

STUDY AREA

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

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

METHODS

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

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

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

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

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

RESULTS AND DISCUSSION

Including the pilot year of the study, a total of 169 martens (80F, 89M) and 80 fishers (44F, 36M) have been radio-collared. An additional 7 animals (4 martens, 3 fishers) were ear-tagged only. Tooth aging has not yet been completed for all animals, and herein we do not report any formal survival estimates. Instead, we provide a simple overview of the fate of collared animals.

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

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

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

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

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

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

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

Regardless of the explanation, and acknowledging the limited sample size, it seems unlikely that the high level of predation we have observed to date on nursing female fishers is sustainable, which may partially explain the recent decline in fisher abundance. However, many of the correlates to the timing of predation mortality that we have mentioned are not new challenges for adult female fisher, and the population appears to have been in decline only for the last ~ 7 years, suggesting that other factors may be altering the 'system'. While it seems unlikely that the fisher mortality pattern we have observed to date is consistent with past dynamics, it remains uncertain whether the changing dynamics are related to short-term (e.g., periodic fluctuations in prey) or long-term (e.g., suspected decline in natal and maternal den availability) changes affecting fisher energetics or activity, the result of relatively rapid changes in the predator community (i.e., the rapid increase in bobcats, which are known or suspected to be responsible for many of the female fisher predation deaths), or some other unknown factor.

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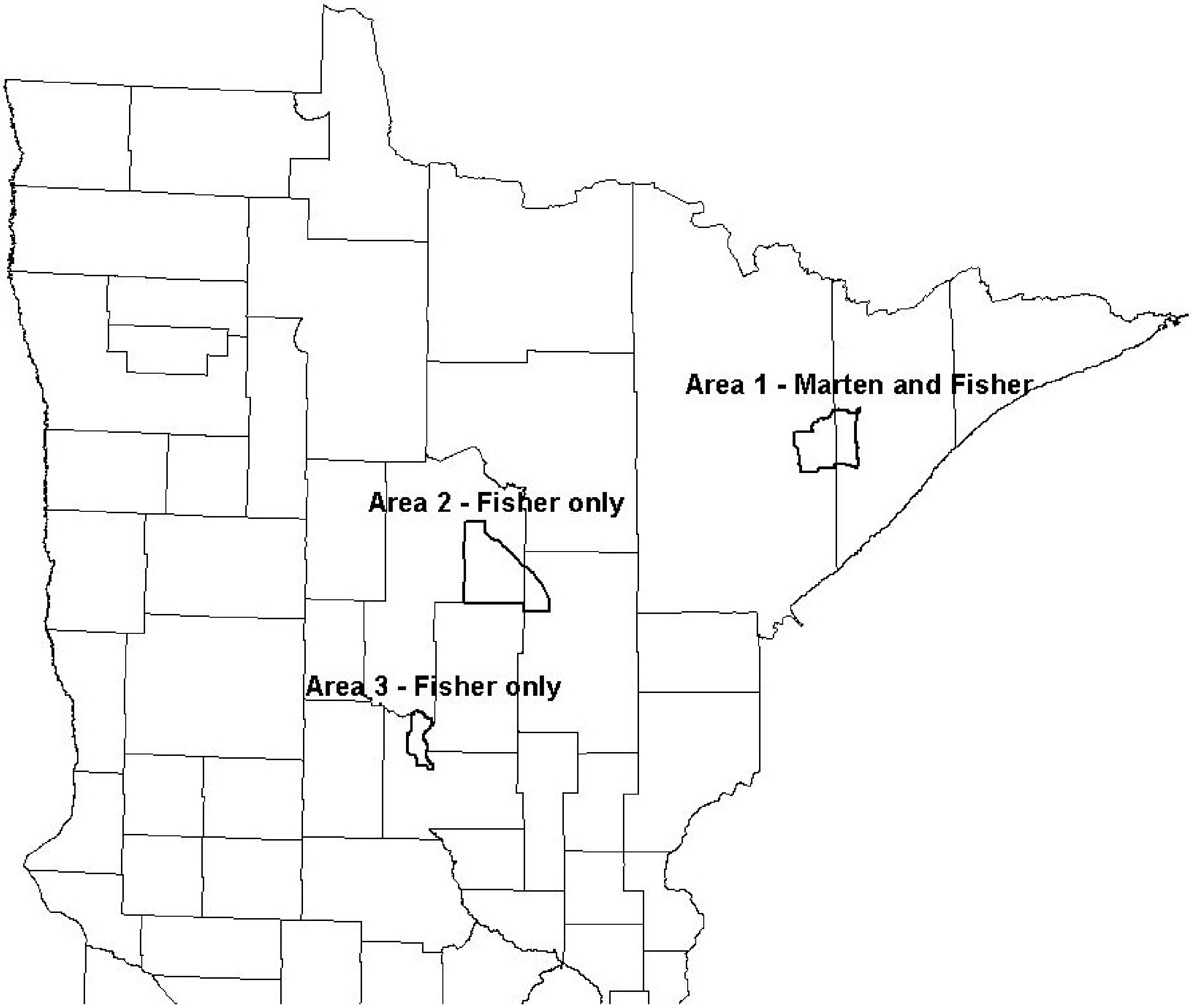

Figure 1. Fisher and marten study areas in Minnesota 2008-2012.

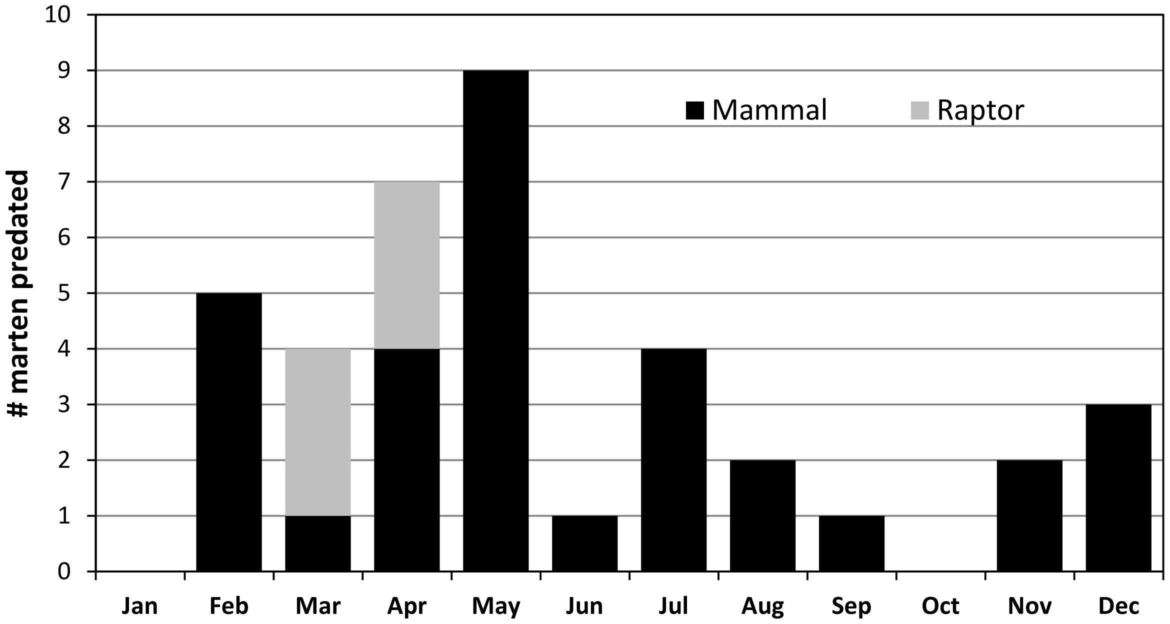

Figure 2. Seasonal timing of marten deaths attributable to predation in northeast Minnesota, 2007-2012.

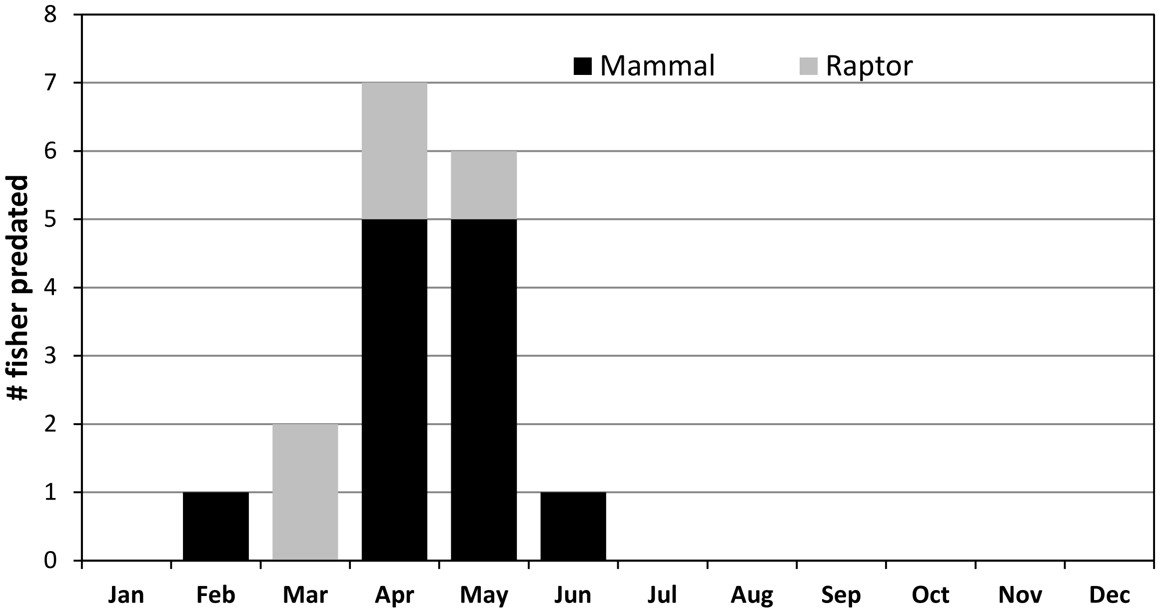

Figure 3. Seasonal timing of fisher deaths attributable to predation in north-central Minnesota, 2007-2012.

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

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

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

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

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PRELIMINARY RESULTS FROM THE 2010-2011 MOOSE HEALTH ASSESSMENT PROJECT

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SUMMARY OF FINDINGS

This project, which began in 2007, represents the first 2 years of the second phase (2010 – 2012) of an overall health assessment of hunter-harvested moose (*Alces alces*) in northeastern Minnesota (MN The objectives of this project are to: (1) Screen hunter-harvested (and presumably healthy) moose from 2010 to 2012 for select disease agents, (2) Monitor changes in disease incidence or prevalence over time, (3) Assess the clinical impacts of liver fluke (*Fascioloides magna*) infection on moose, and (4) Determine the frequency of histological lesions consistent with brainworm (*Parelaphostrongylus tenuis*) infection. Samples were collected from 199 moose (*n*= 128 in 2010 and *n* =76 in 2011). Moose were screened for West Nile virus, eastern equine encephalitis, western equine encephalitis, St. Louis encephalitis, malignant catarrhal fever, borreliosis (*Borrelia burgdorfer*i), anaplasmosis (*Anaplasma phagocytopila*, formerly *Ehrlichia phagocytophila*) and 6 serovars of leptospirosis. There was evidence of exposure to West Nile Virus (25%), malignant catarrhal fever (8%), borreliosis (21%), and leptospirosis (0.6–7.5%). Portions of brain, cerebral spinal fluid, whole blood, and serum were submitted for polymerase chain reaction (PCR) for Flavivirus RNA. Whole livers and brains were collected and examined grossly and histologically for evidence of brainworm and liver flukes; both parasites were documented. Full serum chemistry profiles were conducted on 158 moose and were used to determine if a correlation exists between liver fluke damage and serum liver enzymes. Whole blood samples from 168 moose were submitted for evaluation for tick-borne illnesses; anaplasmosis and piroplasma infections were also documented.

INTRODUCTION

The current aerial survey trend data indicates the moose population in northeastern MN is declining. Since 2002, annual survival and reproductive rates were substantially lower than documented elsewhere in North America (Lenarz et al. 2007). Further, the population estimate has declined over 50% from 2005 (*n*= 8,160) to 2012 (n = 4,230) (Lenarz 2012). Likewise, recruitment and twinning rates have steadily declined since 2002 (Lenarz 2011).

Previous and ongoing research has been unable to determine proximate and ultimate cause(s) of non-hunting moose mortality and the possible related impacts to the long-term viability of the northeastern MN population. In 2007, the MN Department of Natural Resources (MNDNR) began a 3-year moose health assessment project to determine which diseases northeastern MN moose are being exposed to and to establish baseline hepatic mineral levels. Results indicated that hunter-harvested moose in northeastern MN have been exposed to a variety of disease agents such as West Nile virus (WNV), eastern equine encephalitis (EEE), malignant catarrhal fever (MCF), anaplasmosis, borreliosis, and leptospirosis (Butler et al. 2010). While these findings were illuminating, there remained some key factors, the importance of which, we have been unable to determine, including: (1) The role liver damage (due to liver flukes) plays in non-hunting mortality, 2) The impact of arboviruses and how their incidences may be affected by changing climate, and (3) The impact of brainworm on moose survival, due to the difficulty in interpreting brain lesions caused by this parasite. To begin addressing these key factors, a second phase of the moose health assessment project was started in 2010.

Murray et al. (2006) concluded that moose in northwestern MN were dying from high liver fluke loads. However, assessing the extent of liver damage caused by flukes can be subjective. In order to determine if liver damage caused by flukes has clinical implications, serum liver enzymes should be evaluated. Beginning in 2009, we asked hunters to collect whole livers for evaluation. Samples were then ranked for liver fluke loads by a board-certified veterinary pathologist. Results from this pilot year of liver examinations indicated that 35% of livers had fluke-induced lesions with some having nearly 100% of the liver parenchyma affected (Butler et al. 2010). However, poor blood collection techniques prevented assessment of the clinical impacts of the damage caused by the liver fluke infections. In 2010, we asked hunters to alter their blood collection strategies and began collecting both the whole liver and assessing serum liver enzymes, with the goal of determining whether results of gross evaluation of the liver correlated with enzyme indicators of liver function.

Our moose health assessment during 2007–2009 indicated that moose are being exposed to a variety of arboviruses, including EEE, WNV, borreliosis, and anaplasmosis (Butler et al. 2010). As climate changes, the density and distribution of capable arthropod vectors is expected to change as well (Gould and Higgs 2009). Climate is known to play a key role in determining the geographical and temporal distribution of arthropods, characteristics of arthropod lifecycles, dispersal patterns of associated arboviruses, evolution of arboviruses, and the efficiency with which they are transmitted from arthropods to vertebrate hosts (Gould and Higgs 2009). For example, there has been a substantial increase in tick-borne encephalitis in Sweden since the mid-1980s related to milder winters and earlier arrival of spring (Lindgren and Gustafson 2001). In Phase 2 of the moose health assessment study, serum will be screened for these arboviruses and a few additional select disease agents. Combined with results from our 2007–2009 sampling, we will have 6 years of data on the incidence of arbovirus exposure in our moose herd to evaluate any significant trends relative to fluctuations in climate. Additionally, beginning in 2011, samples were submitted for western equine encephalitis (WEE) and St. Louis encephalitis (SLE).

Diagnostics have shown that moose are dying from brainworm in MN. It is also known that moose are able to survive low-dose infections of brainworm and even develop immunity to subsequent infections (Lankester 2002). Researchers have hypothesized that brainworm was responsible for historic declines in moose populations (Karns 1967, Prescott 1974, Lankester 1987), but it is questionable whether brainworm represents a major threat to the northeastern MN population. In 2008, we began collecting whole brains from hunter-harvested moose to determine the frequency of brain lesions consistent with past brainworm infections in presumably healthy moose. These data would allow for better interpretation of migration tracts and could prevent pathologists from wrongly assigning brainworm as the cause of death based solely on the presence of migration tracts. We will continue to collect whole brains to increase our sample and quantify the number of presumably healthy moose have parasitic migration tracts.

METHODS

Hunters (tribal and state) were asked to collect whole livers, blood, hair, and a central incisor. State hunters were only allowed to harvest bulls while some tribal hunters were able to take either bulls or cows. Wildlife Health Program staff provided a presentation and instructions relative to the moose health assessment project at the mandatory MNDNR Moose Hunt Orientation Sessions and at 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 moose registration.

The MNDNR provided hunters with all the equipment needed for sample collection and preservation. Sampling kits included a cooler, 1-60-cc syringe for blood collection, 6-15-cc serum separator tubes, 2-5-cc ethylenediaminetetraacetic acid (EDTA) blood tubes for whole blood collection, 1 heavy-duty bag for liver storage, 2 coin envelopes for the tooth and hair collected, data sheet, protocol, Sharpie marker, 1 pair of large vinyl gloves, and 1 icepack.

Hunters collected blood using the 60-cc syringe after incising the jugular vein as soon after death as possible and recorded time of death and blood collection. Blood was placed in serum-separator tubes and in an EDTA tube and kept cool until they were delivered to official

MNDNR registration stations or tribal natural resource offices. Livers were placed in heavy-duty, pre-labeled bags.

At the stations or offices, serum-separator tubes were centrifuged and the serum decanted. Blood spinning time was recorded. Portable refrigerators were located in advance at the registration stations to maintain the tissue samples. One whole blood sample (EDTA tube) and 1 mL of serum were refrigerated and submitted every 2–3 days to the University of MN (UMN)-College of Veterinary Medicine-Clinical Pathology Laboratory for a full large-animal serum chemistry profile. The remaining whole blood sample was submitted every 2–3 days to the UMN-Department of Entomology for testing for tick-borne illnesses. Remaining serum and the whole livers were frozen. Cerebral spinal fluid was collected when possible. Whole brains were removed with the hunter's permission and placed in formalin. A 1x1x1" piece of brain was removed and frozen. The serum, whole liver, and whole brains were submitted to the UMN Veterinary Diagnostic Laboratory (UMN VDL, St. Paul, MN). The 1x1x1" piece of brain, cerebral spinal fluid, whole blood, and 1 mL of serum were submitted to the Minnesota Department of Health (MDH) for PCR for Flavivirus RNA.

Serum was tested for WNV, EEE, and WEE with a plaque reduction neutralization test (PRNT) and SLE with a serum neutralization test at the National Veterinary Services Laboratory (NVSL) in Ames, Iowa. Serum was screened for leptospirosis (microscopic agglutination test), borreliosis (immunofluorescence assay), anaplasmosis (card test), and MCF via peroxidase-linked assay (PLA) with positive PLA tests further tested with a virus neutralization test (VN) at the UMN VDL. The livers were ranked by a board-certified veterinary pathologist based on parenchymal damage due to liver flukes; ranking included no fluke-induced lesions (no evidence of fluke migration), mild infection (<15% of 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 (15–50% of the liver parenchyma affected by nodules and fibrosis), and marked infection (51–100% of the liver parenchyma affected with deformation of the entire liver by larger nodules with widespread fibrosis). Brains were examined histologically with 4 complete coronary brain, cerebellum, and brain stem sections processed from each moose. An average of 25 histological slides per animal were examined, including the frontal, temporal, parietal, and occipital lobes and the basal nuclei, thalamus, mesencephalon, and brain stem. Central incisors of moose were submitted to Mattson's Laboratory (Milltown, Montana) for aging by cementum annuli (Sergeant and Pimlott 1959).

RESULTS AND DISCUSSION

Samples from 199 moose (*n* = 128 in 2010 and *n* = 76 in 2011) were collected (189 males, 7 females, and 3 sex unknown) (Figure 1). Exact age was determined for 196 of these moose (median = 4, range = 1 – 13 years old).

Eastern Equine Encephalitis

Evidence of exposure to EEE was detected in 1/174 (0.6%) moose. The low detection rate in these moose was unexpected as an average exposure rate of 6.1% of the population was documented during Phase 1 of this study (Butler et al. 2010).The continued surveillance for EEE in Phase 2 of this study may provide greater insight into the annual variation in apparent disease prevalence.

A total of 65 moose were sampled (frozen brain, cerebral spinal fluid, serum, and whole blood) by the MDH by PCR for evidence of any Flavivirus RNA. All results were negative.

Mosquitoes spread EEE, which can cause neurologic signs and often death. It poses a greater mortality threat for most species than WNV, although the effects of EEE infection have not been studied in moose.

West Nile Virus

Evidence of exposure to WNV was detected in 44/174 (25%) moose. These results were similar, though slightly lower, to those reported during the first 3 years of the study (35%; Butler et al. 2010). Positive results indicated that animals were exposed to the WNV, but does not necessarily indicate illness. 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.

Western Equine Encephalitis and St. Louis Encephalitis

Of the 64 sera samples submitted for WEE and SLE testing, none tested positive. Both of these diseases are mosquito-borne. WEE is known to occur infrequently in MN, although when it does, it is often part of a regional outbreak.

Malignant Catarrhal Fever Virus

Evidence of exposure to MCF was detected in 14/174 (8%) moose sampled with PLA. Follow-up testing with VN was negative for 12 of the 14, and the remaining 2 were unsuitable for testing. These PLA results are markedly lower than what we reported from 2007 to 2009 (35%; Butler et al. 2010). The PLA test is more sensitive than VN, meaning it is much better at identifying true positives, whereas VN is more specific and thus better at identifying true negatives. Malignant Catarrhal Fever is a gammaherpes virus, of which there are multiple strains (e.g., wildebeest strain of MCF, sheep strain of MCF, deer strain of MCF). The PLA reacts with multiple gammaherpes viruses. A PLA positive does not indicate the strain of exposure. 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 MCF, but not the wildebeest strain.

We have been collaborating with researchers (Dr. Hong Li, Washington Animal Disease Diagnostic Laboratory) to determine the strain of MCF exposure in the northeastern MN moose population. 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 brainworm infection, including neurological deficits, blindness, and thrashing 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.

Anaplasmosis

No evidence of exposure to anaplasmosis was detected in moose screened for this disease in 2010 (*n* = 100). These results are similar to the results of 2007–2009 screening (1/319, 0.3%; Butler et al. 2010), indicating that exposure to this bacterium is likely occurring, albeit at a low rate. Anaplasmosis testing was no longer available in 2011.

Moose are thought 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*. In sheep, this disease produces significant effects on the immunological defense system, increasing their susceptibility to disease and secondary infections (Larsen et al. 1994).

Borreliosis

Evidence of exposure to borreliosis was detected in 37/174 (21%) moose sampled. These results are similar to results from 2007 to 2009 (23%, Butler et al. 2010).

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.

Leptospirosis

A total of 110 samples were screened for 6 serovars of *Leptospira interrogans*. Results per serovar are as follows:

- *L. interrogans bratislava*:
  - 1/173 (0.6%)
- *L. interrogans canicola*:
  - 1/173 (0.6%)
- *L. interrogans grippothyphosa*:
  - 1/173 (0.6%)
- *L. interrogans hardjo*:
  - 0/173
- *L. interrogans icterohaemorrhagicae*:
  - 1/173 (0.6%)
- *L. interrogans pomona*:
  - 13/173 (7.5%)

While the prevalences are lower for most of the serovars compared with data from 2007–2009, the prevalence of *L. pomona* remained stable (Butler et al. 2010). 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, not uncommon in moose habitat.

General Tick-Borne Illness Screening

Whole blood samples from 168 (n = 109, n = 59 in 2010 and 2011, respectively) moose were submitted to the UMN Department of Entomology, where we are collaborating with Dr. Ulrike Munderloh to determine if hunter- harvested moose are infected with tick-borne illnesses. Samples were screened with a variety of PCR techniques. Results, only available for the 2010 samples, indicate that 10% of the moose were infected with anaplasmosis and 32% were positive for prioplasma primers. A hemolytic *Mycoplasma* was also identified in 19 of the samples. Further analysis is pending.

Brain Histopathology

Seventy-one whole brains were collected (n = 40 and 31 in 2010 and 2011, respectively). Since 2008, a total of 118 whole brains have been collected and examined. No lesions were found in 101 (86%) of the brains, 12 (10%) had lymphocytic infiltration (unspecific chronic inflammatory lesion), and 5 (4.2%) had lesions consistent with larval migration tracts (mild to moderate meningitis, axonal degeneration, and secondary demyelination).

Whole Liver Evaluation

Whole livers were collected from 169 (*n* = 108 and 61 in 2010 and 2011, respectively). Combined with livers collected in 2009 (*n* = 57), 226 livers have been submitted for gross examination. Of the 226 livers examined, 162 (72%) had no fluke-induced lesions, 34 (15%) had mild infection, 22 (9.7%) had moderate infection, and 8 (3.5%) had marked infection. Collection of whole livers will continue in 2012. Additionally, beginning in 2010, serum was submitted for a serum chemistry profile in an attempt to correlate serum liver enzyme levels with the level of fluke-induced damage. These results have not yet been analyzed.

Serum Chemistries

A total of 158 (*n* = 95 and 63 in 2010 and 2011, respectively) serum samples were submitted for a full large animal serum chemistry profile. Analysis of these results is pending. The purpose of collecting these data is to determine if there is a correlation between the liver ranking and serum liver enzymes, as well as to establish baseline “normals” for animals in this population.

Future Research

This project was the first to document EEE activity in NE MN, though extensive surveillance has not previously occurred. Because of its potential to cause illness and even death in humans and domestic animals, we have initiated a collaborative project with the MDH, the UMN, College of Veterinary Medicine, the UMN, Department of Entomology, and the Metropolitan Mosquito Control District. Mosquitoes will be trapped in a weekly basis at various locations throughout moose zones. The objectives of this project are 1.) Assess spatial and temporal distribution patterns of vectors of EEE throughout NE MN moose range 2.) Assess prevalence of EEE (and other arboviruses) in vector populations. Interestingly there appears to be differences in exposure rates between moose zones. Little is known about vector distribution in this area. This project will identify which mosquito species are present, how the species make-up changes over the summer, whether there is an actual difference in vectors between moose zones, and determine the prevalence rates of the arboviruses in the mosquitoes themselves.

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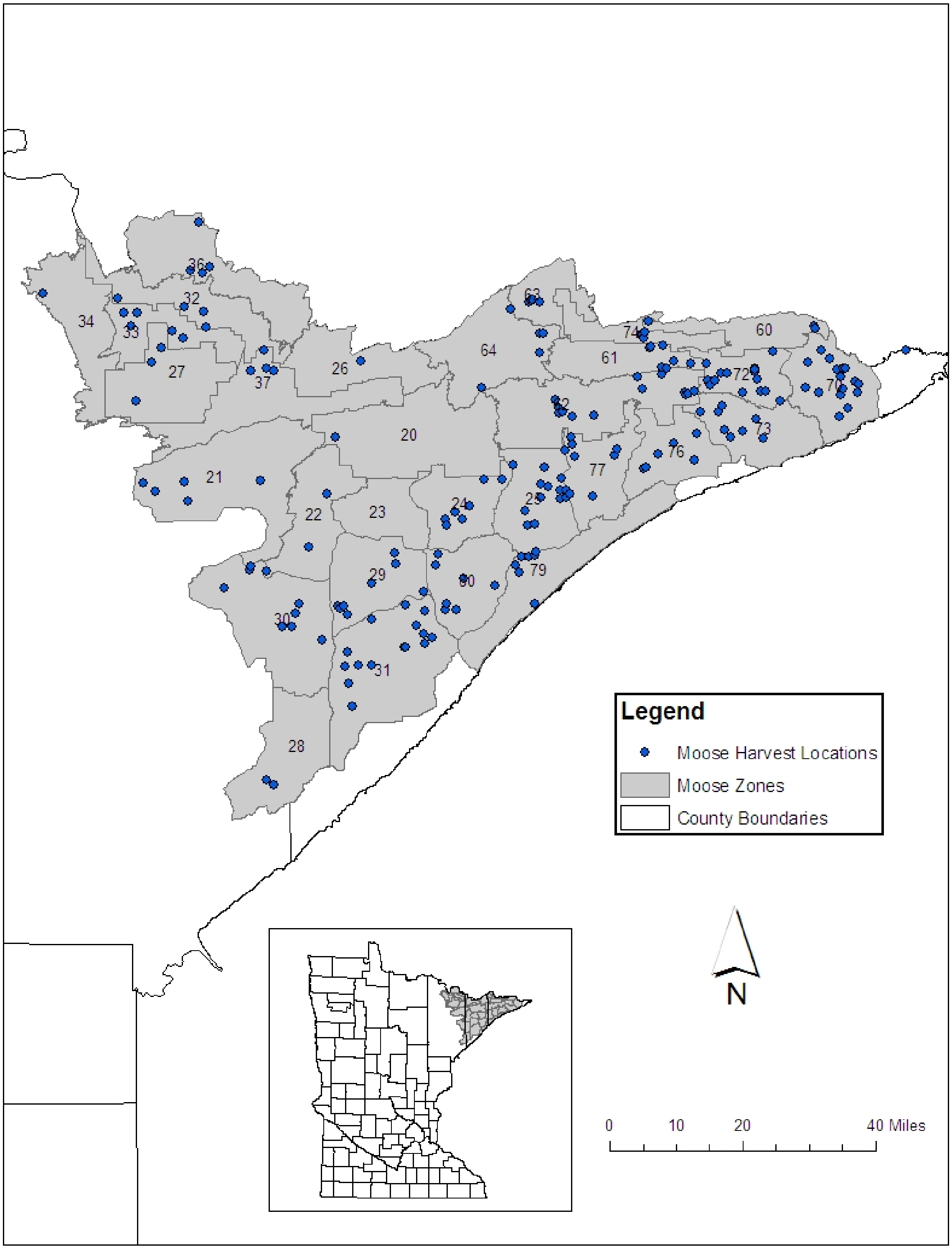

Figure 1. Harvest locations of hunter-harvested moose (n=197) included in 2010 and 2011 moose health assessment project, northeastern Minnesota.

MINNESOTA GRAY WOLF DISEASE SCREENING AND MORPHOLOGY

Erika Butler, Michelle Carstensen, Dan Stark, Erik Hildebrand, and John Erb

SUMMARY OF FINDINGS

A total of 442 wolves (*Canis lupus*) were included in this 2-year study to document the apparent prevalence of diseases and parasites in Minnesota’s wolf population, as well as provide insight into their genetic makeup. Our results indicated serologic exposure of wolves to 8 diseases: canine parvovirus (72%), canine adenovirus (79%) canine distemper virus (18%), eastern equine encephalitis (3.1%), West Nile virus (32%), heartworm (6.2%), Lyme (76%), and neosporosis (84%). Parasites were discovered in 15% of fecal samples examined. Genetic analyses are pending.

INTRODUCTION

Minnesota's gray wolf population was delisted from the Endangered Species Act in January, 2012. Following that ruling, wolves are managed in Minnesota by state statute, rule, and under the wolf management plan (2001) by the Minnesota Department of Natural Resources (MNDNR) This plan is designed to protect wolves and monitor the population while providing owners of livestock and domestic pets more flexibility in addressing wolf depredation. A primary component of wolf population monitoring is to understand what diseases and parasites might be impacting them. Furthermore, the collection of morphological and genetic data will add current and more spatially comprehensive data to the ongoing debates regarding the genetic identity of wolves in Minnesota.

There are a number of diseases and parasites known to affect wolves that can have population-level impacts. Most notably, relatively high prevalence of canine parvovirus (CPV) has been reported in Minnesota and could be adversely impacting pup survival and limiting population growth (Mech et al. 2008). Although further analysis indicated that the strongest effect of CPV and wolf population change occurred from 1987-1993 and after that had little effect despite higher seroprevalence levels (Mech and Goyal 2011). Other diseases, including canine distemper, adenovirus, and parasites may also kill infected wolves and impact population performance. Furthermore, some diseases, such as neosporosis, are of particular concern to livestock producers; gaining a more thorough understanding of the prevalence and distribution of this disease may benefit wolf management strategies.

There is some uncertainty in the taxonomic and genetic identity of wolves in the Great Lakes Region (Leonard and Wayne 2008, Mech 2008, Koblmuller et al. 2009, Nowak 2009, Schwartz and Vucetich 2009, Wheeldon and White 2009, Mech 2010, vonHoldt et al. 2011). Mech has suggested that non-genetic data support that wolves in Minnesota are hybrids between the gray wolf (*Canis lupus*) and the Eastern wolf (*Canis lycaon*) (2011). This portion of the project will systematically assess both genetic and morphological characteristics of a large sample of wolves in Minnesota. Relating wolf morphology to genetics should help determine the taxonomic identity of wolves throughout Minnesota, and reveal any potential geographic patterns of species introgression. The December 2011 federal delisting rule addressed this issue and determined that wolves in the Western Great Lakes are predominantly gray wolves with some admixture of either coyote or eastern wolf and that there is not sufficient evidence to suggest that there was a significant proportion of the population representative of the purported eastern wolf.

METHODS

The MNDNR entered into a contract with the United States Department of Agriculture (USDA)-Animal and Plant Health Inspection Service (APHIS)-Wildlife Services (WS) to collect biological samples from all dispatched wolves immediately after death. Researchers from the MNDNR, the United States Geological Survey (USGS), Camp Ripley military base, and tribal authorities that capture and radiocollar wolves also were involved in sample collections. Conservation officers and Area Wildlife staff assisted in collecting samples from vehicle-killed wolves. All key personnel were trained in proper sample collection and handling, as well as recording morphological measurements. Sampling kits provided to data collectors included the following items: soft-sided cooler, 1-20cc syringe for blood collection, 6-10-cc serum tubes for blood storage, 1-5-cc EDTA tube for whole blood, 1 whirlpak for fecal collection, 1 ear punch, 1 FTA card, 1-2-mL vial with 95% ethanol, 1-2-mL vial with desiccant, tape measure, caliper, data sheet, protocol, Sharpie, 1 pair of large vinyl gloves, and 1 icepack.

Our goal was to collect samples from wolves throughout the extent of their range in Minnesota; however, the vast majority of samples were collected by USDA-WS with an expected bias toward depredating wolves. Opportunistic sampling (e.g., vehicle kills) was encouraged to help increase sample size and provide a better distribution in more remote areas within wolf range.

Blood was collected from the jugular vein whenever possible (cephalic vein or saphenous vein are also options). For *post-mortem* collections, blood was obtained from the site of a bullet wound, heart, or from the chest cavity as soon after death as possible. In all cases, blood was centrifuged and serum extracted. Whole blood samples were kept cool and sent to an entomologist at the University of Minnesota for tick-related disease research. Fecal samples were collected from the rectum and placed in a whirlpak bag. Heart and brain samples were also collected from dead wolves when possible.

Sera were screened for 8 diseases at the Veterinary Diagnostic Laboratory at the University of Minnesota (UMN-St. Paul) and the National Veterinary Services Laboratory (Ames, Iowa). The presence of CPV was confirmed using a hemagglutination inhibition (HI) test; titers ≥256 were considered positive. Exposure to canine adenovirus 1 (CAV 1) was confirmed using a serum neutralization test (SN); titers ≥8 were considered positive. Canine distemper virus (CDV) was also detected using a SN test; titers ≥25 were considered positive. A plaque reduction neutralization test (PRNT) was used to confirm exposure to eastern equine encephalitis (EEE) and West Nile Virus (WNV), and titers ≥10 were considered positive. Heartworm disease was detected by an antigen test. An immunofluorescence assay (IFA) was used for evidence of exposure to Lyme disease; titers ≥160 were considered positive. The MNDNR is collaborating with Dr. J. P. Dubey (USDA-Agriculture Research Service, Beltsville, Maryland) on a *Neospora* research project. Dr. Dubey used both a modified agglutination test (MAT) and a neospora agglutination test (NAT) on samples of serum, heart, brain, or feces to confirm neospora. A titer ≥25 on either the MAT or the NAT test was considered positive.

This was the final year of the two-year project. Our sampling goals were met as we intended to sample a minimum of 400 wolves and samples were distributed throughout wolf range. While we do intend to continue to sample live-caught wolves, the decision of whether or not we continue to sample dead wolves will be made once final analysis is complete.

RESULTS AND DISCUSSION

Samples from a total of 442 wolves (348 adults, 4 yearlings, 79 pups, and 11 of unknown age; 233 males, 201 females, and 8 unknown sex were included. These included wolves that were euthanized by USDS-WS (*n* = 255), live-caught research animals (*n* = 61), vehicle kills (*n* = 41), found dead (*n* = 79), and other (*n* = 6) (Fig. 1). Blood and fecal samples were not collected from wolves that had been dead for an extended period of time.

Serologic Disease Screening

Serological results indicated wolves were exposed to all 8 diseases included in our screening (Table 1). These tests only confirm past exposure, not current infection.

Our results indicated 72% of wolves have been exposed to CPV, which is similar to findings reported by Mech and Goyal (2011) for northeastern Minnesota. Canine parvovirus was first reported in 1967, but it wasn't until 1978 that a new variant of the virus was reportedly killing a high number of newborn wolf pups. It was theorized that this new variant of CPV was a mutation from feline parvovirus. This disease can infect most age classes of canids; however, mortality related to CPV in domestic canids has been primarily associated with younger animals (1–12 weeks of age). Mech and Goyal (2011) evaluated 35 years of relationships between pup survival, population change, and CPV seroprevalence in NE MN. They found the population effect of CPV was temporary, with the strongest effect on pup survival and wolf population change from 1987 to 1993. Following this time frame, little effect was reported and the authors concluded CPV became endemic and the population had acquired enough immunity to negate impacts of infection.

Canine parvovirus is transmitted through the fecal-oral route and causes diarrhea, fever, and dehydration. The disease can be fatal to wolves and is suspected of causing declines or attenuation of wolf populations in Wisconsin (Wydeven et al. 1995) and on Isle Royale, Michigan (Peterson et al. 1998).

Prevalence of CAV1 (78.6%) in wolves in our study was less than the 96% reported in Yellowstone's adult wolf population (Almberg et al. 2009). Canine adenovirus 1 causes hepatitis, a disease of the liver and other body organs. The virus is found worldwide and is spread by body fluids including nasal discharge and urine. Canids of any age are susceptible to the disease. The incubation period is from 6 to 9 days, and signs include fever, loss of appetite, congested mucous membranes, and pain in the region of the liver. Reported mortality in dogs (*Canis familiaris*) is about 10%, and about 25% of the survivors develop a temporary corneal opacity (hepatitis blue eye). Chronic infection may occur, leading to cirrhosis of the liver. It remains unclear how endemic CAV 1 infection might impact wolf populations.

Wolves in Minnesota showed similar exposure to CDV (18%) as Spanish wolves (19%, Sobrino et al. 2007). Canine distemper virus is a *Morbillovirus* that infects a broad class of canids. Animals acquire CDV through inhalation or ingestion of airborne particles (Murray et al. 1999), and clinical signs include pneumonia, encephalitis, and death. Since CDV occurs in several carnivore taxas, there is concern about horizontal transmission among species. Outbreaks of CDV in 1999, 2002 and 2005 in free-ranging wolves within Yellowstone National Park were correlated with high pup mortality rates (Almberg et al. 2009). The CDV appears to be capable of causing dramatic population declines over a short time- frame.

Eastern equine encephalitis is a member of the genus *Alphavirus* in the family *Togaviridae*, which has been a source of epizootics in both domestic and wild animals since the 19th century. Outbreaks are typically concentrated around swampy areas and have been found primarily in the southeastern U. S., but also in Michigan and Wisconsin. Transmission by mosquitoes is thought to be the primary source of exposure; however, direct contact with contaminated blood, feces, vomitus, semen, or assassin bugs also can be a source of infection. Clinical signs vary depending on the species. Little is known about EEE infection in wolves; however, the disease has been documented in domestic dogs (Farrar et al. 2005). Clinical signs in dogs were described as including pyrexia, depression, nystagmus, and lateral recumbency. Farrar et al. (2005) concluded that primarily young dogs are the most susceptible to EEE. This disease had not been known to occur in Minnesota prior to the MNDNR's moose health assessment project initiated in 2007, which discovered 6% of moose (*Alces alces*) in northeastern Minnesota have serological evidence of exposure to EEE (Butler et al. 2010). Our findings suggest northeastern wolves are also exposed to EEE, yet it is unclear what effect, if any, this may have on wolf survival.

West Nile virus is an avian virus that can cause fatal disease in some species of mammals, reptiles and birds. West Nile virus is an arbovirus in the *Flavivirus* genus of the family Flaviviridae. Until 1999, WNV was confined to the eastern hemisphere; however, it has since spread to North America and is now considered established in the U. S. and Canada. West Nile Virus is primarily transmitted by mosquitoes; 59 species are confirmed carriers in North America alone. A recent study of Minnesota's northeastern moose population found nearly 35% serologic prevalence (Butler et al. 2010), and their range overlaps with wolf range. While it remains unclear what effect WNV has on the nearly 32% of wolves that we documented were exposed to the disease, neurological signs have been reported from rare clinical cases in dogs and wolves. For example, a case of WNV was reported in a captive 4-month-old Arctic wolf pup (*C. lupus arctos*, Lanthier et al. 2004) and in a 3-month old wolf pup (Lichtensteiger et al. 2003). Both reportedly exhibited vomiting, anorexia and ataxia prior to death, which occurred 24–48 hours after the onset of neurological signs.

Results from 6.2% of wolves in our study indicated exposure to heartworm, which has been previously documented in Minnesota wolves by Mech and Fritts (1987). Mosquitoes are the major vector of dog heartworm, *Dirofilaris immitis*. Once the worms end up in a canine, they will mature and grow on the right side of the animal's heart and pulmonary arteries. Initial symptoms include detectable heart murmurs and pulse deficits. As the problem progresses, the animal's heart may become enlarged and if the infection becomes severe (up to 200 worms have been found in some animals), blood flow will be blocked. Heart failure may result from a major infection. Heartworm has not been reported in Canada or Alaska, as the mosquitoes that carry it prefer warmer climates.

Our findings indicated a significantly higher prevalence of Lyme disease (76%) than 2.5%, which was previously reported in wolves in Minnesota and Wisconsin (Thieking et al. 1992). Lyme disease is caused by the bacterium *Borrelia burgdorferi*, and can affect dogs, horses and humans. The disease was first discovered in New England in 1975, and has since been reported in at least 43 states and eastern Canada. Infection typically results from bites from infected *Ixodes scapularis* ticks (deer ticks). White-tailed deer (*Odocoileus virginianus*) are the major hosts for the mature ticks, whereas small rodents are the hosts for the immature ticks. These hosts can become infected with *B. burgdorferi*, but never show symptoms of the disease. Wolves in Minnesota and Wisconsin have been found to be infected with the disease, but clinical Lyme disease has not yet been found in wild wolves. A wolf was experimentally infected with *B. burgdrferi* and showed some symptoms of the disease (lymphadenopathy), which suggests that wolves may be susceptible to it (Thieking et al. 1992).

Samples from 239 wolves were submitted for Neospora testing; however, testing hasn't been completed on all the samples. To date, 128 have tested positive. *Neosporas caninum* is a protozoal parasite, which is best known for causing abortion in cattle and neurological disease in dogs. While wild herbivores and canids were thought to act as intermediate and definitive hosts, respectively (Gondim 2006, Dubey et al. 2009), findings originating from this research project confirmed the role of wolves as a natural definitive host (Dubey et al. 2011). While clinical disease due to infection is best described in domestic animals, reports of ill-effects due to *Neospora* infection in wildlife do exist. Gondim et al. (2004) reported that *N. caninum* antibody seroprevalence was detected in 39% of free-ranging gray wolves, 11% of coyotes (*Canis latrans*), 26% of white-tailed deer, and 13% of moose. These data are consistent with a sylvatic transmission cycle of *N. caninum* between cervids and canids. The authors speculated that hunting by humans favors the transmission of *N. caninum* from deer to canids, because deer carcasses are usually eviscerated in the field. Infection of canids, in turn, increases the risk of transmitting the parasite to domestic livestock.

Fecal Parasitology

Fecal samples were collected from 161 wolves and were examined by floatation for any evidence of ova or protozoal infection. Twenty (12 %) of the samples had hookworm ova, 9 (6%) had trematode ova, 41 (25%) had sarcocysts, and 2 (1%) were positive for Neospora. While this provides an idea of the types of parasites present in the wolf population, it does not provide an indication of parasite load or infection rate, as fecal-shedding does not correlate with severity of infection and shedding is often cyclical (Gondim 2006).

Wolves are susceptible to a variety of internal and external parasites. These include at least 24 species of nematodes (roundworms), 21 species of cestodes (tapeworms), 9 species of trematodes (flukes), heartworms, and 3 species of acanthocephalia (spiny-headed worms).

General Tick-borne Illness Screening

A total of 194 blood samples were submitted to the Department of Entomology (UMN), where we are collaborating with Dr. Ulrike Munderloh, to determine if wolves are infected with tick-borne illnesses. Whole blood samples were screened with a variety of polymerase chain reaction (PCR) techniques, which determine disease infection, not just disease exposure (which is detected through serology). Preliminary results from 38 of the 194 samples indicate that 7.9% of the wolves were infected with Anaplasmosis, 40% were positive for prioplasma primers, and 5.3% were infected with Lyme disease. Further analysis is pending.

Morphology and Genetic Analysis

Although 298 skulls have been collected for taxonomic evaluation, presently, only about 75% have been cleaned. We have initiated a collaboration for preparing and curating skulls with Dr. Sharon Jansa at the UMN Bell Museum of Natural History. As collection skulls are prepared for storage, measurements will be made per the protocol described by Nowak (1995). Each skull will be permanently cataloged in the mammal collection at the Bell Museum.

Genetic samples have been collected from 386 wolves. A subset (*n* = 150) have been submitted to the National Wildlife Forensics Laboratory for analysis, as in Fain et al. (2010). Results are pending. New information has been presented in vonHoldt et al. (2011), which indicates wolves in Minnesota are predominantly gray wolves with admixture from coyotes that dates between 600–900 years ago. However, different sources have presented competing information about the genetic identity of wolves in Minnesota; consequently, additional analyses may be required to enhance our understanding of their genetic makeup. Further, analysis of how skull morphology correlates to genetic identification may also contribute to our understanding of the taxonomic relationships of wolves in the region.

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Table 1. Serological results for disease screening of wolves sampled in Minnesota, January 2010–March 2012.

Disease	*n*	No. positives	Apparent prevalence (%)
Canine parvovirus	190	136	71.6
Canine adenovirus	192	151	78.6
Canine distemper virus	194	34	17.5
Eastern equine encephalitis	193	6	3.1
West Nile virus	194	62	31.9
Heartworm disease	195	12	6.2
Lyme disease	195	148	75.6
Neospora*	239	128	54

*some test results are pending; collaboration with Dr. JP Dubey, USDA-ARS

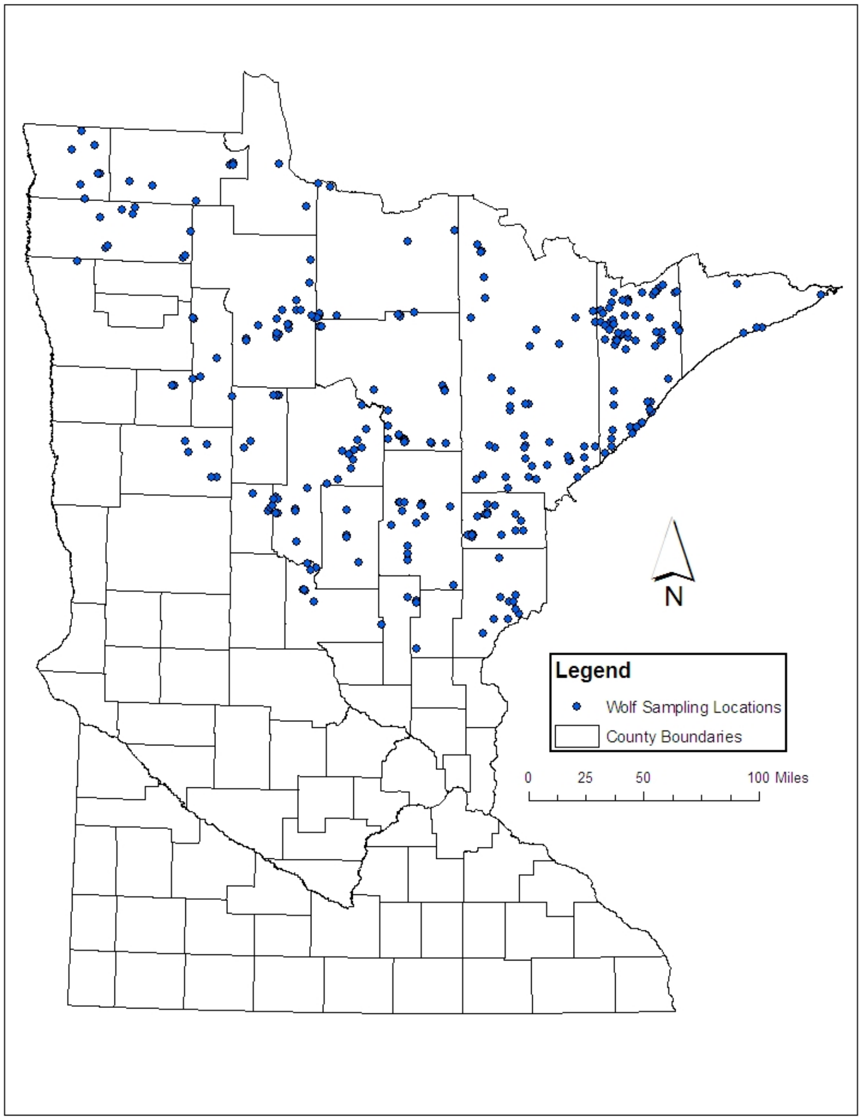

Figure 1. Sampling distribution of wolves (n = 442) included in the study of diseases and genetics of Minnesota's wolf population, 2010-2012.

CHRONIC WASTING DISEASE SURVEILLANCE IN MINNESOTA'S SOUTHEASTERN WILD DEER HERD

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

SUMMARY OF FINDINGS

In fall 2011, the Minnesota Department of Natural Resources (MNDNR) sampled 2,390 hunter-harvested white-tailed deer (*Odocoileus virginianus*) for chronic wasting disease (CWD) in southeastern Minnesota. The surveillance effort focused on testing deer within deer permit area (DPA) 602, and six surrounding DPAs: 233, 293, 341, 342, 343, and 344. All of the samples were negative for CWD. In addition, MNDNR submitted samples from 35 cervids through targeted surveillance, which included sick animals, escaped captive cervids, and roadkills; all these samples were also negative for the disease. The first and only detection of the disease in Minnesota's wild deer population occurred in a hunter-harvested deer from Olmsted county taken during fall 2010. To prevent further disease spread, MNDNR banned recreational feeding of deer in a 4-county area in southeastern Minnesota. MNDNR will continue to conduct CWD surveillance of hunter-harvested deer in fall 2012.

INTRODUCTION

To date, CWD has been diagnosed in 3 captive elk (*Cervus elaphus*) 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 from a mixed deer/elk herd in Lac Qui Parle County was discovered to be infected with CWD. That herd was also depopulated without additional infection being detected. In early 2009, a third captive elk herd (Olmsted County) was found infected with CWD and, following depopulation of >600 animals, a total of 4 elk were confirmed with the disease. The United States Department of Agriculture's (USDA) indemnification document noted there was an apparent longstanding infection within this captive elk facility.

Chronic wasting disease belongs to a family of infectious diseases, called transmissible spongiform encephalopathies (TSEs), which alter the morphology of the central nervous system, resulting in a "sponge-like" appearance of this tissue. Chronic wasting disease only affects elk, mule deer (*O. hemionus*), white-tailed deer, and moose (*Alces alces*). The etiological agent of CWD is an infectious protein, called a prion. Incubation time of the disease can range from 1.5 to nearly 3 years, although infected animals have been shown to shed prions in their feces up to a year before showing signs of illness (Tamguney et al. 2009). Clinical signs are non-specific and may include a loss of body condition and weight, excessive salivation, ataxia, and behavioral changes. There is no known treatment or vaccine for the disease and it is always fatal. Experimental and circumstantial evidence suggest that transmission of the disease is primarily through direct contact with infected animals or their infective saliva or excrement (Mathiason et al. 2006, Safar et al. 2008). However, persistence of prions in the environment and resulting indirect transmission has been shown to occur (Miller et al. 2004, Johnson et al. 2007, and Maluquer de Motes et al. 2008).

The Center for Disease Control (CDC) and other public health agencies have concluded there is no known link between CWD and any neurological disease in humans (MaWhinney et al. 2006). However, both the CDC and the World Health Organization (WHO) recommend that no part of a known positive animal should be consumed by humans. Additionally, there is no evidence that CWD can be naturally transmitted to species other than deer, elk, or moose.

Currently, Minnesota has approximately 644 domestic cervid facilities with approximately 20,000 deer, elk, and other cervidae in captivity. As the current statewide population estimate of wild deer approaches one million, there is an element of inherent risk associated with disease transmission between domestic and wild cervids. Overall, risk is difficult to quantify as deer populations are unevenly distributed across the landscape and range in densities from < 1 to 15 deer/km^2. In addition, domestic facilities are sporadically distributed on the landscape and are mutually exclusive of deer densities.

In response to the discoveries of the first Minnesota CWD-positive captive elk herd in 2002 and CWD in wild Wisconsin white-tailed deer, the MNDNR developed a comprehensive wild deer CWD monitoring program. This included surveillance of targeted animals (e.g., suspect or potentially sick deer exhibiting clinical signs or symptoms consistent with CWD), opportunistic surveillance (e.g., vehicle-killed deer), and hunter-killed deer surveillance. During 2002–2004, nearly 28,000 deer were tested for CWD statewide with no positive results. Following completion of the statewide surveillance, the MNDNR scaled back surveillance efforts and sampled animals in response to elevated risk factors (e.g., detection of CWD-positive animals in captive cervid farms in Minnesota, or proximity of positive CWD cases in wild deer in neighboring states). From 2004 to 2010, an additional 5,700 hunter-harvested deer and over 540 targeted or opportunistic deer were tested for CWD, with no positives detected. Since discovery of our index case, MNDNR has enacted its CWD Response Plan (http://files.dnr.state.mn.us/fish_wildlife/wildlife/disease/cwd/cwdresponseplan.pdf), which indentifies 4 primary goals for managing the disease:

1) Determine and monitor the prevalence and geographic distribution of CWD in the infected area,
2) Prevent or minimize further spread and new introductions of the disease,
3) Support and conduct applied research on CWD and its epidemiology, and
4) Provide accurate and current information about CWD to the public, constituent groups, and agency personnel.

METHODS

Hunter-harvested surveillance was conducted at deer registration stations during the archery, firearm, and muzzleloader seasons within DPA 602. MNDNR also conducted hunter-harvested surveillance within the six surrounding DPAs during four weekends of the regular firearm season. Stations were staffed with MNDNR personnel and students (veterinary medicine and natural resources) trained in lymph node collection during the regular firearm season. Head collection boxes were placed within DPA 602 during the archery and muzzleloader seasons for area wildlife staff to collect necessary samples. All samples were inventoried, entered into a database, and sent to either the University of Minnesota's Veterinary Diagnostic Laboratory (St. Paul, MN) or to Colorado State University (Fort Collins, CO) for enzyme-linked immunosorbent assay (ELISA) testing. Any presumptive positive samples from ELISA testing would be confirmed using immunohistochemistry (IHC) testing at the National Veterinary Services Laboratory in Ames, Iowa.

During fall 2011, registration stations were selected based on deer volume and distribution throughout the surveillance zone to meet a sampling goal of 600 deer minimum within DPA 602 and 300 from each of the surrounding DPA's. At the time of sample collection, hunter information was recorded, including the hunter's name, a 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 .50 caliber muzzleloader donated by the Minnesota Deer Hunters Association.

Within DPA 602, each registration station staffed for sample collection collected:

- All deer ≥1.5 years of age harvested by hunters were required to be sampled through extraction of retropharyngeal lymph nodes
- Both fawn and adult deer were issued carcass tags by authorized MNDNR staff
- MNDNR mailed samples from deer daily to University of Minnesota's Veterinary Diagnostic Laboratory in order to achieve a three business day turnaround time for results
- Carcasses were prohibited from being taken out of DPA 602 until they were test-negative

In the six surrounding DPA's, hunters were asked to voluntarily submit a retropharyngeal lymph node sample. Those samples were submitted on the Monday following each weekend during the regular firearm season, with a 7-14 day turnaround time for results. For all samples, hunters were able to check their test results on the MNDNR website using either their MNDNR number or the carcass tag number they were issued at the time of sample collection.

MNDNR continued 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

During fall 2011, the MNDNR sampled 2,390 hunter-harvested deer within the surveillance area, of which 1,125 were within DPA 602 (Figure 1).

From May 2011 to May 2012, MNDNR collected a total of 35 samples from targeted surveillance efforts. This included samples from 7 escaped captive cervids, 25 free-ranging sick deer, 1 free-ranging elk, and 2 vehicle-killed deer; all samples were negative for CWD.

Another key step in preventing further spread of CWD was to ban the recreational feeding of deer in a 4-county area (Dodge, Goodhue, Olmsted and Wabasha), surrounding the location of the CWD-positive deer found in fall 2010 (Fig. 2). The ban was aimed at reducing the potential for the disease spread by eliminating artificially-induced deer concentration sites. MNDNR Enforcement staff continues to educate and enforce the rule.

Given the results of the CWD surveillance efforts of fall 2011, evidence suggests that Minnesota is on the front end of a CWD outbreak in wild deer. The lack of detecting any additional infected deer in the immediate vicinity of the index case or in surrounding DPAs is encouraging. It may be likely that this disease is recent on the landscape and that few individuals have been exposed. Continued surveillance will be necessary to monitor this outbreak and determine what additional management actions may be needed to prevent CWD from becoming endemic in southeastern Minnesota.

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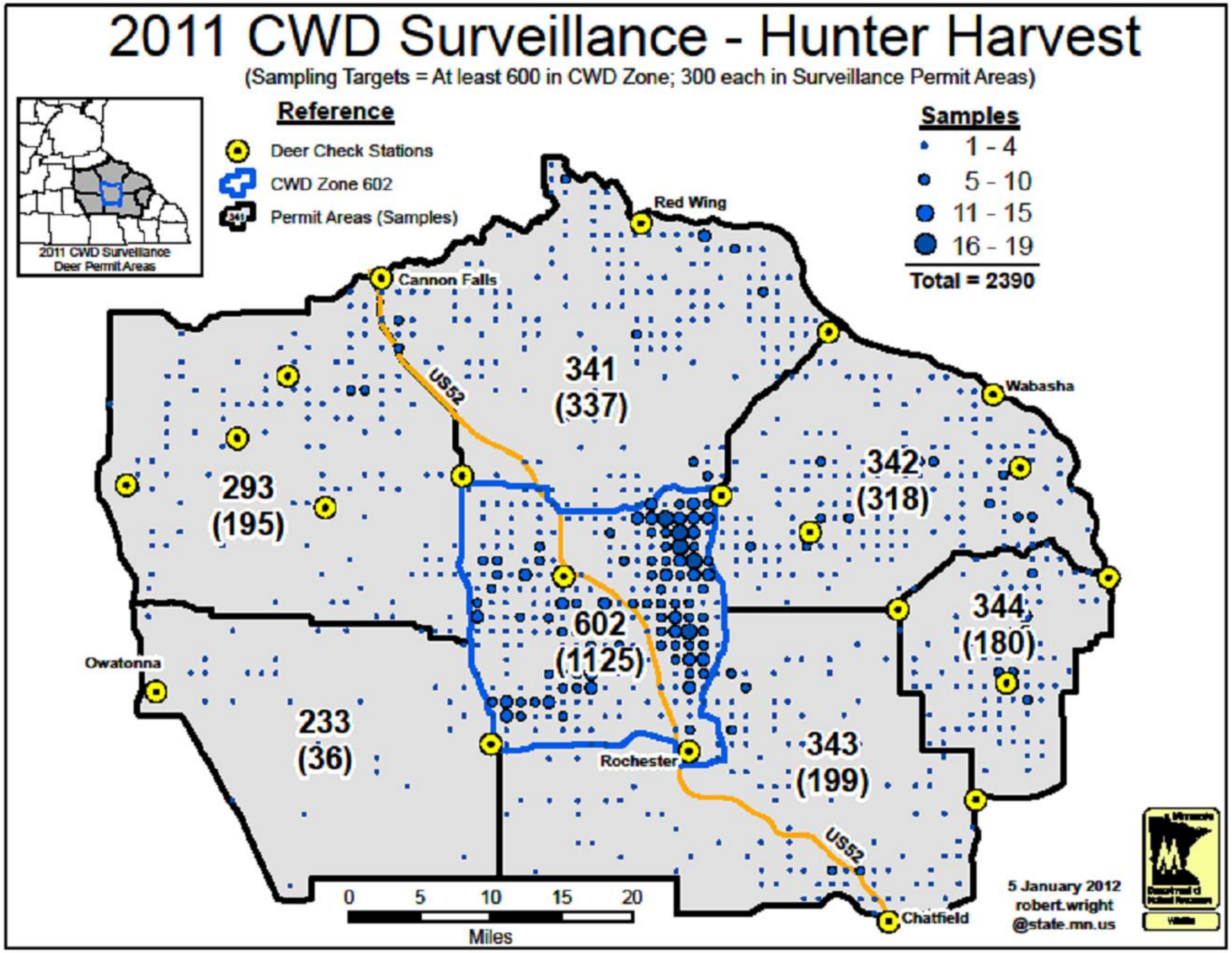

Figure 1. Sampling distribution of hunter-harvested deer (n=2,390) tested for chronic wasting disease in southeastern Minnesota, fall 2011.

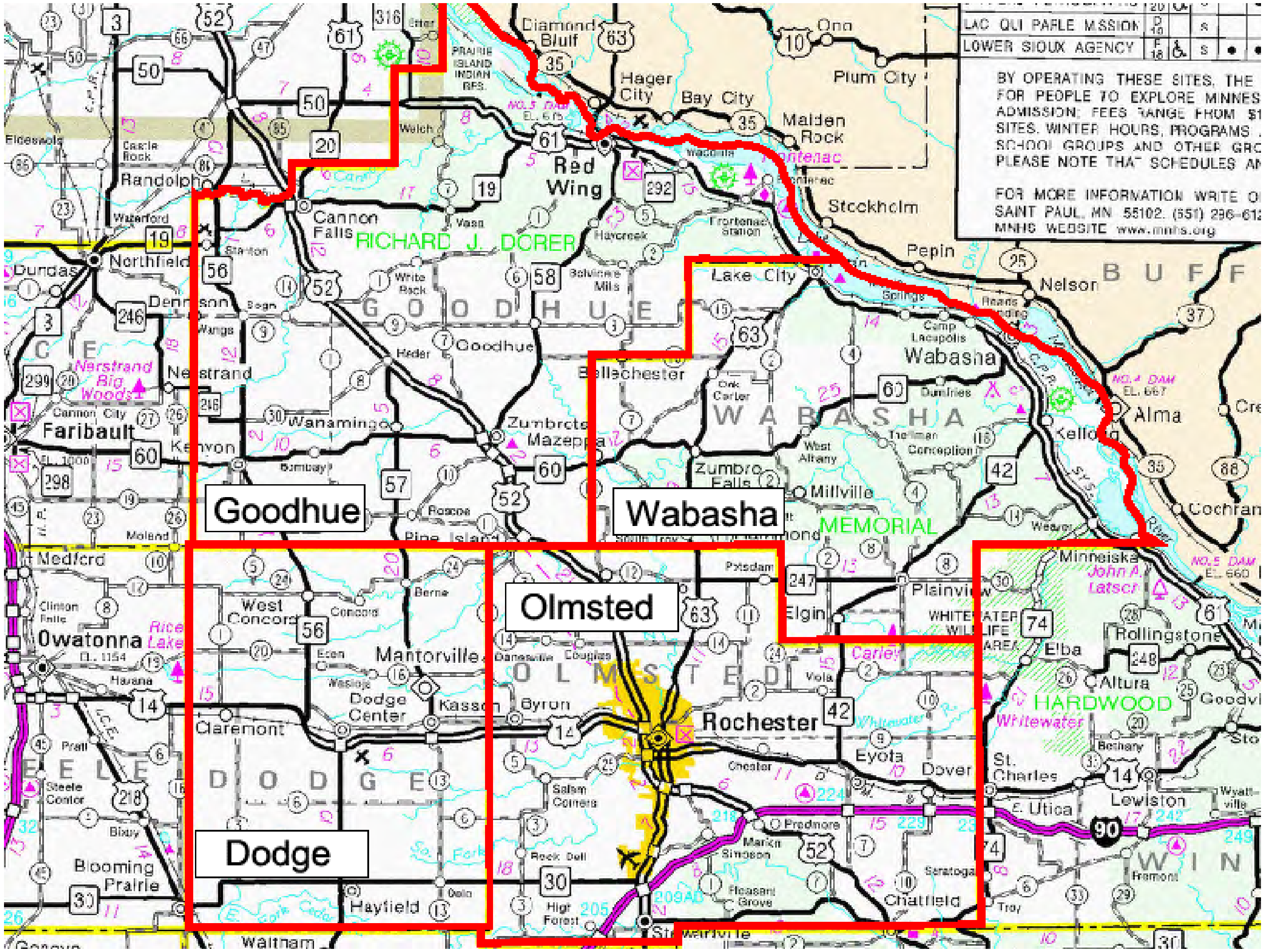

Figure 2. Four-county area in southeastern Minnesota where recreational feeding of wild white-tailed deer was banned in February 2011, following the discovery of chronic wasting disease in Olmsted County.

SPATIAL PATTERNS OF WHITE-TAILED DEER MOVEMENT RELATED TO BOVINE TUBERCULOSIS TRANSMISSION RISK IN NORTHWEST MINNESOTA

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SUMMARY OF FINDINGS

The goal of this pilot research study is to provide a better understanding of white-tailed deer (*Odocoileus virginianus*) movements and habitat use in the transitional landscape of northwestern Minnesota, where a recent outbreak of bovine tuberculosis heightened awareness of disease transmission risks between deer and cattle. In total, 21 deer (5 males, 16 females) were collared during this study and 10 deer (48%) remained alive until the planned collar blow-off date of April 15, 2012. Collar malfunctions occurred in 2 deer (10%), where no movement data were recorded. The overall mortality rate was 53% (n = 10), which was attributed primarily to wolves (n = 8, 80%), as well as hunter-harvest (n = 1, 10%) and unknown cause (n = 1, 10%). Mean home range size for deer (*n* = 9) surviving through the end of the study was 46.7 km^2 (SE = ±10.1). Seven deer were migratory, traveling 4–20 km to distinct winter ranges over 2-3 day periods. Deer visits occurred on 6 farms in the study area, with 1 farm accounting for 61% of the visits. Five deer accounted for all farm visits, including 2 deer visiting only one farm, 2 deer visiting two farms, and 1 deer visiting 3 farms. Over 75% of deer visits occurred in areas where cattle were present, either on a pasture or in an area with a feeding site and/or stored feed (hay bales). Most of the farm visits occurred during the spring (March through May) and primarily during the night (from 12am to 6am). This study provided baseline information regarding cattle-deer interactions critical to transmission of bTB in this region, and highlighted the potential for deer to function as vectors for disease transmission in transitional areas where habitat use between wildlife and livestock overlap.

INTRODUCTION

The Minnesota Department of Natural Resources (MNDNR) and the University of Minnesota (UMN) collaborated on a 15-month pilot study to gain a better understanding of movements and habitat use by white-tailed deer (*Odocoileus virginianus*) in northwest Minnesota. This is an area where continuous forest changes into a more agricultural landscape and deer use of this "transitional" habitat is not as well understood. The 2005 discovery of bovine tuberculosis (bTB) in wild deer in this area also increased concerns that a better understanding on how deer use such a diversified habitat is needed.

We were primarily interested in learning how deer use agricultural lands relative to state forest and wildlife management areas. In addition, we wanted to find out how farming practices, such as feed storage and animal husbandry, influenced deer use of agricultural lands. This project collected thousands of spatial locations of a small number of deer over the course of 15 months. By utilizing this information to improve our understanding of how deer may use farmed and pastured areas differently than natural habitats, we have gained insight into which practices may better minimize the risks of disease transmission between wild deer and cattle.

The UMN's Department of Veterinary Population Medicine previously developed a risk assessment process that was used by the Minnesota Board of Animal Health to evaluate the risk of deer and cattle interactions at farms within the bTB Management Zone (Knust et al. 2011). In this study, the UMN quantified the microhabitat use of deer on farms and the potential for bTB transmission among cattle and deer, and which herds are more at risk for deer-cattle interactions as a consequence of the farm management practices. Further, we hope to leverage the results obtained in this study with another ongoing study evaluating cattle

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movements in northwest Minnesota and potentially across the entire state. Combined, information generated from these studies should allow simulations of how bTB can spread across a network of farms, where disease is introduced by infected cattle and spread by deer as a transmission vector. The research should also provide a further understanding of steps that can be taken to mitigate these risks.

Secondarily, the location data ("fixes") stored on the radiocollars allowed MNDNR to estimate home range size and migration patterns for the study animals. Recognizing that the results will not adequately represent the larger deer population, our findings have provided wildlife managers and researchers with useful information and may help design a larger study in the future, should funding become available.

METHODS

The study area is approximately 360 km^2 and includes a mosaic of state forest and wildlife management lands, private recreational lands, and private farms (including row-crop agriculture, farmsteads, and stored forage). Included in the area are >25 farms with a variety of livestock and agricultural uses (Figure 1). The study area lies just outside the southern boundary of the bTB Management Zone and contains 2 formerly bTB-infected cattle farms; however, the disease has not been detected in wild deer in this area. Deer density ranged from 15-20 deer/km^2. Major predators include gray wolves (*Canis lupus*), black bear (*Ursus americanus*), coyote (*Canis latrans*), and bobcats (*Felis rufus*). Agricultural lands were surveyed to delineate and evaluate parameters (e.g., locations of stored forage, water sources, cattle pastures, etc.) that might attract deer to these areas.

In winter 2011, deer were captured by helicopter netgunning (Quicksilver Air, Inc., Fairbanks, Alaska) and Clover-trapped within the study area. Captured deer were chemically immobilized (100mg xylazine, 400mg ketamine HCl), and blood, urine, and fecal samples were collected for health screening. Methods for serological health screening were described in Carstensen et al. (2011). We also measured rump fat by ultrasound and extracted a last lower incisor to determine exact age by cementum annuli (Mattson's Laboratory, Milltown, Montana). Deer were ear-tagged and fitted with a satellite-linked radiocollar (ARGOS, SirTrack, Hawkes Bay, New Zealand). Body temperature was monitored at 5-min intervals throughout the processing period. A long acting antibiotic (LA-200, oxytetracycline) was administered intramuscularly (1 mL/10kg body weight). Before release, anesthesia was reversed by intravenous injection of 15mg/deer of yohimbine HCl. An observer monitored each deer's recovery and recorded the time deer were up and moving away from the recovery area.

Radiocollars were programmed to record locations every 90 minutes and transmit these "fixes" every 3 days through the ARGOS satellite system. Battery life of radiocollars is expected to be 15 months (to allow for one full year of seasonal movements). Collars were programmed to remotely discharge on April 15, 2012. The research team will retrieve all collars and download the complete set of spatial data. In the interim, fixes are downloaded weekly and examined for temporal and spatial movement patterns to determine mortality, movements, and habitat use. For any study animals that died during the study period, MNDNR wildlife staff investigated the cause of mortality, recovered the collar, and collected medial retropharyngeal lymph node samples from the deer (when possible) for bTB testing.

Deer movements and home range estimates were generated using Home Range Tools (HRT) for ArcGIS® (Rodgers et al. 2007). Minimum covex polygons (MCPs) were constructed by connecting peripheral points containing 99% of available fixes (White and Garrott 1990, Rodgers et al. 2007).

For evaluation of deer use on the agricultural landscape, a descriptive analysis was used to evaluate patterns of deer visits to farms during the entire study period. This will include the number of visits to each farm by season and time of day, number of farms visited by each individual deer, differences in use of farm areas by age and sex of deer, and variation in home range of each deer during the study period. Also, a resource utilization model will be developed that compares characteristics of locations used by each deer to available locations that are not used; thus, identifying higher risk areas for deer locations based on resource availability.

RESULTS AND DISCUSSION

Deer Capture and Handling

In January 2011, 16 deer (4 males, 12 females) were captured by helicopter netgunning within (*n* = 11) and slightly northeast (*n* = 5) of the study area. Capture locations were driven by deer distribution at the time of capture and access to private land to process deer. Due to collar failure immediately following release, one deer (ID 519) was censored from the study because no GPS fixes were transmitted, although its collar was recovered by timed blow-off and the deer remained alive through the end of the study. By the end of February 2011, 3 deer were killed by wolves and one died from unknown causes (Table 1). To compensate for the high winter mortality so early in the study, the sample size was augmented with 5 deer (1 male, 4 females) captured using Clover-traps in March 2011 (Table 1). One of these deer (ID 577) was fitted with a test collar provided by SirTrack (Iridium satellite system prototype), and this collar failed to record or transmit locations immediately after the animal's release. Although this deer was censored from the study, it was killed by wolves in early April and the collar was recovered. A second deer (ID 447) from this group slipped its collar (likely caused by a premature expulsion of the blow-off device) on 22 May, 2011, and was subsequently censored from the study.

In total, 21 deer (5 males, 16 females; 6 yearlings, 15 adults) were collared during this study and 10 deer (48%) remained alive until the planned collar blow-off date of April 15, 2012. Collar malfunctions occurred in 2 deer (10%), where no movement data were recorded. The overall mortality rate was 53% (n = 10), which was attributed primarily to wolves (n = 8, 80%), as well as hunter-harvest (n = 1, 10%) and unknown cause (n = 1, 10%). The collars functioned well, as weekly satellite downloads of these animals obtained approximately one-third of recorded fixes (Table 2). This provided sufficient data to track major animal movements and monitor survival, yet preserves battery life by restricting the amount of time collars communicated with the satellite system. The timed blow-off mechanisms worked perfectly for the 10 deer that survived to the end of the study. The success rate of obtaining fixes was >97% for recovered collars (Table 3).

The number of mortalities we observed from February to April 2011, specifically due to wolf predation, was higher than expected. Winter conditions were moderately severe (WSI = 159, Red Lake Wildlife Management Area) in the study area, with prolonged snow cover of >14 inches from late-January through early April. In Minnesota's forest zone, DelGiudice et al. (2006) reported a 37% winter mortality rate for adult deer during the severe winter of 1995-1996 (WSI = 195), with wolves accounting for 63% of those deaths. During more moderately severe winters (WSI = 124 to 159) in north-central Minnesota, DelGiudice et al. (2006) reported winter mortality rates ranging from 7 to 19%, with wolf predation accounting for 50-80% of the deaths. In contrast, the winter mortality rate for adult female deer in Minnesota's farmland zone has been reported as only 5%; however, there is an absence of wolves and typically more mild winter conditions (Brinkman et al. 2004). Little information exists on winter mortality rates for deer in Minnesota's transition zone, and although the sample size was limited in this study, our preliminary findings suggest there might be some unique attributes in northwestern Minnesota that make deer population dynamics different that both farmland and forest zones. Interestingly, during winter 2012, a historically mild winter with (WSI ≤ 20, Red Lake Wildlife Management Area), none of the remaining study animals were killed by wolves.

Disease Screening and Parasitology

Serological screening of deer at capture for 9 common cattle diseases indicated exposure to bovine parainfluenza 3 virus (24%), malignant catarrhal fever (19%), and infectious bovine rhinotracheitis (9%). Fecal parasitology indicted 13 (65%) of deer had evidence of liver fluke (*Fascioloides magna*) infection and strongyle-type ova was detected in 4 (20%) deer. Detailed discussion of these findings can be found in Carstensen et al. (2011).

Home Range Size and Deer Movements

Mean home range size for deer (*n* = 9) surviving through the end of the study was 46.7 km^2 (SE = ±10.1; Table 2, Figure 2). Deer that died (or slipped their collar) during the study had significantly smaller home range sizes to survivors ($13.9km^2 \pm 5.3$; Table 3). This apparent difference in home range size might be due to the fact that surviving deer had more than 4x the number of days on the air, thus were tracking movements over a longer time period.

Given the timing of deer capture (mid-January and early March), we assumed these animals were on their winter range (if migratory) or possibly year-round residents at the start of the study. This was an incorrect assumption, as movements to distinct winter ranges didn't occur until late-January or February. Seven deer had home ranges $>40km^2$ and can be attributed to a few long-distance movements from one end of their range to the other. These movements began in late January for 5 deer, moving 4-20 km in a 2-3 day period. The other 2 deer moved 14-19 km in mid to late March, again over a 2-3 day period. Of these 7 deer, 2 were killed during winter, but the other 5 returned the same distance (in a 2-3 day period of travel) to the area they were originally captured in late March or early April. Interestingly, only 3 of the 5 surviving migratory deer returned to their winter ranges during the mild winter of 2012; however, the start of their movement was much later (late February-early March) and they returned to their spring-summer-fall ranges sooner (mid-March).

Brinkman et al. (2005) reported 78% of deer in Minnesota's farmland zone as migratory (43% obligate and 35% conditional migrators), with a mean migration distance of 10 km. Further, those authors determined mean winter and summer home ranges (95% MCPs) as $5.2km^2$ and $2.6km^2$, respectively. Conversely, forest zone deer in northeastern and north-central Minnesota were 89% and 68% migratory, respectively (Nelson 1995, Fieberg et al. 2008). Further, migration distances were typically 10-14 km, but ranged from 2-135 km; onset of migrations varied annually, but ranged from early November to January (Fieberg et al. 2008). In both studies of forest zone deer, severe winters coincided with a higher number of conditional migrators making movements to a distinct winter range (Nelson 1995, Fieberg et al. 2008).

Deer Use of the Agricultural Landscape

Data on location of cattle, feeding areas and stored feed were collected by ground-truthing farm landscapes at 4 different times (December 2010 – before the capture; June 2011, October 2011 and March 2012) for the 30 farms within the study area. The farms within the study area are mostly small beef cow-calf operations. Primary variables of interest included locations of cattle, stored feed, and feeding sites.

Results show that deer visits occurred in 6 farms in the study area, with 1 farm accounting for 61% of the visits (Figure 3). Five deer accounted for all farm visits, including 2 deer visiting only one farm, 2 deer visiting two farms, and 1 deer visiting 3 farms (Figure 3). Over 75% of deer visits occurred in areas where cattle were present, either on a pasture or in an area with a feeding site and/or stored feed (hay bales) (Figure 4). Most of the farm visits occurred during the spring (March through May) and in the month of October (although the latter was performed mainly by 1 deer at 1 farm) (Figure 5). Deer visits increased during the crepuscular period achieving its maximum during the night (from 12am to 6am) (Figure 5).

These study results provide baseline information regarding cattle-deer interactions critical to transmission of bTB in this region, and highlight the potential for deer to function as vectors for disease transmission. The large home ranges for many of the study deer overlapped with multiple farms. In this study, 3 deer visited more than one farm which increases potential for disease transmission. Currently, the surveillance system for bTB is not cost-effective in situations of low disease prevalence. Time from infection to detection is extremely long, with the potential for severe consequences in terms of the spread of disease to other cattle herds and wildlife. Clearly there is need to improve both the sensitivity and cost-effectiveness of the surveillance system by detecting outbreaks faster and reducing the need for extremely costly control measures. When funding sources are allocated for such events, the resources need to be focuses toward the subset of the population that pose the highest risks. Further, the importance of risk mitigation and efforts to prevent of disease transmission between livestock and wildlife are often understated; enhancement and enforcement of appropriate biosecurity measures should be a priority within the agricultural community.

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Table 1. Current status and fate of free-ranging white-tailed deer (*n* = 21) captured and radiocollared in January and March 2011, northwest Minnesota.

Deer ID	**Capture Date**	**Method**	**Age Class**	**Age[1] (yr)**	**Sex[2]**	**Fate**	**Cause**	**Estimated Mortality Date**
469	1/15/11	Helicopter	Adult	4.5	F	Alive	collar blow-off	
461	1/15/11	Helicopter	Yearling	1.5	F	Dead	wolf-kill	3/31/11
497	1/15/11	Helicopter	Yearling	1.5	F	Alive	collar blow-off	
467	1/15/11	Helicopter	Yearling	1.5	M	Dead	wolf-kill	2/18/11
466	1/15/11	Helicopter	Adult	8.5	F	Alive	collar blow-off	
496	1/15/11	Helicopter	Adult	2.5	F	Dead	unknown	2/23/11
472	1/15/11	Helicopter	Adult	5.5	F	Alive	collar blow-off	
524	1/15/11	Helicopter	Adult	6.5	F	Dead	wolf-kill	3/10/11
473	1/15/11	Helicopter	Adult	4.5	M	Dead	unknown	7/28/11
495	1/15/11	Helicopter	Adult	2.5	M	Alive	collar blow-off	
471	1/15/11	Helicopter	Yearling	1.5	F	Dead	wolf-kill	4/5/11
491	1/16/11	Helicopter	Yearling	1.5	F	Alive	collar blow-off	
348	1/16/11	Helicopter	Adult	9.5	F	Dead	wolf-kill	2/12/11
460	1/16/11	Helicopter	Adult	2.5	F	Dead	wolf-kill	2/10/11
519	1/16/11	Helicopter	Adult	3.5	M	Alive	collar blow-off, malfunction[3]	
350	1/16/11	Helicopter	Adult	11.5	F	Alive	collar blow-off	
336	3/7/11	Clover-trap	Yearling	1.5	M	Dead	hunter-harvested	11/5/11
578	3/8/11	Clover-trap	Adult	4.5	F	Alive	collar blow-off	
577[4]	3/8/11	Clover-trap	Adult	11.5	F	Dead	wolf-kill	4/10/11
579[5]	3/8/11	Clover-trap	Adult		F	Alive	collar blow-off	
447	3/10/11	Clover-trap	Adult	3.5	F	Unknown	slipped collar	

[1] Age (in years) at capture was determined by cementum annuli. Analysis for adult deer captured in March is pending.

[2] F = female, M = male

[3] Deer 519's collar failed to transmit immediately after capture. No location data were obtained for this deer; however, it did survive through the study period.

[4] Deer 577 was fitted with a SirTrack test-collar (Iridium satellite system) and no movement data was recovered; mortality date is based on a public report of a severely injured deer and carcass remains.

[5] Deer 579 was unable to be aged due to a broken tooth with missing cementum.

Table 2. Fix success rates and home range size of free-ranging deer (n = 9) surviving through the end of the study, April 15, 2012, northwest Minnesota.

Deer ID	Days on Air	No. Fixes[1]	Fix Success Rate[2] (%)	99% MCP[3] (km^2)
469	457	7819	99.5	68
497	457	7717	98.3	92
466	457	7811	99.4	3
472	457	7772	98.7	33
495	447	7366	97.9	20
491	456	7785	99.0	76
350	456	7831	99.4	64
578	405	6909	98.0	46
579	405	7024	99.5	18
Mean	444	7559	98.9	46.7
SE	7	122	0.2	10.1

[1] Total number of fixes included only data downloaded from the satellite system from deployment through June 14, 2011.

[2] Fix success rate was calculated by number locations received through the satellite divided by the number of available locations, assuming collars recorded 16 locations/day.

[3] MCP = Minimum Convex Polygon, contained 99% of all locations.

Table 3. Fix success rates, and home range size of free-ranging deer (n = 10) that had either died or slipped their collar during the study.

Deer ID	Days on Air	No. Successful Fixes[1]	No. Failed Fixes	Success Rate (%)	99% MCP[2] (km^2)
461	77	1325	8	99.4	40.1
467	43	774	7	99.1	1.0
496	43	773	17	97.8	0.5
524	61	1124	4	99.6	8.0
471	90	1693	82	95.2	10.4
348	28	517	13	97.5	47.9
460	43	763	24	96.8	0.3
447	89	1641	68	95.9	4.1
473	200	3271	167	95.1	10.0
336	244	4807	125	97.5	17.0
Mean	92	1669	52	97.4	13.9
SE	23	429	18	0.5	5.3

[1] Total number of successful fixes included all data from deployment until collar was recovered from the field, which extended beyond the estimated mortality dates.

[2] MCP = Minimum Convex Polygon, contained 99% of all locations.

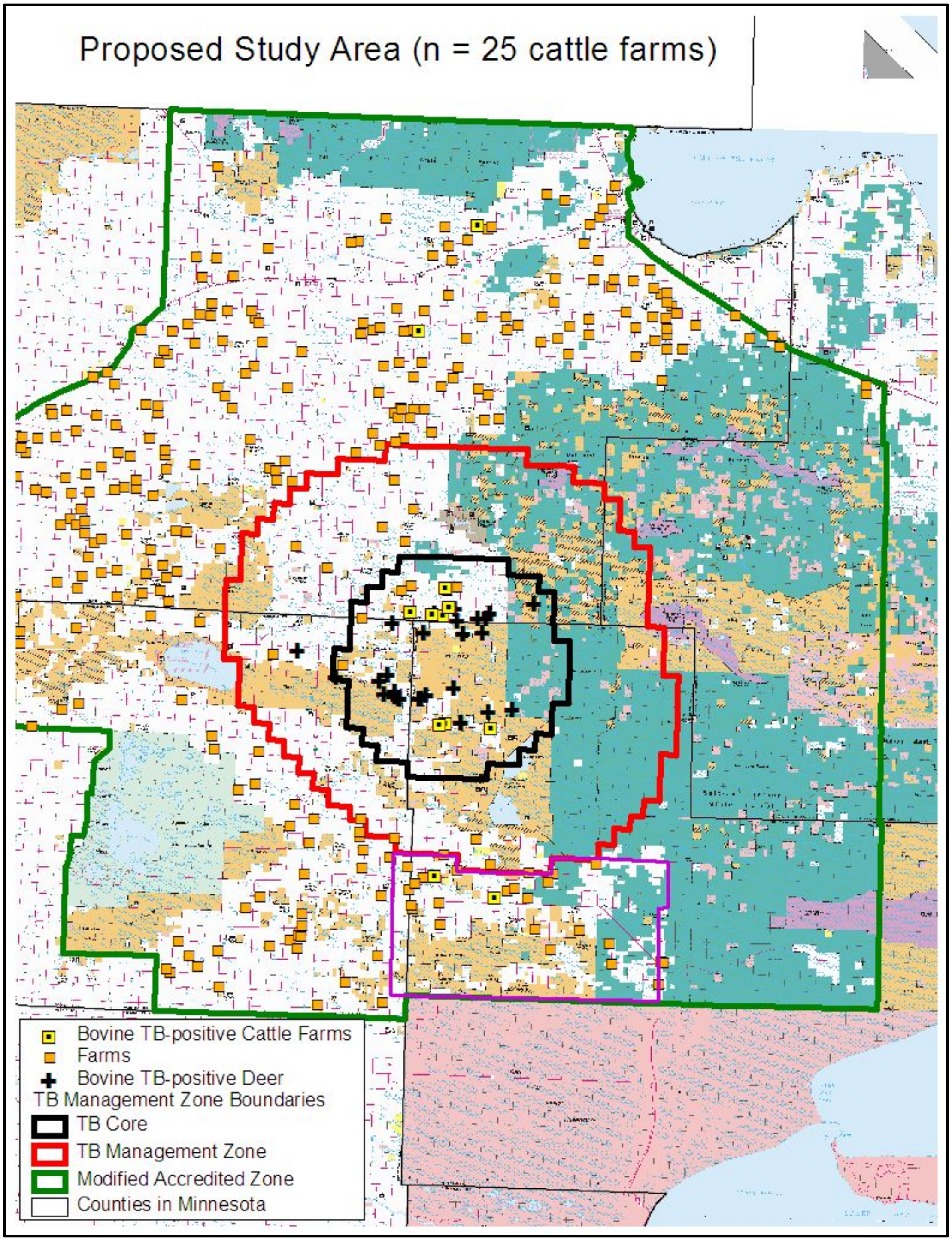


Figure 1. The 360km^2-study area (outlined in purple) contains >25 cattle farms including 2 previously infected with bovine tuberculosis. The study area is immediately south of Bovine Tuberculosis Management Zone, where 27 deer and 8 cattle farms tested positive for the disease.

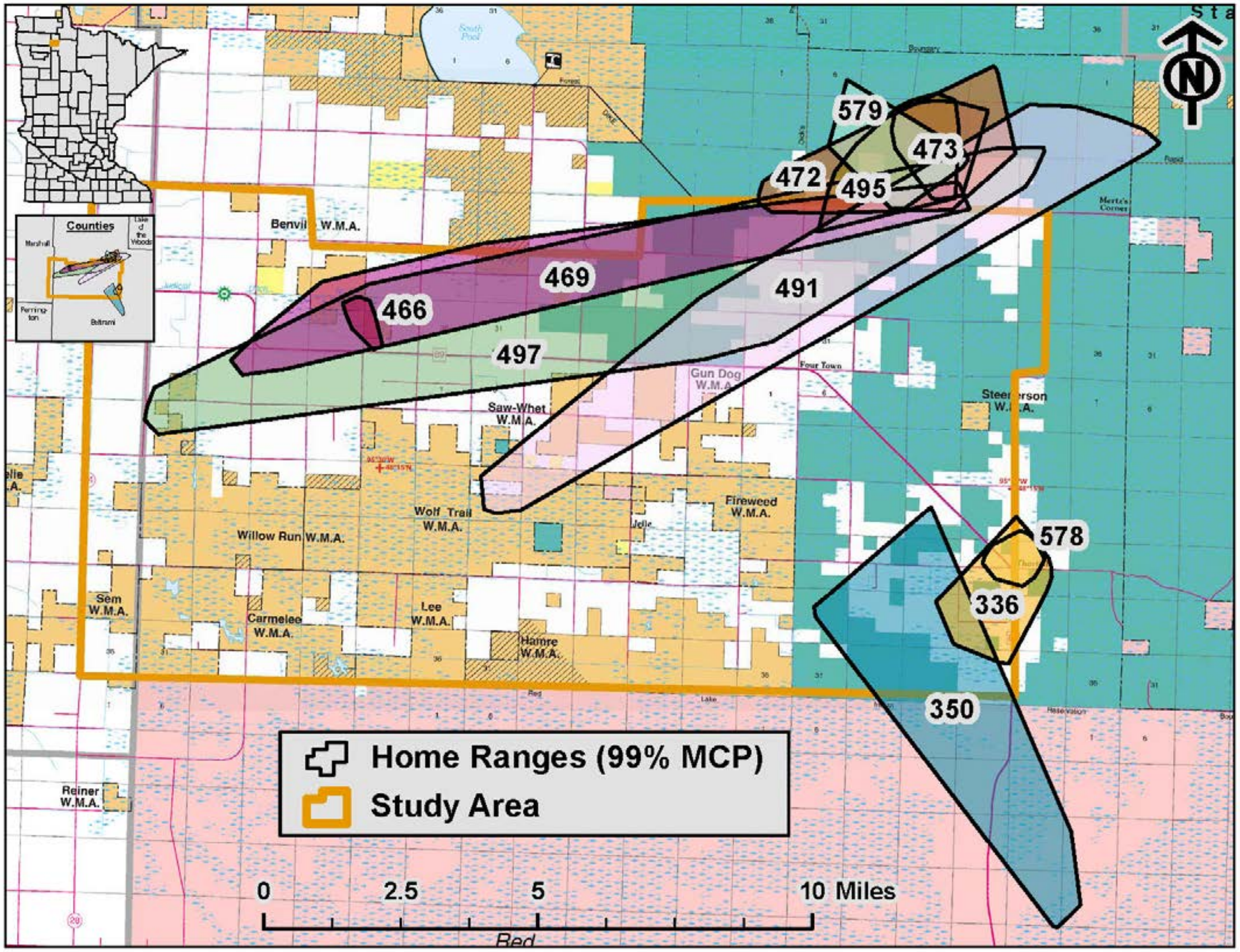


Figure 2. Home ranges, determined by 99% minimum convex polygons, for white-tailed deer (*n* = 11) that survived ≥ 200 days of the study, January 2011–April 2012, northwest Minnesota.

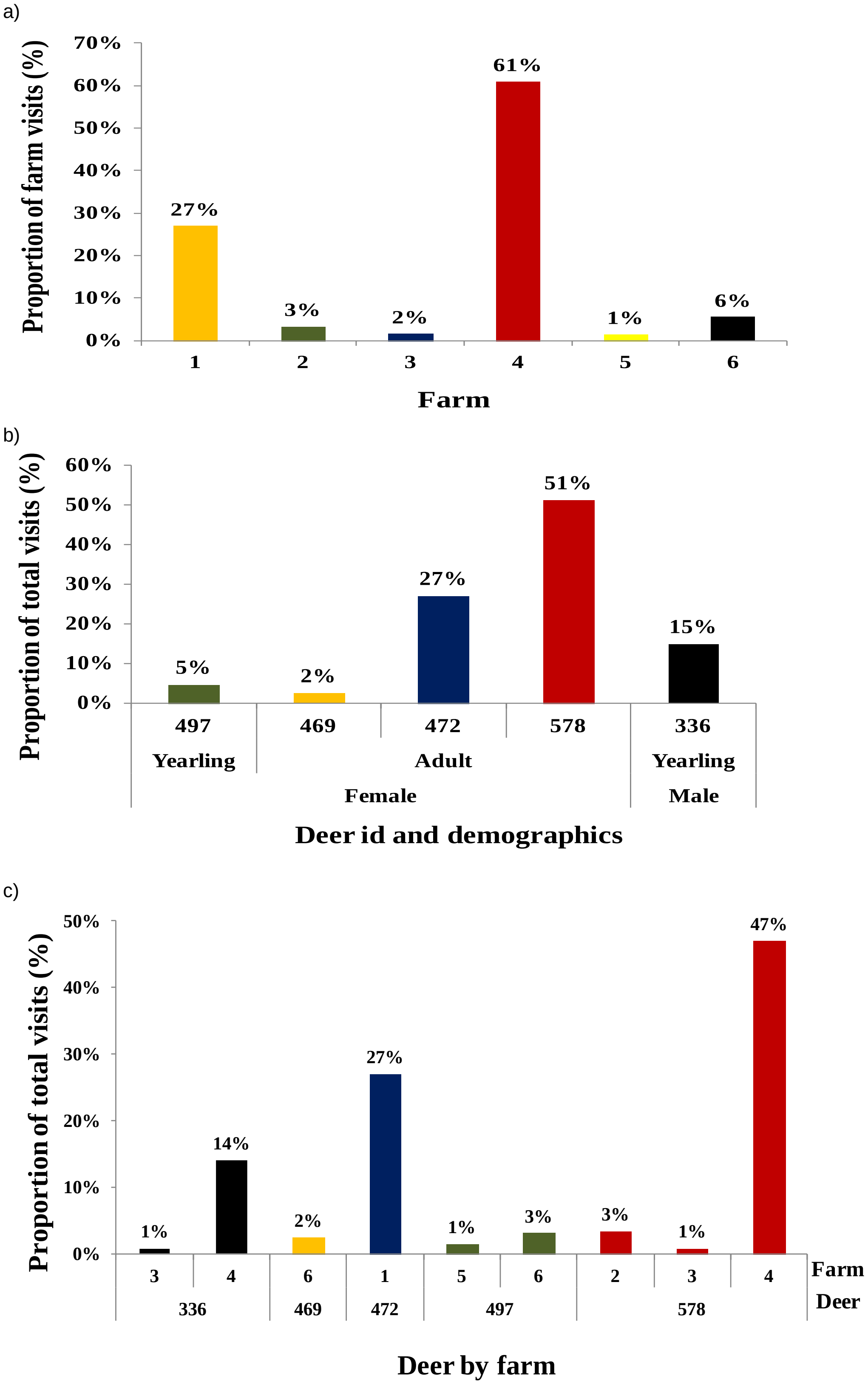


Figure 3. Six farms within the study area received visits by radio-collared deer during the study (a), the age/sex composition of those deer (*n* = 5), and the proportion of total visits by those deer per farm (c), northwest Minnesota.

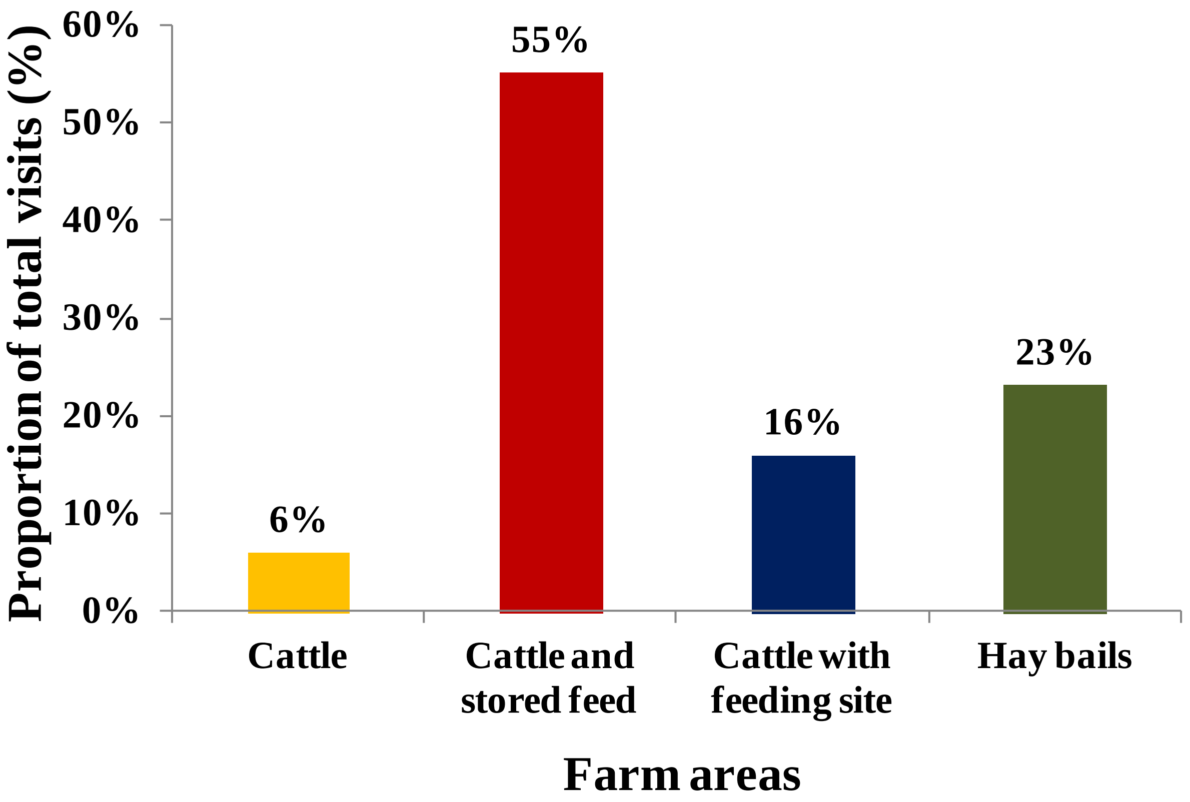
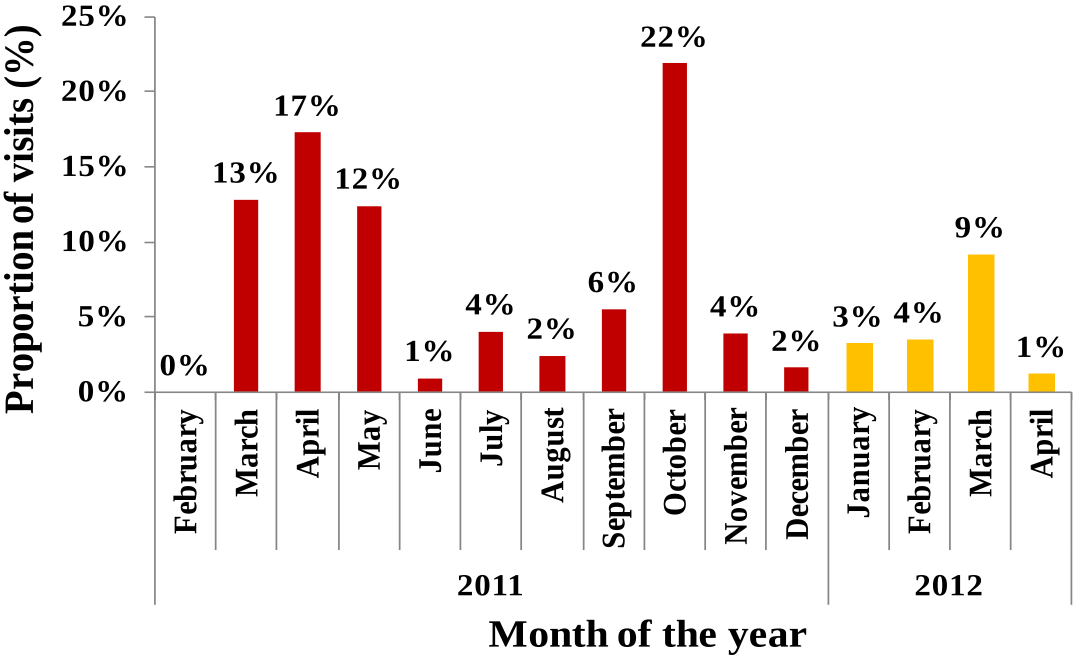


Figure 4. Attributes of farms within the study area and the proportion of their use by radio-collared Deer, January 2011–April 2012, northwest Minnesota.

a)



b)

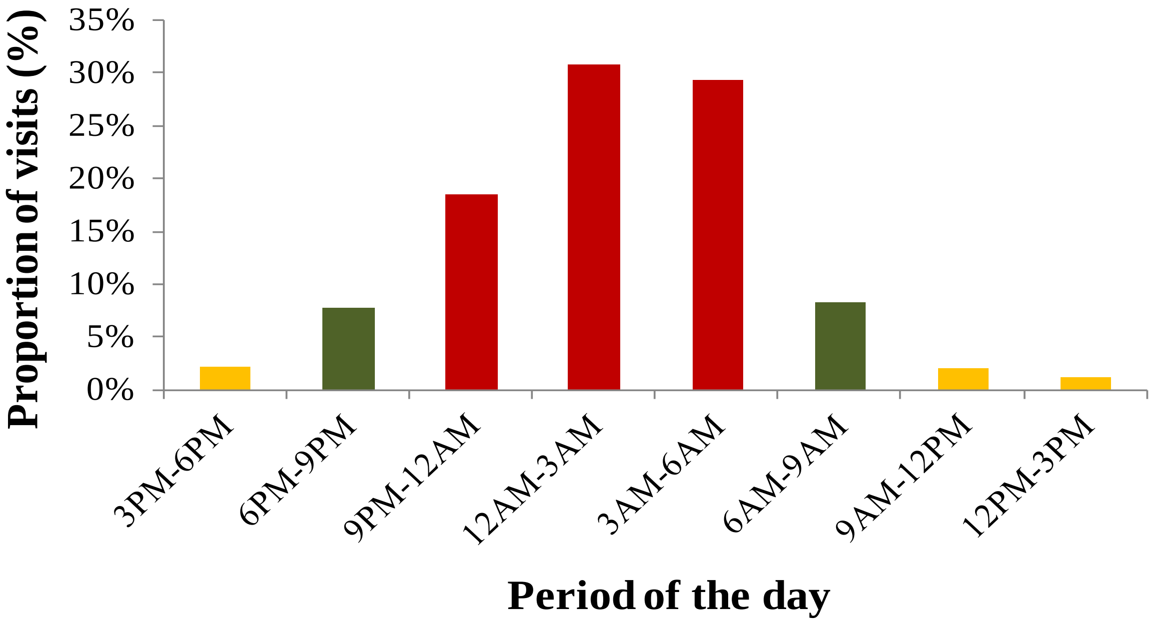


Figure 5. Monthly (a) and daily (b) use of farms by radio-collared deer (*n* = 19) from January 2011–April 2012, northwest Minnesota.

MANAGING BOVINE TUBERCULOSIS IN WHITE-TAILED DEER IN NORTHWEST MINNESOTA: A 2011 PROGRESS REPORT

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

SUMMARY OF FINDINGS

A total of 561 hunter-harvested white-tailed deer (*Odocoileus virginianus*) were tested for bovine tuberculosis (bTB) in northwest Minnesota during fall 2011, with no positive cases detected. This marked the 7th consecutive year that the Minnesota Department of Natural Resources (MNDNR) has conducted surveillance for this disease in deer since 2005, when bTB was first detected in a northwest cattle farm. The disease has since been found in a total of 12 cattle operations and 27 free-ranging white-tailed deer (*Odocoileus virginianus*). Both deer and cattle have the same strain of bTB, 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. No new infections have been detected in either cattle or deer since 2009. The state regained its bTB-Free accreditation in October 2011; however, some testing requirements remain on cattle herds within the endemic area. MNDNR plans to continue to monitor infection in the local deer population through hunter-harvested surveillance in fall 2012, and any further aggressive management actions (e.g., sharpshooting deer in key locations) will be dependent on future surveillance results.

INTRODUCTION

Bovine tuberculosis (bTB) is an infectious disease that is caused by the bacterium *Mycobacterium bovis*. Bovine tuberculosis primarily affects cattle; however, other mammals may become infected. The disease was first discovered in 5 cattle operations in northwest Minnesota in 2005. Since that time, 7 additional herds were found infected; resulting in a reduction of the state's bTB accreditation to Modified Accredited in early 2008. In fall 2008, Minnesota was granted a split-state status for bTB accreditation that maintained only a small area (2,670mi^2) in northwest 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 northwest Minnesota, which can be attributed to a spillover of the disease from infected cattle. In 2010, The United States Department of Agriculture (USDA) upgraded Minnesota's bTB accreditation to Modified Accredited Advanced within the split-state zone and bTB-Free throughout the remainder of the state. With no new infections discovered in MN cattle in 2009 and 2010, USDA upgraded the split-state portion to bTB-Free in October 2011. Although bTB 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 bTB in North America. In 1995, bTB was detected in wild deer in Michigan and do serve as a reservoir of the disease in that state.

Bovine tuberculosis 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 bTB by ingesting the bacteria from contaminated feed. Incubation periods can vary from 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 throughout the thoracic cavity. In severely infected deer, lesions can usually be found throughout the animal's entire body.

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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. While it is possible to transmit bTB 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 2011, a fall hunter-harvested surveillance strategy was developed to collect 500 samples from the bTB Management Zone, which is approximated by Deer Management Area (DMA) 101. When MN regained its bTB-Free accreditation in October, the existing Memorandum of Understanding (MOU) with USDA, signed by both MNDNR and BAH, was no longer in effect. To that end, MNDNR and USDA renegotiated a sampling scheme that would still satisfy our commitment to ensuring the disease is not present in wild deer within the bTB Management Zone at >1.0% with 99% confidence.

At the registration stations, hunters were asked to voluntarily submit lymph node (LN) samples for bTB testing. Hunter information was recorded, including the hunter's name, 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 (MDHA). In addition, the Roseau River chapter of MDHA raffled additional firearms and a life-time deer hunting license for hunters that submitted samples from within the bTB Management Zone or bTB Core Area.

Sampling procedures included a visual inspection of the chest cavity of the hunter-killed deer. Three pairs of cranial LNs (parotid, submandibular, and medial retropharyngeal) were visually inspected for presence of gross lesions and collected for further testing. Samples were submitted to the Veterinary Diagnostic Laboratory (VDL) at the University of Minnesota for histological examination and acid-fast staining (when lesions are present only). 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 voluntarily surrendered 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.

In early winter, MNDNR conducted an aerial survey of the bTB Core Area to assess deer numbers and distribution (Figure 1). This information was used to guide future management activities and estimate the percentage of deer removed from the area through hunting and agency culling.

RESULTS AND DISCUSSION

In fall 2011, we collected 561 samples from hunter-harvested deer; 349 samples from within the bTB Management Zone, including 151 samples from within the bTB Core Area (Figure 2). MNDNR collected approximately 70% of the sampling goal from within the bTB Management Zone; however exceeded the overall sampling goal by 12% when including deer tested just outside this zone.

Testing of all lymph node samples at NVSL has confirmed that there were no positive cases of bTB detected during the fall 2011 surveillance. Thus, 2011 marks the second consecutive year in which no new cases of the disease were detected in wild deer. Apparent prevalence of bTB in the local deer population, sampled throughout a 1,730 to 2,670mi^2 Surveillance Zone, indicates a significant decreasing trend from 2006–2011 (Table 1, Figure 3). Further, disease prevalence in the bTB 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 bTB-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 2011 estimated that the deer population in the bTB Core Area was a minimum of 160 ± 45 deer (Figure 1). This was markedly lower than the 2011 population estimate of 531 ± 48 (Figure 4, Table 2). This was surprising as winter deer removal efforts have been suspended for 2 years and a lag effect from those operations would be unexpected. However, winter conditions in 2011 were moderately severe in the northwest and over-winter deer survival may have been adversely impacted. In a pilot study involving 16 radio-collared deer south of the bTB Management Zone, 50% of the deer were killed by predators during winter 2011. Further, winter movements of deer are highly influenced by winter weather conditions. It is likely that the bTB 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 bTB Core Area as deer numbers are reduced and habitat availability increases. The extremely mild winter of 2012 likely played a role in decreased deer movement into the bTB Core Area, which provides good wintering habitat, and might explain the decrease in estimated deer numbers. Lastly, snow conditions during the March 2012 survey were generally poor and deer visibility may have been compromised.

The proximity of the bTB-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 10,344 deer in the northwest, and a total of 27 confirmed culture-positive deer (Figure 5). Further, the lack of infected yearlings or fawns and limited geographic distribution of infected adults further supports that deer are not a wildlife reservoir for this disease in Minnesota (Carstensen and DonCarlos, 2011).

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 6). Enforcement officers continue to enforce this rule and compliance is very high within the bTB Management Zone.

With the recent upgrade in status to bTB-Free across the state and a lack of available funding to continue support payments to farms that participated in the buy-out program, BAH has announced that farms will be allowed to repopulate with cattle within the bTB Management Zone beginning July 1, 2012. Although farmers will no longer be required to obtain permits or test individual animals prior to moving cattle, whole-herd testing within the bTB Management Zone will continue. MNDNR will conduct fall hunter-harvested surveillance in 2012, with a sampling scheme similar to what occurred in 2011. If no new cases of bTB are detected in wild deer, the surveillance effort will be suspended. A recheck in 2015 is possible if funding can be identified.

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, Paul Telander, and John Williams. For all the help with field collections, we'd like to thank area staff from Thief Lake, Red Lake, Roseau River, and Thief River Falls, as well as students and faculty from the University of Minnesota, College of Veterinary Medicine. Also thanks to Bob Wright, Erik Hildebrand, and John Heineman for conducting a deer survey within the bTB Core Area. Also thanks to John Giudice for analyzing the survey data. GIS support was provided by Julie Adams, Bob Wright, and Chris Scharenbroich. A special thanks to David Kuehn for agreeing to temporarily leave retirement for a few weeks of continuous deer surveillance work.

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Table 1. Number of deer sampled for bovine tuberculosis (bTB) and testing results listed by sampling strategy, 2005 to 2011, northwestern Minnesota.

Sampling strategy	2005	2006	2007	2008	2009	2010	2011	Totals
Hunter-harvested (Oct-Jan)	474	942	1,166	1,246	1,488	1,639	561	7,516
# bTB-positive	1	5	5	0	1	0	0	
Apparent prevalence	0.21%	0.53%	0.43%	0.0%	0.07%	0.0%	0.0%	
Sharpshooting (Feb-May)	n/a	n/a	488	937	738	450	n/a	2,613
# bTB-positive			6	6	2	0		
Apparent prevelance			1.23%	0.64%	0.27%	0.0%		
Landowner/tenant	n/a	90	n/a	125	n/a	n/a	n/a	215
# bTB-positive		1		0				
Total deer tested	474	1,032	1,654	2,308	2,226	2,089	561	10,344
Total # bTB-positive	1	6	11	6	3	0	0	27

Table 2. Population estimates[a] and 95% confidence intervals[b] of deer within the Bovine Tuberculosis Core Area, 2007–2012, northwest Minnesota.

Year	Aircraft	Design	Var.est	n	N	Srate	Svar	SE	Xbar	SE	95%CI	**PopEst**	SE	**95% CI**	CV(%)	RP(%)
2007	OH-58	StRS3	SRS	72	164	0.439	NA	NA	5.7	0.46	4.9-6.5	**935**	76.0	**784-1086**	8.1	16.2
2008	OH-58	GRTS.SRS	Local	72	164	0.439	21.94	4.53	4.9	0.56	3.8-6.0	**807**	75.2	**659-954**	9.3	18.3
2009	Enstrom	GRTS.stRS2	Local	79	164	0.482	20.63	2.56	4.1	0.27	3.5-4.6	**664**	44.4	**577-751**	6.7	13.1
2010	OH-58	GRTS.SRS	Local	72	164	0.439	29.30	6.70	2.6	0.39	1.8-3.3	**422**	64.4	**296-548**	15.3	30.0
2011	OH-58	GRTS.SRS	Local	72	164	0.439	21.01	2.80	3.2	0.30	2.7-3.8	**531**	48.6	**436-627**	9.2	18.0
2012	OH-58	GRTS.SRS	Local	72	164	0.439	3.06	0.57	1.0	0.14	0.7-1.3	**160**	22.3	**120-210**	13.6	26.7

[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** based on sampling variance only (adjusted for spatial correlation in 2008-2011); they do not include uncertainty associated with detectabilty or animal movements (temporal variation due to animals moving onto or off the study area).

Aerial Survey of Deer in the Core Area of the Bovine TB Management Zone March 5th - 7th, 2012

Legend

- 6 Surveyed Plots and Totals Observed
- ● Deer Observations
- ● TB-Positive Deer
- ▣ TB-Positive Farms
- ★ Towns
- Township Roads
- County Roads
- State Highways
- County Boundaries
- 🟥 Core Area
- ■ Management Zone
- ■ Red Lake Reservation
- 🟩 Public Lands

About This Map

Randomly selected PLS sections in the Bovine TB Management Zone were surveyed via helicopter to estimate the deer population of the Core Area, where management efforts are focused.

Deer per section ranged from 0 to 8, averaged 1.0 and summed to 72 for 72 sections. Using this information, the population in the Core Area is estimated to be at least **160 +/- 45 deer**. This is necessarily a minimum estimate because the number of deer undetected during the survey is unknown.

0 0.5 1 2 3 4 5

Miles

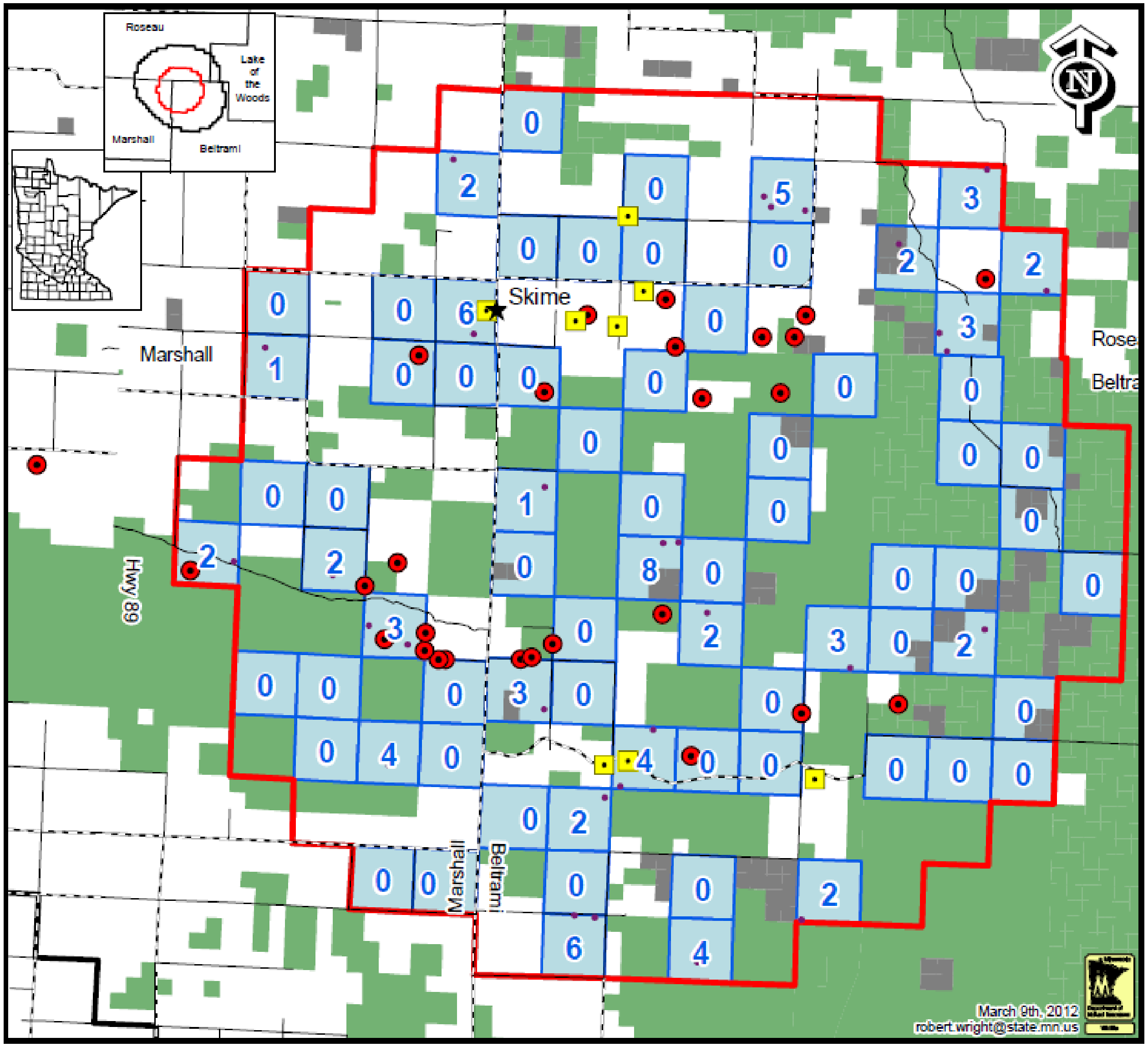


Figure 1. Results of aerial white-tailed deer survey of the Bovine Tuberculosis Core Area in March 2012, northwest Minnesota.

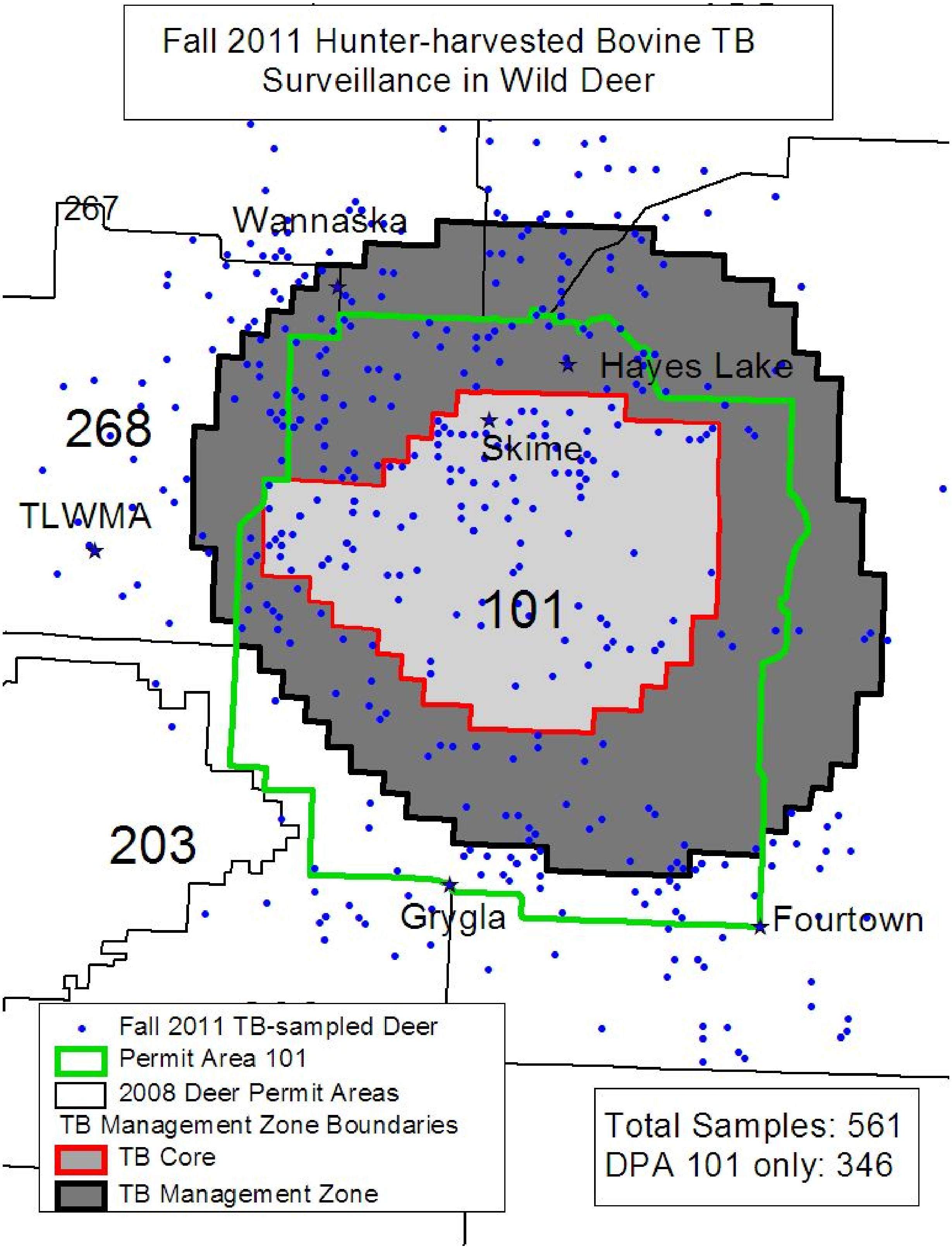
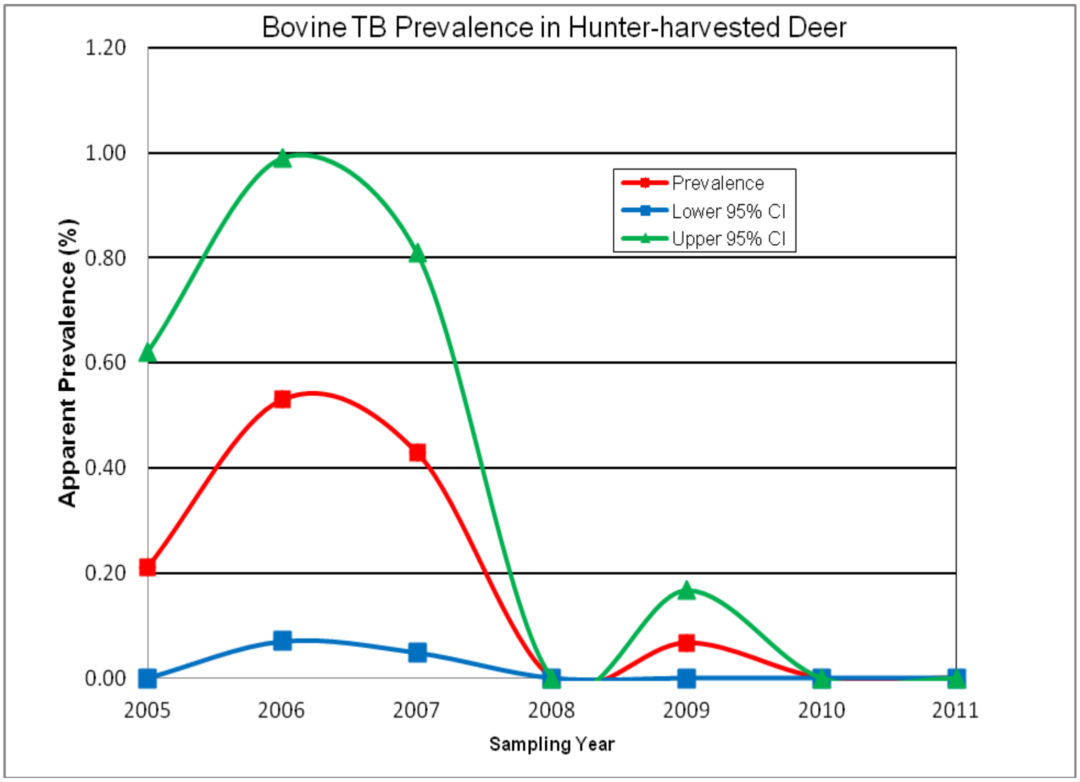


Figure 2. Locations of hunter-harvested deer (*n*=561) sampled for bovine tuberculosis (bTB) during fall 2011 in northwest Minnesota.



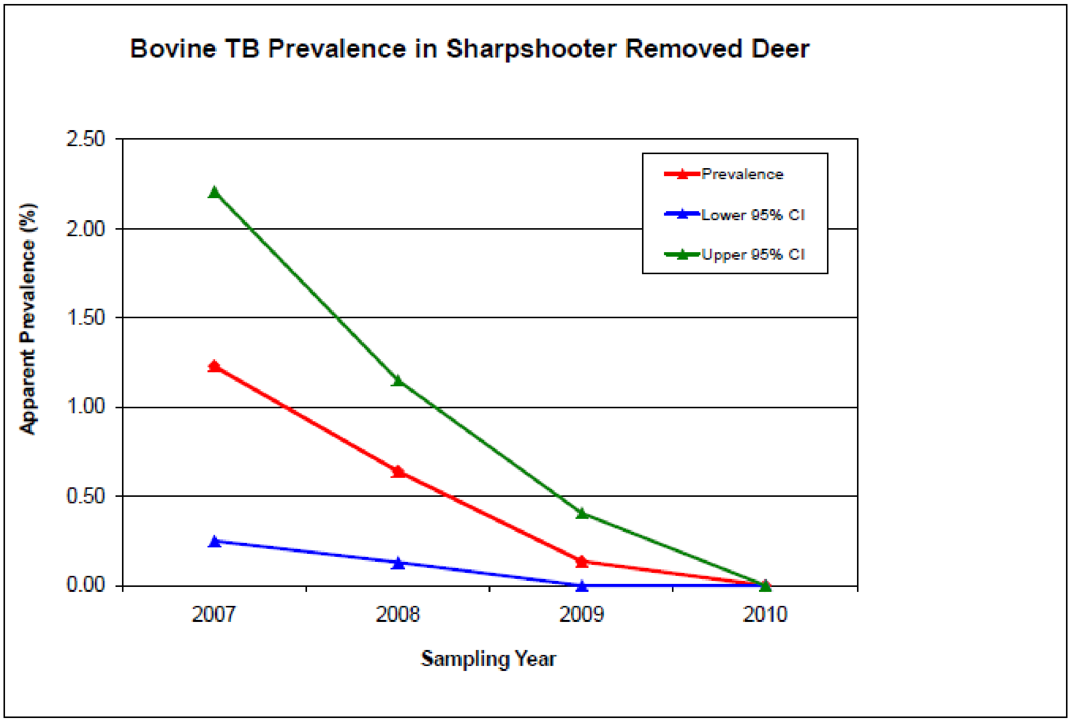


Figure 3. Prevalence of bovine tuberculosis (bTB) in hunter-harvested deer from 2005–2011 in the bTB Surveillance Zone and disease prevalence from sharpshooter removed deer from 2007–2010 in the bTB Core Area, northwest Minnesota.

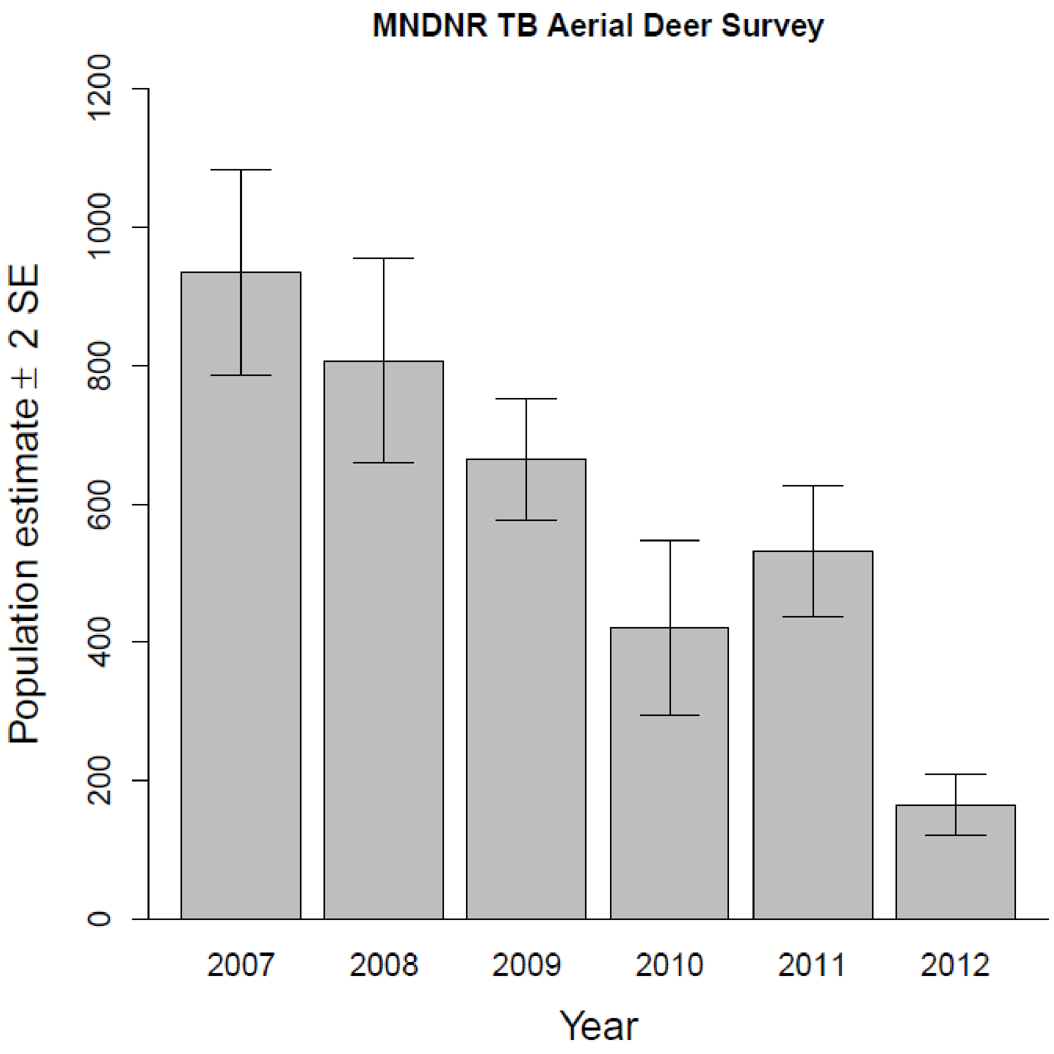


Figure 4. Population estimate of deer within the Bovine Tuberculosis Core Area, winters 2007–2012, northwest Minnesota.

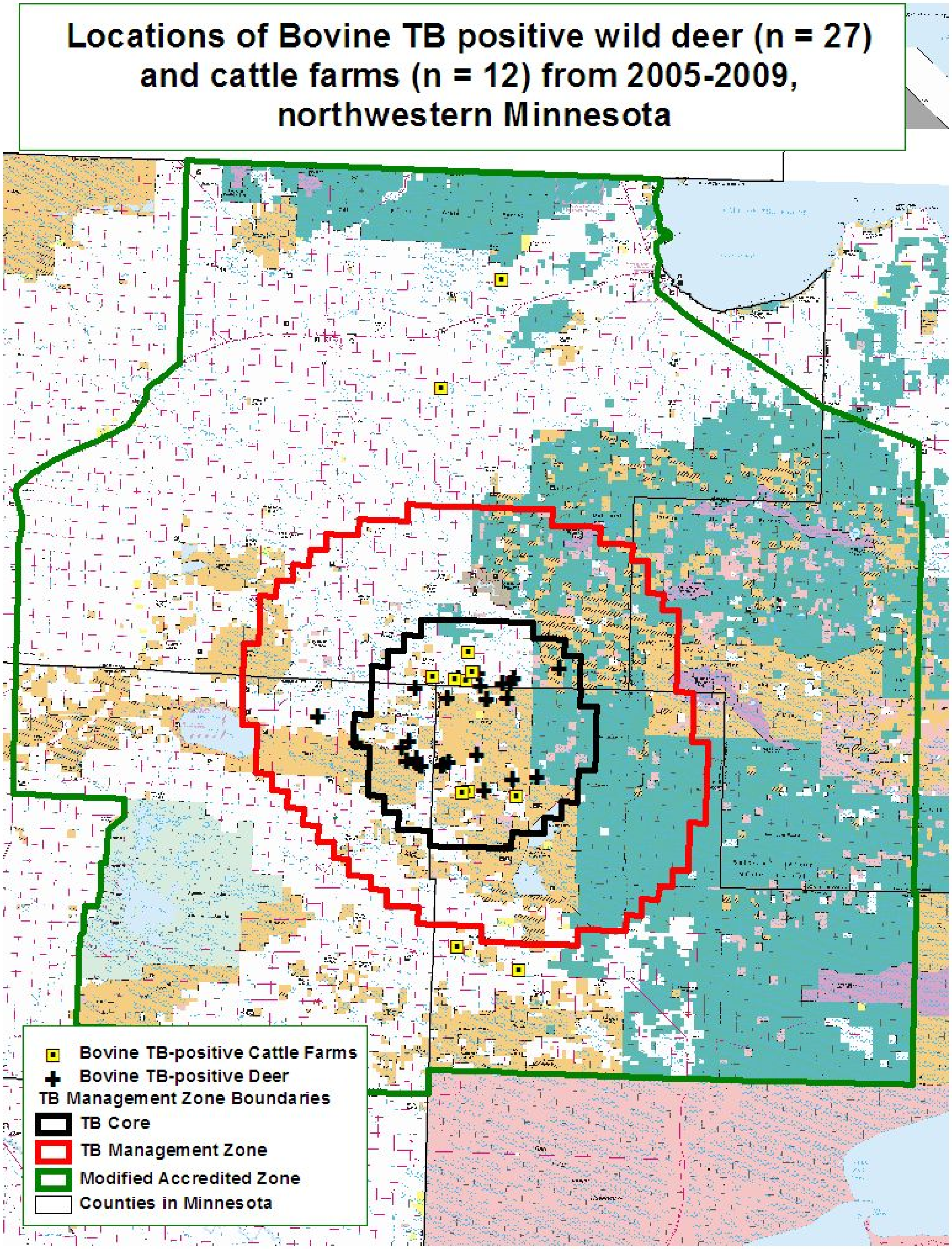
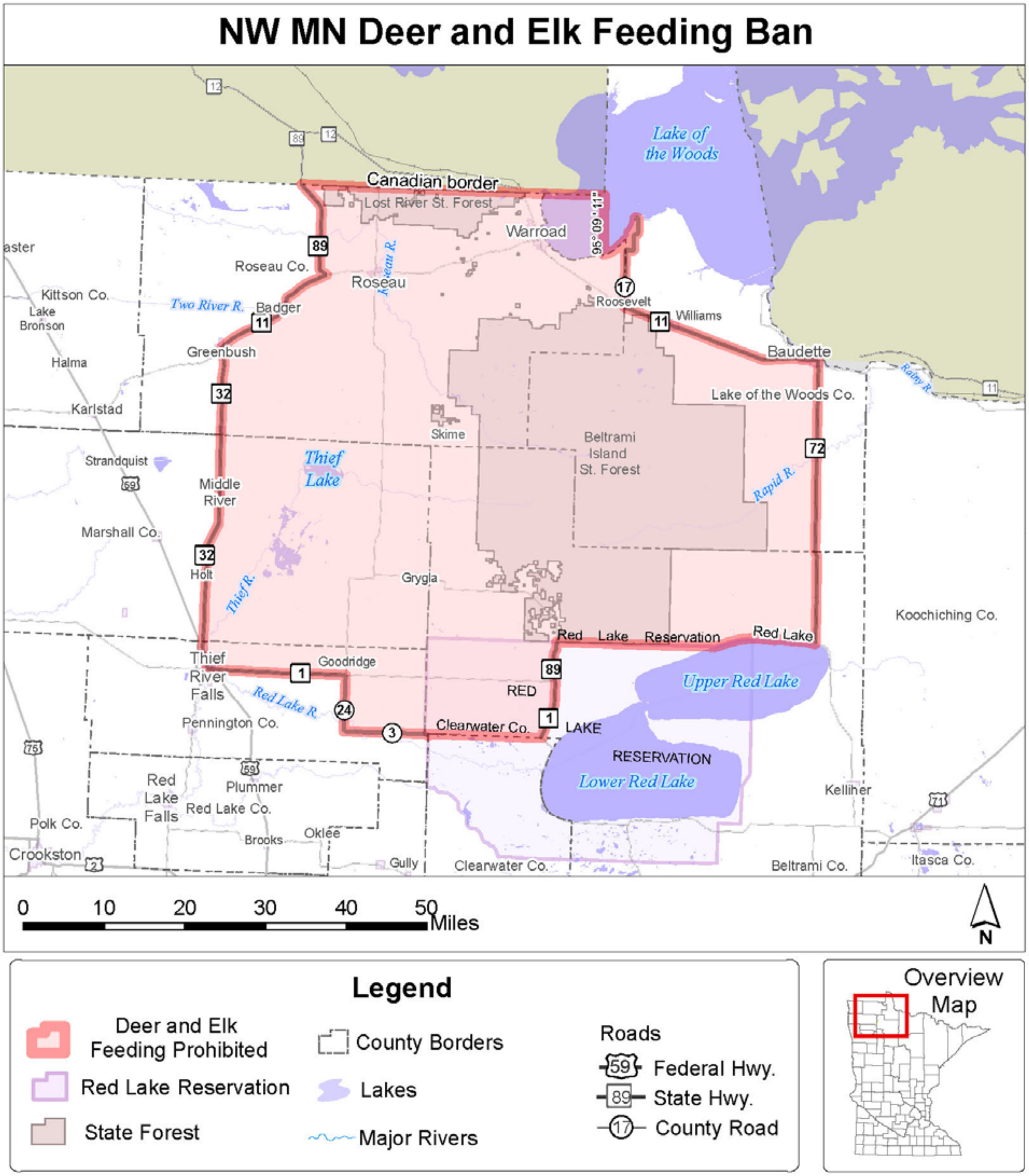


Figure 5. Locations of all white-tailed deer found infected (*n*=27) with bovine tuberculosis (bTB) since fall 2005 in northwest Minnesota, with the 12 previously-infected cattle operations are also included.



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 6. Area in northwest Minnesota where recreational feeding of deer and elk was banned in November 2006, as a preventative measure to reduce risk of disease transmission.

Biometrics Unit and Surveys

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(651) 296-5200

New Ulm Regional Office
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New Ulm, Minnesota 56073
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ESTIMATING ANIMAL ABUNDANCE WITH A HIERARCHICAL CATCH-EFFORT MODEL[1]

Katherine St. Clair[2], **Eric Dunton**[3], and **John Giudice**.

SUMMARY OF FINDINGS

The primary objective of this paper is to compare methods for modeling the probability of removal when variable amounts of removal effort are present. A hierarchical modeling framework can produce estimates of animal abundance and detection from replicated removal counts taken at different locations in a region of interest. A common method of specifying variation in detection probabilities across locations or replicates is with a logistic model that incorporates relevant detection covariates. As an alternative to this logistic model, we propose using a catch-effort model to account for heterogeneity in detection when a measure of removal effort is available for each removal count. This method models the probability of detection as a nonlinear function of removal effort and a removal probability parameter that can vary spatially. Simulation results demonstrate that our model is effective in estimating abundance and removal probability. We also found that our catch-effort model fits better than logistic models when estimating wild turkey abundance using harvest and hunter counts collected by the Minnesota Department of Natural Resources during the spring turkey hunting season.

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PROBABILITY OF DETECTION IN CROWING SURVEYS OF RING-NECKED PHEASANTS[1]

Alison Harwood[2,3], **Kurt Haroldson**, Brock McMillan[4], and **John Giudice**.

SUMMARY OF FINDINGS

There currently is no reliable and cost-effective population estimator for ring-necked pheasants (*Phasianus colchicus*). Consequently, most pheasant monitoring programs rely on population indices (e.g., roadside counts or crowing indices). The validity of using indices to make inferences about population size is based on the assumption that detection probability (*p*) is constant or, more realistically, variation in *p* among comparison groups is small relative to the variation in population size sought to be detected. We applied time-of-detection methods to replicate (within season) pheasant-crowing surveys conducted on 18 study sites in southern Minnesota in 2007 to estimate $\bar{p}$, $\mathrm{var}(\hat{\bar{p}})$, and σ^2 (random spatial variation). We also conducted a Monte Carlo simulation to examine the bias-variance tradeoff associated with using a time-of-detection model to estimate and adjust for non-response bias. More specifically, we used estimates of $\bar{p}$, $\mathrm{var}(\hat{\bar{p}})$, and σ^2 to simulate variation in replicated pheasant-crowing counts on 18 study sites where true population size was a positive function of percent undisturbed grasslands. Estimated mean detection probability in our study was 0.533 (SE = 0.030) and $\hat{\sigma}^2$ was 0.081 (95% CI: 0.057–0.126). On average, both adjusted (for $\hat{p}$) and unadjusted counts of crowing males qualitatively described the simulated relationship between pheasant abundance and grassland abundance. However, using a time-of-detection model to estimate and adjust for $\hat{p}$ produced, on average, nearly unbiased (0.008) estimates of β_1 (the slope of the simulated pheasant-grassland relationship). Conversely, using unadjusted counts tended to result in a negatively biased estimate of β_1 (–0.206). Adjusted counts were more variable than unadjusted counts (IQR = 2.8 vs. 1.8), but MSE (a measure of bias-variance tradeoff) was smaller for adjusted counts (MSE = 0.003 vs. 0.045). These findings support using time-of-detection methods to estimate and adjust for non-response bias in replicated pheasant crowing surveys.

---

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COMPARING STRATIFICATION SCHEMES FOR AERIAL MOOSE SURVEYS[1]

John R. Fieberg and **Mark S. Lenarz**

ABSTRACT

Stratification is generally used to improve the precision of aerial surveys. In Minnesota, moose (*Alces alces*) survey strata have been constructed using expert opinion, informed by moose density from previous surveys (if available) and recent disturbance and cover-type information. Stratum-specific distributions of observed moose from plots surveyed during 2005-2010 overlapped, suggesting some improvement in precision might be accomplished by using a different stratification scheme. We explored the feasibility of using remote-sensing data to define strata. Stratum boundaries were formed using a 2-step process: 1) we fit parametric and non-parametric regression models using land-cover data as predictors of observed moose numbers; 2) we formed strata by applying classical rules for determining stratum boundaries to the model-based predictions. Although land-cover data and moose numbers were correlated, we were unable to improve upon the current stratification scheme based on expert opinion.

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[1] Abstract from paper accepted for publication in *Alces*.

COULD YOU PLEASE PHRASE "HOME RANGE" AS A QUESTION?[1]

John Fieberg and Luca Börger[2]

ABSTRACT

Statisticians frequently voice concern that their interactions with applied researchers start only after data have been collected. The same can be said for our experience with home-range studies. Too often, conversations about home range begin with questions concerning estimation methods, smoothing parameters, or the nature of autocorrelation. More productive efforts start by asking good (and interesting) research questions; once these questions are defined, it becomes possible to ask how various design and analysis strategies influence one's ability to answer these questions. With this process in mind, we address key sample design and data analysis issues related to the topic of home range. The impact of choosing a particular home-range estimator (e.g., minimum convex polygon, kernel density estimator, local convex hull) will be question dependent, and for some problems other movement or use-based metrics (e.g., mean step lengths, time spent in particular areas) may be worthy of consideration. Thus, we argue the need for more question-driven and focused research and for clearly distinguishing the biological concept of an animal's home range from the statistical quantities one uses to investigate this concept. For comparative studies, it is important to standardize sampling regimes and estimation methods as much as possible, and to pay close attention to missing data issues. More attention should also be given to temporally changing space-use patterns, with biologically meaningful time periods (e.g., life history stages) used to define sampling periods. Lastly, we argue the need for closer connections between theoretical and empirical researchers. Advances in ecological theory, and its application to natural resources management, will require carefully designed research studies to test theoretical predictions from more mechanistic modeling approaches.

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UNDERSTANDING THE CAUSES AND CONSEQUENCES OF ANIMAL MOVEMENT: A CAUTIONARY NOTE ON FITTING AND INTERPRETING REGRESSION MODELS WITH TIME-DEPENDENT COVARIATES[1]

John R Fieberg, and Mark. Ditmer[2]

Summary

1. New technologies have made it possible to simultaneously, and remotely, collect time series of animal location data along with indicators of individuals' physiological condition. These data, along with animal movement models that incorporate individual physiological and behavioral states, promise to offer new insights into determinants of animal behavior. Care must be taken, however, when attempting to infer causal relationships from biotelemetry data. The possibility of unmeasured confounders, responsible for driving both physiological measurements and animal movement, must be considered. Further, response values (y_t) may be predictive of future covariate values (x_{t+s}; $s > 1$). When this occurs, the covariate process is said to be endogenous with respect to the response variable, which has implications for both choosing statistical estimation targets and also estimators of these quantities.

2. We explore models that attempt to relate x_t = log(daily movement rate) to y_t = log(average daily heart rate) using data collected from a black bear (*Ursus americanus*) population in Minnesota. The regression parameter for x_t was 0.19 and statistically different from 0 ($P <$ 0:001) when daily measurements were assumed to be independent, but residuals were highly autocorrelated. Assuming an autoregressive model (ar(1)) for the residuals, however, resulted in a negative slope estimate (-0.001) that was not statistically different from 0.

3. The sensitivity of regression parameters to the assumed error structure can be explained by exploring relationships between lagged and current values of *x* and *y* and between parameters in the independence and ar(1) models. We hypothesize that an unmeasured confounder may be responsible for the behavior of the regression parameters. In addition, measurement error associated with daily movement rates may also play a role.

4. Similar issues often arise in epidemiological, biostatistical, and econometrics applications; directed acyclical graphs, representing causal pathways, are central to understanding potential problems (and their solutions) associated with modeling time-dependent covariates. In addition, we suggest that incorporating lagged responses and lagged predictors as covariates may prove useful for diagnosing when and explaining why some conclusions are sensitive to model assumptions.

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RECENT POPULATION TRENDS OF MOUNTAIN GOATS IN THE OLYMPIC MOUNTAINS, WASHINGTON[1]

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ABSTRACT

Mountain goats (*Oreamnos americanus*) were introduced in Washington's Olympic Mountains during the 1920s. The population subsequently expanded and increased in numbers, leading to concerns by the 1970s over the potential effects of non-native mountain goats on high-elevation plant communities in Olympic National Park. The National Park Service (NPS) transplanted mountain goats from the Olympic Mountains to other ranges between 1981 and 1989 as a tool to manage overabundant populations, and began monitoring population trends of mountain goats in 1983. We estimated population abundance of mountain goats during 18-25 July 2011, the sixth survey of the time series, as a means to assess current population status and responses of the population to past management. We surveyed 39 sample units, comprising 39% of the 59,615-ha survey area. We estimated a population of 344±72 (90% confidence interval [CI]) mountain goats in the survey area. Retrospective analysis of the 2004 survey, accounting for differences in survey area boundaries and methods of estimating aerial detection biases, indicated that the population increased at an average annual rate of 4.9% since the last survey. That is the first population growth observed since the cessation of population control measures in 1990. We postulate that differences in population trends observed in western, eastern, and southern sections of the Olympic Mountains may reflect effects of climate variation across the pronounced precipitation gradient that exists.

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EVALUATION OF THE EFFECT OF GRAZING ON SELECTED GRASSLAND VARIABLES

David Rave, Kevin Kotts, and John Fieberg

SUMMARY OF FINDINGS

In 2011, we began a study in or near Working Land Initiative Areas to determine the effects of grazing on habitat metrics thought to be associated with the abundance of grassland bird communities. Specifically, we wanted to look at grazing impacts on woody vegetation, white lady slippers (*Cypripedium candidum*), and the diversity of plant species in upland and wetland areas. Preliminary sites were selected and initial data collection began in 2011.

INTRODUCTION

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). 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). Although burns can be effective, they are weather dependent, and require a large investment in personnel time, often making it difficult to attain habitat management goals. A variety of other options for creating disturbances in native grass stands are available to wildlife managers (e.g., grazing, haying, biomass harvest), but less is known about the response of native grass stands to these treatments. In particular, several managers have expressed interest in using grazing to help manage grasslands, but grazing has seen limited use due to questions surrounding its effectiveness as a management tool.

Historically, the major factors influencing grassland ecosystems were fire, grazing by herbivores, and climatic variations (Kirsch et al. 1978). Wildlife managers attempt to simulate these effects by manipulating vegetation. Grazing and mowing (Kirsch et al. 1978) and prescribed burning (Kirsch and Kruse 1972) are used to set back succession on managed areas. The suppression of these types of disturbances in prairie grasslands results in the invasion of woody species (Sauer 1950; Stewart 1956). Kirsch and Kruse (1972) found that species diversity of bird and vegetative species, as well as nest success, increased in managed versus non-managed grass fields in North Dakota.

Many species of upland nesting birds utilize residual vegetation as nest sites. Leopold (1933) noted that most waterfowl and gallinaceous birds depend upon residual vegetation for initial nesting attempts. Further, Bue et al. (1952) determined that ducks nesting in western South Dakota chose the tallest, most dense nesting cover available. Many studies have detected a positive relationship between upland nesting birds and grassland disturbance accomplished using grazing (Mundinger 1976, Brown 1978, Duebbert et al. 1986), prescribed burning (Kirsch and Kruse 1972, Tucker et al. 2004, Thatcher et al. 2006), and both treatments in combination (Fuhlendorf and Engle 2004, Trager et al. 2004,Powell 2006). Evaluating treatment response over time is also important as Powell (2006) found that the effects of prescribed burning and grazing on habitats of grassland nesting birds benefited different species depending upon number of years post treatment.

Appropriate response measures will be necessary to evaluate the effectiveness of grazing as a management tool. Sample and Mossman (1997) found that differences in habitat structure are likely more important to bird communities than differences in vegetative species composition. They suggest the following features of grassland habitat are important to grassland nesting birds: 1) vegetation height and density, 2) height and cover of woody vegetation, 3) litter depth and cover, 4) standing residual (dead) and live herbaceous cover, and 5) ratio of grass vs. forb cover.

This study will attempt to determine if grazing has positive (or negative) impacts on habitat metrics thought to be associated with the abundance of grassland bird communities. Specifically, we will conduct tests of the following null hypotheses:

1. Grazing does not affect the density of woody vegetation.
2. Grazing does not influence the abundance of small white ladies slippers (*Cypripedium candidum*).
3. Grazing does not influence the plant species diversity in upland areas.
4. Grazing does not influence plant species composition in temporary and seasonal wetlands.

STUDY AREA

The study will be conducted in Chippewa, Grant, Kandiyohi, Lac Qui Parle, Renville, Stevens, and Swift counties, within the prairie portion of Minnesota (Figure 1), and will be 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 will all be located on state managed WMAs or federally managed WPAs.

METHODS

Managed grasslands will be non-randomly selected from fields that will be grazed in 2011 or 2012. Paired subplots of equal size (30 ft^2) will be delineated at each site, and one subplot of each pair will be randomly assigned to the “grazing” treatment; the other, the control, will be enclosed using cattle exclusion fencing, and thus protected from grazing. We will attempt to minimize differences between paired subplots when choosing areas for inclusion in the study. Paired subplots will be erected in each of FY12 and FY13, then subplots will be monitored through the end of the growing season 2014 to determine grazing effects.

For each response variable; stem density of woody plant species, blossom densities of small white lady’s slippers (*Cypripedium candidum*), species composition in upland areas, and species composition in seasonal/temporary wetlands; we will select paired subplots, for a total of 12 - 16 paired subplots.

Visual obstruction measurements (VOMs, Robel et al. 1970) will be taken every month from May through August in grazed and control subplots of each field following methods described by Zicus et al. (2006). Three VOM readings will be taken at the 3 quarter points along the longest straight-line transect across each subplot. At each field sampling point, VOMs will be measured in each cardinal direction. This will provide 12 VOMs for each subplot on a given date.

A Daubenmire square (Daubenmire 1959) will be used to determine coverage by various species across grazed and control subplots. We will sample at 3 locations along the transect in all subplots of each field every month from May - August. The 1m^2 Daubenmire frame will be placed on the ground in 3 locations in each subplot. Each plant species (and % coverage) within the frame will be recorded.

RESULTS

We selected 3 Wildlife Management Areas (WMA) to work on in 2011. The Bjornson WMA was selected because it was scheduled to be grazed and there is a high density of white ladies slippers on the area. We expected high annual variability in the blossom density of lady’s slippers, thus, in June 2011 we erected 3 paired subplots in a single pasture to examine the grazing response of small white ladies slippers. Prior to cattle grazing the unit we counted the blossoms within each of the grazed and control subplots. In August 2011, we erected 3 paired subplots in 3 pastures on the Simon Lake WMA, and 2 paired subplots on the Helcene WMA. Because of the July, 2011 Minnesota state government shutdown, we were unable to erect all the subplots we had hoped in summer 2011. In FY 2012 we plan to erect another 12 paired subplots in several pastures to examine the grazing response of woody vegetation, and begin to examine species composition changes in uplands and seasonal/temporary wetlands.

DISCUSSION

Management of native grass stands has become an important component of wildlife management in prairie portions of Minnesota (K. Kotts, personal communication). Historically, spring prescribed burning has been the preferred management option used to create disturbances in these fields. However, the amount of habitat manipulated by spring burns is often dictated by spring weather conditions. Knowledge of the response of native grasses to management treatments other than spring burning may allow managers to treat additional acres, or manage grasslands in a more efficient manner.

Further, determining alternate management scenarios for grasslands, particularly those that may have a financial incentive for the landowner (e.g. biofuel harvest, haying, or grazing), may entice some landowners to maintain their land in a grassland program such as CRP,WRP, or managed pasture rather than convert the land into cropland. This would have landscape wide benefits for wildlife, erosion control, and clean water.

Standardizing the many variables associated with grazing (e.g. stocking rates, grazing period), among grazing treatment fields will be challenging. Variation in the implementation of the grazing treatment will add variability to the response measures, and will likely reduce statistical power to detect treatment effects.

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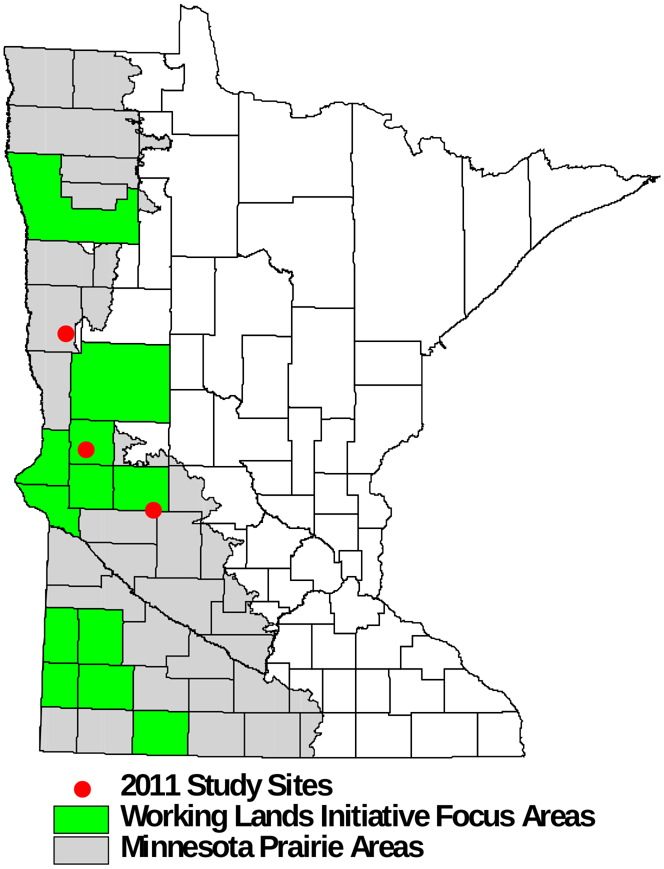

Figure 1. Minnesota counties showing 2011 study sites, Minnesota prairie areas and Working Lands Initiative focus areas, 2011.

NESTING ECOLOGY OF RING-NECKED DUCKS IN THE BOREAL FOREST OF NORTHERN MINNESOTA

Charlotte Roy and Christine Herwig

SUMMARY OF FINDINGS

We have completed 4 years of fieldwork on this research project. Thus far, we have searched 118 wetlands, located 88 ring-necked duck (*Aythya collaris*) nests, marked 50 hens, and followed 22 broods. Nest success during this study has ranged from 0.12-0.46, with high success in a wet year, lower success in dry years, and the lowest success in a year with extreme flooding of nesting habitat. Hen survival during the breeding season has not been previously estimated in Minnesota, and ranged 0.54-0.88, again with high survival in a wet year and low survival in a dry year. Brood survival also has not been estimated previously in Minnesota and was 0.23 for 2008-2010 combined. Additional data collection will enable more robust estimates of these parameters.

INTRODUCTION

The ring-necked duck is a characteristic and important species for the Laurentian Mixed Forest province of Minnesota (Minnesota Department of Natural Resources [MNDNR] 2006), also known as the Boreal or Coniferous Forest biome. Recent surveys 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, limiting informed interpretation of these local survey data and our understanding of how vital rates affect population growth of ring-necked ducks in the forest. These data are particularly pertinent given the increasing development and recreational use in the forest (MNDNR 2006) and predictions that the spruce-fir forest will shift north of Minnesota as a result of global climate change (Iverson and Prasad 2001).

Nest success, hen survival, and brood survival in the boreal forest are largely unknown. Some data are available for nest success and brood survival in north-central Minnesota (Hohman and Eberhardt 1998) and Maine (McAuley and Longcore 1988, 1989), but data for the boreal forest of the upper Midwest dates back to the early 1970s (Sarvis 1972). Limited data are available for nest success outside the forest; Maxson and Riggs (1996) studied nest success of ring-necked ducks in the forest-prairie transition during 1985–1987, and Koons and Rotella (2003) compared nest success of ring-necked ducks to that of lesser scaup (*Aythya affinis*) in the parkland of Manitoba. However, neither study examined hen or brood survival during the breeding season. In general, nesting and brood-rearing information for diving ducks are limited in comparison to the data available for dabbling ducks (Yerkes 2000).

Gathering information on vital rates during the breeding season is an important first step to understanding recent population patterns of ring-necked ducks in Minnesota. Although sensitivity analyses of vital rates on population growth rates are not available for ring-necked ducks, sensitivity analyses for mid-continent mallards indicated that nest success explained the most variation (43%) in population growth rates (Hoekman et al. 2002). A similar analysis for the Great Lakes Region indicated that duckling survival (32%) and nest success (16%) accounted for the greatest variation in mallard population growth rates during the breeding season (Coluccy et al. 2008).

OBJECTIVES

1. To obtain baseline information on ring-necked duck nest success, hen survival, and brood survival before fledging in the forest.

2. To examine how these vital rates vary along a gradient of development and recreational use (e.g., number of dwellings, boat access, proximity to roads).

METHODS

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 during 2004–2010 and 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 (US Geological Survey Gap Analysis Program [GAP] types 12 and 13 surrounded by GAP types 10, 14, and 15). We identified 103 lakes within a 40-km (25-mile) radius of Bemidji with land cover attributes similar to those used in the 2 surveys. In 2009, we scouted wetlands in early spring and focused nest- searching efforts on the wetlands where ring-necked ducks had been seen. In 2010, we used scouting data from 2007 to 2009 to identify lakes where ring-necked ducks had been observed. We excluded lakes considered unsafe to search or where we had been denied access. This process resulted in 95 basins as potential targets for nest-searching in 2010 and 2011. In 2011, 8 more wetlands were added due to preseason scouting and the location of a radio-marked hen on 1 wetland.

We searched for ring-necked duck nests in the springs and summers of 2008–2011. To locate 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. We determined water depth and distance to open water at each nest after it hatched or failed.

Late in incubation, we trapped hens on nests with Weller traps (Weller 1957) to attach radio-transmitters. Because we were initially concerned that a surgical transmitter attachment method might be too disruptive to incubating hens, we tried a bib-type transmitter attachment method, which had been used with previous success in wood ducks (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 with 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. Due to concerns about low hen survival in 2009 and low brood survival during 2008 and 2009, we changed the transmitter attachment method in 2010. We tried the surgical transmitter attachment method that we had been using for the MNDNR-funded study on post-fledging ring-necked ducks (Korschgen et al. 1996). However, we used a local anesthetic (i.e., lidocaine) instead of isoflurane so that we could do surgeries in the field (Corcoran et al. 2007). We also used propofol, injected intravenously, to reduce nest abandonment (Rotella and Ratti 1990, Machin and Caulkett 2000) on 6 hens in 2010 and on all hens in 2011. When propofol was used, hens were placed on nests rather than being released from the edge of the wetland.

Nests were monitored every 4–7 days to determine fate (abandoned, depredated, or successful) and Mayfield nest success (Mendall 1958, Mayfield 1975). After nests hatched, we attempted to monitor broods every 3–7 days. At each observation, we counted the ducklings present, and when possible, aged them from a distance based on plumage characteristics (Gollop and Marshall 1954). Broods were monitored until ducklings reached age Class III (39-49 days old) or until total brood loss occurred. We considered hens to have lost their entire brood when hens were observed without any ducklings for 3 consecutive observations or if the hen was found >16 km (10 miles) from the nesting lake. We continued to monitor hens after the brood-rearing period for as long as they could be tracked before migration to examine their survival using the Kaplan-Meier method (Kaplan and Meier 1958).

In 2011, the state government shutdown occurred 1 July to 20 July, during peak weeks of ring-necked duck hatching. We were still finding nests at the time of the shutdown (2 nests were located 2 days before the shutdown) and 5 nests were still active when the shutdown began. We attempted to check nests that had been active and locate broods when state government activities resumed. However, the shutdown precluded data collection according to the methods described above.

RESULTS

Thus far, we have searched 118 wetlands (Figure 1), located 88 active nests, marked 50 hens, and followed 22 broods. We searched for nests on 37 wetlands for a total of 73 searches (17 wetlands searched once and 20 wetlands searched >1 time) between 22 May and 22 July 2008, 37 wetlands searched 54 times (21 wetlands once and 16 wetlands searched >1 time) between 29 May and 22 July 2009, 73 wetlands searched 128 times (35 wetlands once and 38 wetlands searched >1 time) between 19 May and 12 July 2010, and 76 wetlands were searched 107 times (54 wetlands once and 22 wetlands searched >1 time) between 23 May and 30 June 2011.

Nest Survival

We located 18 (14 active, 4 depredated when found) ring-necked duck nests on 10 wetlands in 2008, 20 active nests on 11 wetlands in 2009, 32 active nests on 17 wetlands in 2010, and 22 active nests on 16 wetlands in 2011. 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 for nests that were incubated was 9.1 ± 0.6 (mean ± SE, range = 7–15, *n* = 12 nests with 109 eggs) and 86.6 ± 0.1% of eggs hatched in nests that hatched. In 2009, 7 nests hatched, 9 were depredated, and 4 were abandoned, with at least 2 cases of abandonment likely due to trapping. The average clutch size for incubated nests was 8.3 ± 0.3 (range = 7–11, *n* = 19 nests with 158 eggs) and 89.5 ± 0.6% of the eggs hatched in nests that were successful. In 2010, 13 nests hatched, 9 were depredated, 6 were abandoned after trapping and transmitter attachment, 2 were abandoned for other reasons, 1 had an unknown fate because we could not determine the outcome based on evidence at the nest site, and 1 failed because the hen died during transmitter-implantation surgery. We began using propofol on all hens captured later in the field season because 5 of 13 hens marked without propofol had abandoned their nests. Average clutch size for incubated nests was 8.3 ± 0.3 (range = 5–10, *n* = 30 nests with 250 eggs) and 84.5 ± 0.1% of eggs hatched. In 2011, 6 nests hatched, 3 were abandoned (2 to investigator disturbance and 1 for unknown reasons), and 13 were depredated. Average clutch size was 8.8 ± 0.4 (range = 4-11, n = 19 nests with 166 eggs) and hatching success was 85.0 ± 0.2%. Mayfield nest success for a 35-day period of laying and incubation was 30% in 2008, 27% in 2009, 46% in 2010, and 12% in 2011.

Hen Survival

We put transmitters on 8 hens in 2008, 14 hens in 2009, 19 hens in 2010, and 9 hens in 2011. 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 observed 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 3 weeks post-marking and the other 4 weeks 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 (17, 20, 32, 33, 55, and 84 days post-marking). Evidence obtained at the recovery sites 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 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. In 2010, only 1 hen died during the monitoring period. She died 17 days after marking and appeared to have been killed by a mammalian predator. She did not have a brood. Twelve of 19 transmitters dehisced 55.1 ± 6.0 days (range = 30–121 days) after attachment. In 2011, 2 hens were depredated, one by a mink and the other by an unknown predator. One hen did not recover from anesthesia and her nest was censored. One hen dehisced her transmitter in mid-August, 53 days after marking. Hen survival through mid-September was 0.80 ± 0.18 for 2008, 0.54 ± 0.08 for 2009, 0.88 ± 0.11 for 2010, 0.69 ± 0.19 for 2011 (through 19 August because too few hens were marked to justify tracking through September).

Brood and Duckling Data

In 2008, 7 radiomarked hens had broods (*n* = 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) age classes (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. We also monitored the brood of 1 unmarked hen that was not trapped in time to give her a transmitter. Her brood made it to the IC stage, but they were not observed again and 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.

We observed 6 broods in 2010 (*n* = 40 ducklings); 3 broods survived to age Class III and likely fledged 14 ducklings, 1 brood was located as Class IA ducklings, but the hen was not located again, 1 brood survived until age Class 1A, and another brood survived to age Class IB. Seven marked hens were believed to have hatched ducklings, but were not located with broods before total brood loss.

In 2011, following the government shutdown, we were able to locate 5 hens and follow 2 broods that were still alive. Both broods fledged; one brood of 3 ducklings made it to flight (50 days) and the other had 6 ducklings survive until at least class III (42 days, and most likely flight). Thus, excluding 2011, brood survival to fledging was 0.23 ± 0.09.

Brood movements also were observed. In 2009, for example, a hen moved her 3 (IC) young from the nesting wetland to another wetland (~1,205 m) from which they fledged. In another instance, a hen and her brood of 6 (IB) were seen walking to another wetland ~365 m from their nesting wetland. In 2010, 4 hens moved their broods to nearby wetlands at various ages (i.e., IA, IC, IIA, IIB). One of these hens later returned with her brood to the wetland where they hatched and her brood later fledged.

DISCUSSION

Our success finding nests has been comparable to that in other studies that found ring-necked duck nests (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 are comparable to his estimates of 44% based on 188 nests. The causes of nest failure in our study (17% flooding, 70% depredation, and 13% abandonment) were similar to those of other studies (16–24% flooding, 67–80% depredation, and 5% abandonment; Mendall 1958, McAuley and Longcore 1989), if we exclude nests where abandonment was attributed to investigator disturbance. Early estimates of hatching success appeared to be 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, and early summer 2010 was very rainy as well, which may have chilled eggs and flooded nests. The spring of 2011 was unusually wet and we noticed unusually high depredation rates of nests, which we hypothesize may have been due to contraction of available nesting habitat due to extremely high water levels.

Our hen survival rates for the period June–mid-September were comparable to reports for hen mallards during April–September (0.80, Cowardin et al. 1985; 0.60, Blohm et al. 1987; 0.67, Brasher et al. 2006). This was unexpected because we marked hens late in incubation, and incubation is a period when hen mortality is expected to be greatest. Brood survival rates also seemed 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% (*n* = 381). Further investigation is necessary to get better estimates of hen survival and brood survival in ring-necked ducks in Minnesota.

This study is planned to continue through 2012. Results should be viewed as preliminary and are subject to change with further data collection.

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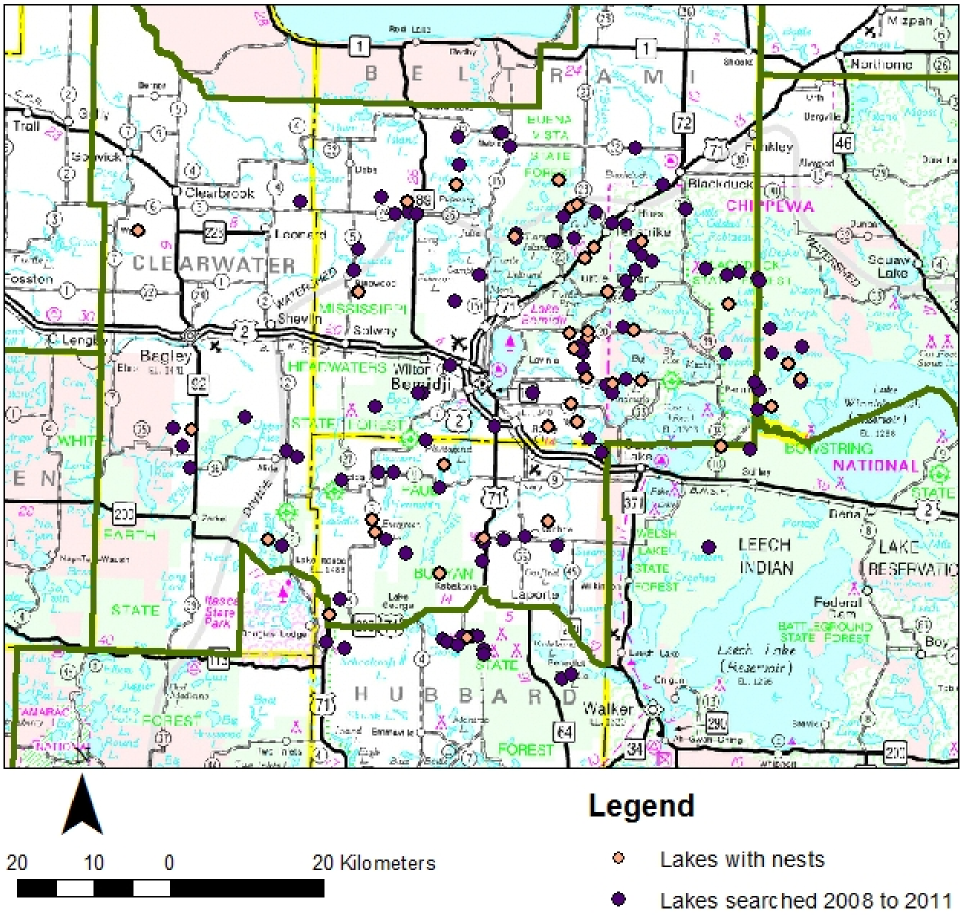

Figure 1. Wetlands searched for ring-necked duck nests in north-central Minnesota during 2008–2011.

INVESTIGATION OF TREMATODES AND FAUCET SNAILS RESPONSIBLE FOR LESSER SCAUP AND AMERICAN COOT DIE-OFFS

Charlotte Roy and Christine Herwig

SUMMARY OF FINDINGS

Two waterfowl die-offs occurred on Lake Winnibigoshish in 2011. During spring 2011, several hundred Lesser lesser Scaup scaup (*Aythya affinis*) were observed moribund, unable to fly and/or keep their heads up, and necropsy later confirmed trematodiasis. Later in the fall of 2011, 5-10% of a raft of 800-1,000 Lesser Scaup appeared sick near Third River Flowage. Trematodiasis was again confirmed through necropsy of dead individuals. Additionally, a few sick birds were observed at Bowstring and Round lakes, but whether they were infected with trematodes and the origin of the disease was unknown.

The invasive faucet snail is the only known first host of the trematodes, so we sampled faucet snails where they are known to occur in Minnesota; Lake Winnibigoshish, Upper and Lower Twin lakes, the Shell River, First Crow Wing Lake, and Crow Wing River. We detected the trematodes *Cyathocotyle bushiensis* and *Sphaeridiotrema globules* in all sites, with the exception of Upper Twin Lake where we only found *S. globules*. We also sampled Bowstring and Round lakes for faucet snails, because these lakes are known to be important to migrating Lesser Scaup. We report the first detection of faucet snails in Round Lake, although we have not yet detected the trematodes there. We still have not detected faucet snails or the trematodes at Bowstring Lake.

INTRODUCTION

During the autumn of 2007, 7,000 Lesser Scaup (*Aythya affinis*) and hundreds of American american Coots coots (*Fulica americana*) died on Lake Winnibigoshish in north-central Minnesota (Lawrence et al. 2008). In 2008, 2,000 more birds died (Lawrence et al. 2009). These deaths were attributed to trematodiasis caused by non-native intestinal trematodes (*Cyathocotyle bushiensis*, *Sphaeridiotrema globules,* and *Leyogonimus polyoon).*

The trematode species responsible for the die-offs have a complex life cycle that involves two intermediate hosts. The faucet snail (*Bithynia tentaculata*), a non-native species from Europe (Sauer et al. 2007), is the only known first intermediate host of these trematodes in the Midwest and also serves as the second host for *C. bushiensis* and *S. globules.* The second host of *L. polyoon* is one of a variety of larval aquatic insects, including damselflies (Zygoptera) and dragonflies (Odonata) (National Wildlife Health Center, unpubl. data). Adult trematodes develop in waterfowl after they consume infected snails and in American Coots (*Fulica americana*) and Common common Moorhens moorhens (*Gallinula chloropus*) after consumption of infected insects. Parasite eggs are then defecated by sick birds and later ingested by snails, continuing the cycle. Because of this complex life cycle, the dynamics of faucet snail distribution and transmission of these parasites to Lesser Scaup and other birds are poorly understood.

The faucet snail was first detected in Lake Winnibigoshish in the spring of 2008, following the waterfowl die-off the previous fall. The faucet snail was detected in Upper and Lower Twin lakes and the Shell River in Minnesota in 2009. In 2010, the Crow Wing River was designated as infested with faucet snails, and in 2011, First Crow Wing Lake and Second Crow Wing Lake were added to the list of waters infested with faucet snails. These new infestation sites may afford us additional opportunities to learn about this disease cycle. We will examine the factors associated with faucet snail abundance and distribution, parasite prevalence within snails, and the influence of snail densities and site attributes (e.g., water depth, distance from shore, substrate composition) on Lesser Scaup foraging.

OBJECTIVES

1. Improve understanding of Lesser Scaup foraging as it relates to faucet snail and other food source distribution and density, including water depth, distance from shore, and substrate composition
2. Examine factors (e.g., temperature, substrate, vegetation, other snail species) that are associated with the distribution and movement of faucet snails
3. Examine the factors that influence the prevalence of the parasites in faucet snails (e.g., snail density, temperature, microhabitat, time of year)
4. Examine how faucet snail distribution varies during spring, summer, and fall

METHODS

During 2011, we sampled faucet snails during spring, summer, and fall at the same points within a lake or river (Table 1a,b). In small lakes (<1000 acres404 hectares), we used transects that traversed the entire length of the lake and across a range of depths. In large lakes, we used index areas with points stratified by depth for sampling. In Lake Winnibigoshish, we had 2 index areas, the West Winni Index Area and the East Winni Index Area, which were 5-6 K km along the longest dimension and approximately 2 K km in width (Fig. 1). In rivers, we sampled points at regular intervals (500 m) along the infested corridor for a maximum length of 10 Kkm.

We used 2 sampling methods; we used a bottomless sampling cylinder (0.2 m^2) at 30 and 60 cm depths for comparisons with an ongoing study on the Upper Mississippi River, and we also sampled with a benthic sled to standardize our protocol for all depths. We dragged the sled a distance of 1.2 m at deeper depths to examine how snail distribution varied within a water body. We collected data on microhabitat variables at each point to examine relationships to snail distribution, the snail community, and parasite prevalence. These included substrate (e.g., silt, rock, sand, vegetated, muck, etc.), temperature (C^o), water depth (cm), and a secchi depth (cm) reading was taken 8 times (4 times on the way down and 4 times on the way up) from the shaded side of the boat and averaged. At each snail collection site, we determined pH, dissolved oxygen (mg/L), conductivity (µS/cm), and salinity ($^o/_{oo}$) with a Hach Company (Loveland, Colorado) HQd portable meter that was calibrated daily for pH and weekly for conductivity. Flow (mps) was measured at 60% of the total depth (from the surface) with a Global Water Instrumentation (Gold River, California) flow probe when flow was detectable and averaged over a 40 s interval (the USGS "6 tens method").

Invertebrate samples were stored in the refrigerator until processed. We used a magnifying lens and microscope as needed to identify all invertebrates to Order and noted their presence in each sample. We identified all snails to genus and counted their numbers in each sample. We determined the size of *B. tentaculata* and similarly sized *Amnicola* spp. with calipers, as measured along the central axis from the apex. Parasite prevalence was determined for all samples possessing at least 50 *B. tentaculata* (R. Cole, NWHC, unpubl. data). For samples possessing 10-49 *B. tentaculata,* we collected additional snails while in the field from the same location at the same time to increase the number of samples for which we could do prevalence. These additional snails were not used in the determination of snail abundance at the site. Trematode stages (cercariae or metacercariae), species (*C. bushiensis, S. globulus, L. polyoon*), and numbers were also recorded in the lab.

Each season, we collected a water sample at each sample lake or river and sent it to the Minnesota Department of Agriculture for analysis. Total phosphorus (ppm), nitrite plus nitrate nitrogen (ppm), chlorophyll a (ppb), total alkalinity (ppm), ammonia nitrogen (ppm), and calcium (ppm) were quantified but have not yet been interpreted.

We also identified sites where Lesser lesser Scaup scaup foraged and collected benthic samples at these locations. These sites were identified through observations of birds from shore or from a boat. We determined the location of rafts of scaup using a compass from 2-3 observation points, which was plotted in ArcMap version 10 (Environmental Systems Research Institute, Inc., Redlands, California) to determine the area occupied by the birds. We then sampled at 100 m intervals along a transect through this area. Food densities, water depths, distance from shore, lake size, and substrate composition at these foraging locations were recorded using the same techniques as snail sampling.

We also collected scaup carcasses during die-offs at study lakes (Fig. 1) for confirmation of trematodiasis by the NWHC in Madison, Wisconsin. Additionally, Bowstring and Round lakes are known for having large number of scaup, particularly in the fall, and have been the sites of trematodiasis die-offs in the past. We monitored Bowstring and Round lakes for scaup die-offs during the spring and fall and collected benthic samples during the summer of 2011 to search for faucet snails which have not previously been detected at these lakes.

RESULTS

Faucet snails

We detected faucet snails at both index areas on Lake Winnibigoshish, Upper and Lower Twin lakes, the Shell River, First Crow Wing Lake, and Crow Wing River (Table 1). We also report the first detection of faucet snails in Round Lake (Fig. 1).

We did not find faucet snails on Bowstring Lake, but we did detect metacercarial stages of the trematodes in banded mystery snails (*Vivaparus georgianus*) collected from Bowstring, suggesting that either the faucet snail is present but not yet detected here, or that another snail species is serving as the first intermediate host. Samples of *V. georgianus* were sent to University of Wisconsin-LaCrosse for experimental challenges with miracidial stages of trematodes to determine whether it can serve as the primary host of these trematodes.

Preliminary analysis indicates that faucet snails are more abundant during the summer than during fall or spring. Faucet snails also appear to move into shallower depths in the summer, perhaps to reproduce on vegetation, with a return to deeper depths in fall and spring. Additional data collection will help determine whether this is a robust seasonal pattern. At the Twin lakes, populations of faucet snails may be expanding and increasing; further data collection will be necessary to determine whether separation of population growth and expansion from seasonal patterns is possible. More formal analyses will be included in subsequent reports.

Trematodes

Both *C. bushiensis* and *S. globulus* were detected on Lake Winnibigoshish, Lower Twin Lake, the Shell River, Bowstring Lake, and the Crow Wing River. Only *S. globulus* was detected in samples from Upper Twin Lake. Faucet snails collected from Round Lake were not infected with trematodes, but we only found snails at one location which likely limited our ability to detect infections. Prevalence (proportion of snails infected) of *C. bushiensis* was generally higher than that of *S. globulus* within a water body. Among water bodies, the West Winni Index Area and Shell River had very high prevalence relative to that of the Twin lakes and East Winni Index Area. Prevalence was generally highest in the spring and fall, and lowest in the summer, but some exceptions did occur. Intensity (number of parasites in infected snails) of parasites also appeared highest in the spring and fall, but the highest intensity observed was of *S.*

globulus in the summer at West Winni Index Area. Intensities of both *C. bushiensis* and *S. globulus* were highest at West Winni Index Area and were consistently high enough to result in mortality of waterfowl within 8-10 days after consumption of 4-20 snails (Mucha and Huffman 1991). Not coincidentally, this is where the waterfowl die-offs have been occurring on Lake Winnibigoshish (Fig. 1).

Scaup

We also collected benthic samples from below rafts of foraging scaup (Table 1). Foraging could only be confirmed in the spring, as we could not get close enough to the raft during hunting season to evaluate whether they were foraging. We observed large rafts of scaup during our visits to Bowstring, Round, and Winnibigoshish lakes (Table 2). Staff from the Minnesota Department of Natural Resources-Grand Rapids office also made regular visits to Winnibigoshish, Round, and Bowstring lakes throughout the season to check for sick birds (Table 2).

Two die-offs occurred on Lake Winnibigoshish in 2011; during spring 2011 a raft of 2,000 birds were observed on the west side and several hundred sick birds were documented (Table 2). Nine birds (2 Greater Scaup, 7 Lesser Scaup) were collected and sent to the NWHC. Trematodiasis was confirmed. During fall 2011, a raft of 800-1,000 Lesser Scaup was observed near Third River Flowage at Lake Winnibigoshish, with 5-10% of the raft exhibiting symptoms of trematodiasis. Five Lesser Scaup were collected from this die-off and sent to the NWHC for confirmation of trematodiasis and it was confirmed. We also collected benthic samples from below the raft of sick scaup. A few sick birds were also observed at Bowstring and Round lakes (Table 2).

DISCUSSION

This report summarizes activities for the first full year of field work (spring, summer and fall 2011). Fall 2010, a pilot season, was included in an earlier report. Data entry and analysis is preliminary and still underway. More formal analyses will be included in subsequent reports. We plan to continue using the same methodology, adding additional lakes with faucet snails as they become known, through fall 2012.

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Table 1a. Sampling sites for faucet snails in northern Minnesota water bodies during spring, summer, and fall 2011. Number of points refers to the number of points sampled each season of sampling.

Location	**No. seasons sampled**	**No. sample points**	**Faucet snails detected**
East Winni Index Area	3	80	Yes
West Winni Index Area	3	80	Yes
Bowstring Lake	1 (Summer)	61	No
Round Lake	1 (Summer)	42	Yes
Upper Twin Lake	3	24	Yes
Lower Twin Lake	3	39	Yes
First Crow Wing Lake /River	1 (Fall)	18	Yes
Shell River	3	22	Yes
Total		874	

Table 1b. Sampling sites for faucet snails associated with Lesser Scaup in northern Minnesota water bodies during spring and fall 2011. Number of points refers to the number of points sampled each season of sampling.

Location	**Season sampled**	**No. sample points**	**Faucet snails detected**
Lake Winnibigoshish	Spring/Fall	2/3	Yes
Sugar Lake	Fall	2	No
Bowstring Lake	Fall	8	No
Round Lake	Fall	6	Yes
Total		21	

Table 2. Reports of scaup observed by Minnesota Department of Natural Resources staff on lakes in northern Minnesota during spring and fall 2011. Scaup that failed to escape approach or had drooping heads were considered to be sick; dead birds were typically found along the shoreline.

Location	**Date**	**Total no. of scaup observed**	**No. of sick or dead scaup observed**
SPRING			
Winnibigoshish	4/19/11	1,800	None
Winnibigoshish	4/20/11	1,600[a]	200 sick
Winnibigoshish	4/22/11	1,500	None
Winnibigoshish	4/25/11	2,000	None
Bowstring	5/3/11	1,200-2,500	None
Round	5/3/11	200-400	None
Winnibigoshish	5/5/11	7,200	6-12 sick, 1 collected[b]
Winnibigoshish	5/6/11	9,000	300 sick, 6 collected
Winnibigoshish	5/11/11	900-1,000	~20% suspected sick
Winnibigoshish	5/19/11	Not reported	3 sick, 10 dead
FALL			
Bowstring	10/4/11	10	None
Round	10/4/11	0	None
Winnibigoshish	10/4/11	0	None
Round	10/13/11	100-1,000	None
Bowstring	10/19/11	5,000	None sick, 1 dead
Winnibigoshish	10/20/11	2,000	2 dead near Third River Flowage
Round	10/24/11	5,000-7,000	5 suspected sick
Winnibigoshish	10/24/11	200-300	Not assessed
Bowstring	10/25/11	2,000-3,000	1 suspected sick, 1 dead
Winnibigoshish	10/26/11	800-1,000	10-20 suspected sick, 40-60 dead, 5-10% mortality of raft
Bowstring	10/27/11	300-400[a]	25 sick
Winnibigoshish	10/31/11	Not reported	42 dead near Third River Flowage

Round	11/1/11	10,000	1 sick, 2 dead
Bowstring	11/1/11	20,000	None
Winnibigoshish	11/2/11	Not reported	3 sick, 20 dead
Round	11/8/11	5,000-10,000[a]	None
Bowstring	11/8/11	20,000-30,000[a]	225 sick
Winnibigoshish	11/10/11	200-300	2 suspected sick, 1 dead
Winnibigoshish	11/12/11	400-500 (Sugar)[a]	6 sick near Third River Flowage
Round	11/14/11	3,000-5,000	None
Bowstring	11/14/11	10,000-15,000	None
Winnibigoshish	11/18/11	Few scaup- freezing up	None

[a] Benthic samples collected below scaup.

[b] Scaup were collected and sent to the NWHC to be tested for trematodiasis.

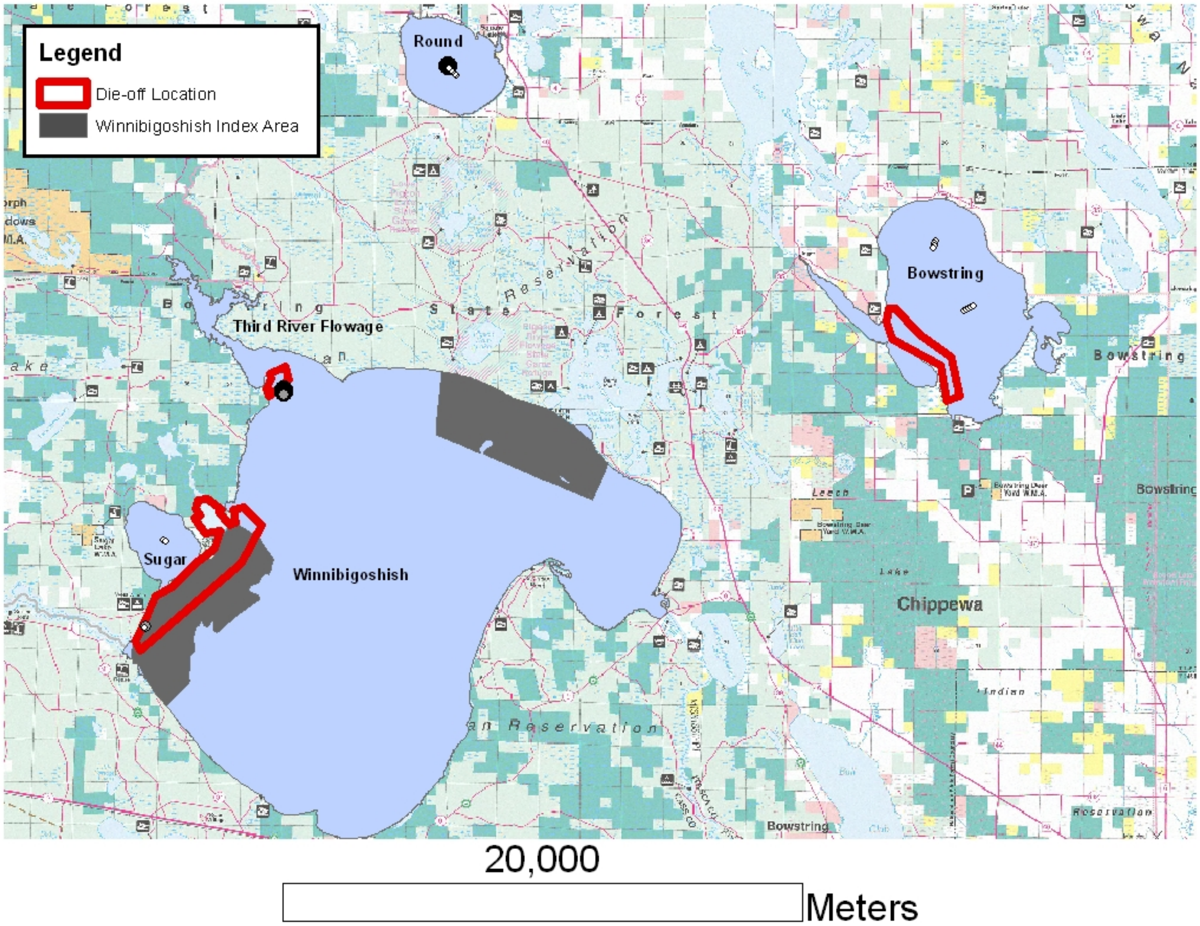

Figure 1. Relative locations of the index areas on Winnibigoshish and the other lakes where birds with trematodiasis were observed. Locations of waterfowl die-offs in Minnesota during spring and fall 2011 are also depicted, along with points sampled under Lesser Scaup rafts on each water body.

CHARACTERISTICS OF SHALLOW LAKES IN MINNESOTA: REGIONAL COMPARISONS AND RELATIONSHIPS AMONG LANDSCAPE SETTING, AMBIENT NUTRIENTS, AND FISH COMMUNITIES

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SUMMARY OF FINDINGS

Minnesota's shallow lakes provide numerous direct human benefits such as clean water, hydrologic storage to limit flooding, recreational opportunities, and access to unique wild areas. They also contribute valuable ecosystem services including carbon sequestration and habitat for native species. Unfortunately, water and habitat quality of Minnesota's shallow lakes have deteriorated dramatically during the past century. We are studying factors influencing ecological features and causes for deterioration of these sites, comparing costs of possible rehabilitation strategies, and synthesizing results to provide guidance for future shallow lake management. We evaluated approximately 130 shallow lakes in 5 ecological regions of Minnesota. Efforts include extensive sampling of shallow lakes to identify direct and indirect causes of deterioration, evaluation of responses of 12 lakes currently undergoing rehabilitation, and an economic analysis to identify enhancement strategies most likely to produce the greatest improvements in water quality per unit cost. Ultimately, our results will provide guidance to municipalities, state, county, and local governments, and private organizations in identifying cost-effective approaches for maintaining and restoring ecological integrity of shallow lakes throughout Minnesota.

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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. Collectively, these shallow lakes represent an international resource, providing critical waterfowl habitat and ecological benefits within Minnesota and the Mississippi Flyway. Currently, only about 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 quality, seasonal duck abundance and habitat use, and hunter satisfaction, MN DNR 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. Rehabilitation 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 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 fish removals 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. 2009).

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 sometimes 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. 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 (Tonn and Magnuson 1982; Rahel 1984). 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 shallow lake characteristics.

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. 2010). 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. 2012). 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. 2009). Results from our earlier work indicate that fish abundance and community structure exert major influences on shallow lake invertebrates, yet this relationship varies widely across ecological regions. We are also comparing relative influences of fish communities, lake isolation, and watershed size and composition on shallow lake invertebrate communities. Contributions from Sean Vaughn (Division of Waters, MDNR) and Robert Wright (Section of Wildlife, MDNR) 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.

We plan 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 modify ecological characteristics of shallow lakes. An overarching finding of the prior work was that regional differences often 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 perceptions of staff from the MDNR shallow lakes program 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, as well as 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 to turbid-water states in prairie lakes. We need to understand 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.

Working Hypotheses

We hypothesize that fundamental drivers are ultimately responsible for most of the variation in ecosystem characteristics of Minnesota's shallow lakes: climate, ambient nutrient levels, fish abundance and community type, landscape features, land use, and morphometric features of individual lakes. These factors, in turn, are expected to induce strong, predictable spatial gradients in shallow lake characteristics across Minnesota. Thus, shallow lakes should 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 influence of these drivers, as well as synergistic combinations of 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 fundamental drivers such that within-lake interactions will also contribute to observed spatial and temporal patterns.

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. 2009). 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 for cost-effective approaches in different regions of the state.

APPROACH AND METHODS

Our research has 3 general objectives; approaches and specific methods for each objective are summarized below.

Objective 1: Extensive Lakes - Identify and estimate major factors responsible for deterioration of shallow lakes in 6 areas of Minnesota (hereafter Extensive Lakes).

We selected study lakes in 6 areas among 5 Ecoregions of Minnesota. The following numbers of lakes were sampled during 2010 and 2011 (Figure 1): Twin Cities 22, Windom 22, Alexandria 23, Itasca 22, Chippewa 15, and Red Lake 23. We sampled a total of 127 lakes for this extensive aspect of our study. We had planned to sample approximately 17 more lakes during 2010, but this was not practical due to low-water conditions, unexpected characteristics of lakes (such as alteration due to damming by beavers, extreme depth, or other features not noted until field visits), and because in at least one case land ownership changed before onset of our study.

Lakes were sampled to assess general ecological features and to determine whether basins exhibit characteristics of clear- or turbid-water regimes. Lake watershed characteristics associated with each study lake were also determined. 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.

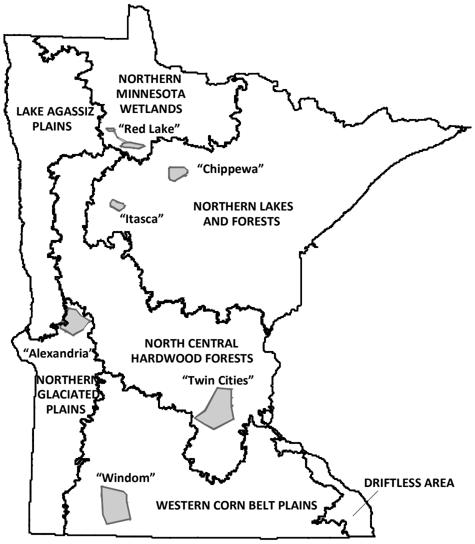

Figure 1. Map showing locations of study areas (shaded gray) in relationship to Minnesota's aquatic ecoregions (thick black lines).

All sites were visited during July-early August 2010 and 2011. At each study lake, we sampled fish populations, abundance of submerged aquatic plants, aquatic invertebrates, water transparency, and a suite of chemical constituents in lake waters (Table 1). Water samples were collected in the field and are being tested for turbidity and concentrations of dissolved inorganic and total nitrogen, dissolved and total phosphorus, dissolved inorganic and organic carbon, chlorophyll *a* (chla; as a proxy for phytoplankton biomass). Additional laboratory analyses are being conducted on water column particulate matter (seston) to determine concentrations of carbon, nitrogen, and phosphorus suspended in lake water columns. Field crews collected approximately >2,500 samples of aquatic invertebrates from study lakes. Samples are currently being processed and we expect that resulting electronic data sets will be developed by summer 2013.

Table 1. Summary of lake variables sampled during summer 2010 and 2011. Similar data were gathered from Extensive (N=127) and Intensive (N=8) lakes except that Intensive lakes were sampled once monthly during June, July, and August. Extensive lakes were sampled a single time during July (2010) or August (2011).

Biological	Physical	Chemical
Fish abundance (gill and trap nets)	Turbidity	Total Nitrogen
Submerged aquatic plants (rake and mass) methods	Specific Conductivity	Dissolved Inorganic Nitrogen
Aquatic invertebrates (sweep nets, column samplers)		Total Phosphorus
Phytoplankton abundance (chlorophyll *a*)		Dissolved Phosphorus
		Dissolved Inorganic Carbon
		Dissolved Organic Carbon
		Seston Phosphorus
		Seston Carbon
		Seston Nitrogen

We are also investigating influences of earthworms on lake productivity and soil properties in 10 small watersheds within or near Itasca State Park, Minnesota. Worms were extracted from soils to measure biomass, and soil properties and lake properties characterized.

Objective 2: Intensive Lakes - evaluate and refine specific strategies for improving water quality and ecological characteristics of shallow lakes across Minnesota (hereafter intensive lakes).

We are evaluating responses of 12 shallow lakes (hereafter Intensive lakes) following recent lake restoration treatments such as draw downs or fish community manipulations. Treatments and timing varied, but generally included combinations of either partial or full drawdown, rotenone additions, and in some cases stocking of piscivorous fish (e.g. walleye) (Table 2).

Ecological characteristics of Intensive lakes were sampled during spring and summer 2010 and 2011 including parameters comparable to those measured in the Extensive sites. Landscape-level analyses are being 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 manipulation. Specific efforts will be directed to evaluating responses of the Intensive lakes to management efforts at each lake in a case-study format.

Table 2. Shallow lake enhancement strategies implemented on selected Intensive case study lakes.

Lake	**County**	**Size (acres)**	**Enhancement Strategy**	**Years Post-Treatment in 2011**
Nora	Pope	60	Full drawdown implemented in 2007. Began to refill in 2008, 40-50% open water by 2009. Metal half-riser structure with stoplogs functions as a fish barrier.	4 yrs
Sedan	Pope	62	Partial drawdown began in 2007, with a full drawdown occurring in 2008. Began to refill in 2009. Concrete variable crust structure with stoplogs regulates water level.	3 yrs
Wilts	Grant	55	Water levels were low in 2008 and lake is isolated, thus a decision was made to rotenone-treat the lake in fall 2008. Isolated basin.	3 yrs
Augusta	Cottonwood	499	This lake has a long history of drawdown to achieve wildlife benefits (pre-2004), but the most recent full drawdown occurred in 2008. Lake was re-flooded in 2009. Water control structure exists on lake outlet; control structures and high-velocity fish barrier installed on other adjacent waters within immediate watershed.	3 yrs
Hjermstad	Murray	60	Partial drawdown implemented in 2008, and lake was rotenone-treated under the ice during 2008-09. Fathead minnows persisted, so the lake was stocked with piscivores (walleye fry) in 2009 to attempt to suppress antecedent minnow populations. Water control via weir with stop logs; hanging finger fish barrier in place.	3 yrs
Maria	Murray	425	Full drawdown implemented from fall 2006 through fall 2007. Electric fish barrier was placed at lake outlet. Lake was rotenone-treated under the ice in February 2007. As of 2010, water levels remain low, much of lake remains covered with very dense stands of emergent cattail and fish persist in basin. Water control via weir with stop logs.	4 yrs
Spellman	Yellow Medicine	300	A managed drawdown occurred on this basin from 2006-08. 2009 was the first year with full water in the south basin. Box inlet culvert, outlet pipe, and finger-gate fish barrier in place.	3 yrs
Teal	Jackson	91	Partial drawdown implemented in 2008, and	3 yrs

			lake was rotenone-treated under the ice during winter 2008-09. Water control structure allows partial drawdown; no fish barrier in place at present.	
Froland	Pope	14	Fish removal via rotenone Oct 2010	1 yr
Leverson	Grant	20	Fish removal via rotenone Oct 2010	1 yr
Todd North	Grant	28	Fish removal via rotenone Oct 2010	1 yr
Todd South	Grant	22	Fish removal via rotenone Oct 2010	1 yr

Objective 3: Assess cost-effectiveness of alternative lake management methods - develop region-specific guidelines useful for identifying cost effective reclamation approaches

An economic analysis is being conducted using results from Extensive and Intensive Lakes to assess costs of water quality improvements (such as cost per unit of algae reduced [µg/L chla]) following application of various management options being used in Minnesota. We plan to contrast cost effectiveness of combinations of upland vegetation restoration (conversion of agriculture to grass) and in-lake habitat enhancements (fish removal, installation of barriers, etc.) required to achieve given levels of lake water quality improvement. We expect that costs of management options will vary widely among ecological regions due to variability in lake characteristics, lake watersheds, 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.

RESULTS AND DISCUSSION

During 2010 and 2011, we completed sampling of all Extensive and Intensive Lakes and most laboratory work. Here we show preliminary trends in water quality, submerged aquatic plants, and fish populations in Extensive sites. We also show data illustrating patterns in Intensive Lakes and compare these trends to values observed in Extensive sites. Results shown are preliminary and are subject to change after additional data analysis and interpretation.

Regional patterns in total phosphorus (TP) – In 2010, TP values in study lakes showed an increasing trend along a general north-south gradient from Red Lake to Windom areas, with highest median values recorded for lakes within the Windom core (Figure 2). Smallest variation in TP among lakes was observed within the Red Lake and Itasca study areas, where mean and range values were < 5 um L^{-1}. Median values in the Metro were comparable to those observed in other areas, but showed greater variability due to one record of extremely high TP (>20 um L^{-1}). Final TP values for 2011 are not yet available, but are expected soon.

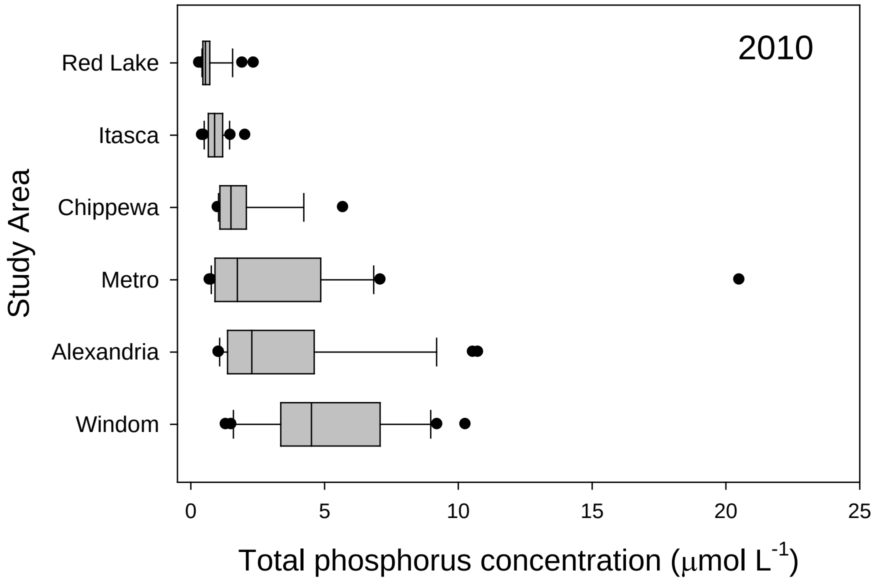

Figure 2 Box plots showing mean abundance of total phosphorus (TP) for shallow lakes sampled within 6 study regions during 2010. Vertical lines within boxes depict median TP values for each study region; extent of boxes depict 25th and 75th percentiles. Whiskers show 10th and 90th percentiles, with dots indicating more extreme values.

Submerged aquatic plants, phytoplankton, and regime implications – In general, phytoplankton abundance was higher in lakes in west-central and southern study areas, especially in Windom lakes, where values for chla often exceeded 100 ug L^{-1}. Other recent research on shallow Minnesota lakes suggests that sites where summer chla > 31 ug L^{-1} most often showed characteristics of turbid-regimes (Zimmer et al. 2009). Many lakes in Red Lake, Itasca, Chippewa, and Metro study areas were characterized by phytoplankton levels falling in a range expected for clear-regimes. In contrast, many Alexandria and Windom lakes showed phytoplankton levels in excess of thresholds expected for turbid-regime lakes (Figure 3). This indicates higher probability for lakes in Alexandria and Windom to show characteristics of turbid regimes.

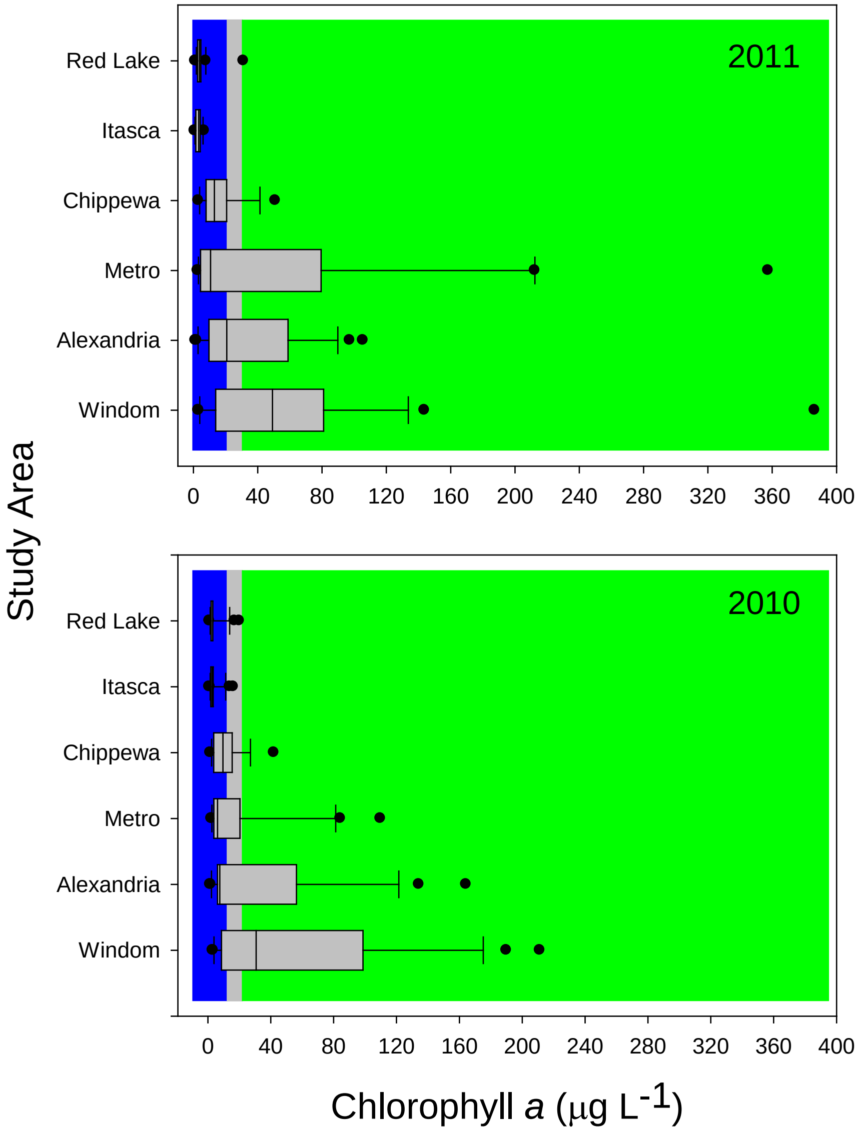

Figure 3 Box plots showing mean abundance of phytoplankton (chlorophyll *a* concentration) for shallow lakes sampled within 6 study regions during 2010 and 2011. Background colors depict expected chlorophyll *a* regions for clear- (blue), transition (grey), and turbid-regimes (green) based on threshold values after Zimmer et al. (2009).

Submerged aquatic plants are key ecosystem components of lakes and dense plant communities are known to favor clear-water regimes in Minnesota's shallow lakes. Our preliminary data show that abundance of submerged plants varies widely from lake-to-lake and among study areas in Minnesota. In general, lakes supporting high mass of submerged plants showed relatively low abundance of phytoplankton (chla). Alternatively, lakes with chla concentrations above 31 µg L^{-1} showed lower abundance of submerged macrophytes (Figure 4). This pattern was especially obvious during 2010, but more lakes showed transitional characteristics in 2011 (with higher chla and submerged plants). Preliminary data also suggested that these patterns vary considerably among study areas. For example, most Windom, Alexandria, and Metro-area lakes showed high abundance of either macrophytes or phytoplankton (but not both). In contrast, lakes in our Red Lake, Itasca, and Chippewa study areas were dominated by macrophytes, but abundance of macrophytes and phytoplankton were much lower than in other ecoregions. This seems to suggest that primary producers in Itasca and Chippewa lakes were limited by factors other than light limitation.

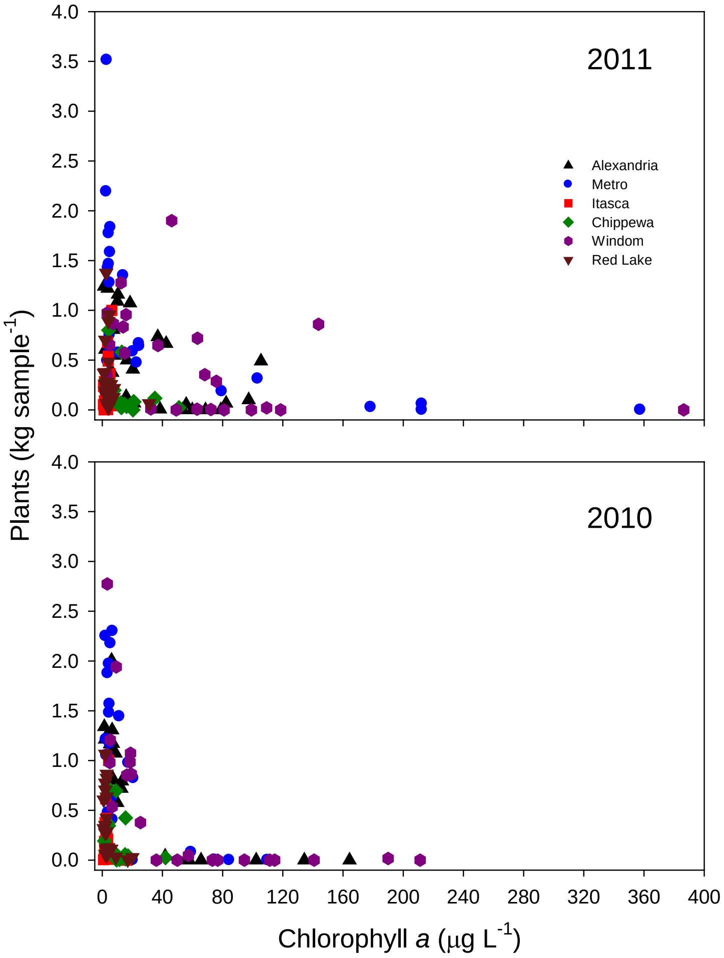

Figure 4 Abundance of phytoplankton (chlorophyll *a* concentration) and submerged macrophyte biomass for shallow lakes in 5 study regions during 2010 and 2011. Plant mass indicates average wet weight of plants collected on rake casts in each lake. Colors depict study area as indicated in caption.

Regional patterns in fish communities – Complex fish communities were found in lakes of all study areas. We observed no fishless lakes in our Chippewa study area, but we sampled 7 fishless sites in our Itasca study area; smaller numbers of fishless sites were observed in all other areas (Figure 5). Highest fish species richness (13) was observed in Windom lakes during 2011. Common carp (*Cyprinus carpio*) were less widely distributed among our lakes than we expected. During 2010, we collected carp from 23, 14, and 14 % of lakes in Windom, Metro, and Alexandria areas, respectively. In 2011, carp were sampled in 30, 13, and 29 % of Windom, Metro, and Alexandria lakes. Carp were not collected from lakes in Chippewa, Itasca, or Red Lake study areas during 2010 or 2011. In contrast, bullheads were ubiquitous and were collected from lakes in all study areas during 2010 and 2011. Bullhead abundance was highest in Windom, Metro, and Alexandria areas (> 46% in 2010 and $\geq$ 60% during 2011). Still, bullheads were collected in only 9% of Itasca lakes during both study years.

We also summarized relative abundance (mean total mass sampled) of predominant fish feeding guilds (planktivores [e.g. fathead minnows, shiners, yellow perch], benthivores [e.g. bullheads, common carp], piscivores [e.g. northern pike, walleye, largemouth bass]) for lakes within each study area (Figure 6). In general fish mass was roughly comparable among Alexandria, Chippewa, Metro, and Windom lakes, but showed a trend toward lower abundance in Itasca and Red Lake areas during both study years. Highest mass of planktivorous fishes was collected from lakes in Chippewa and Windom areas; piscivores were usually collected in lower numbers than other guilds, but were sampled at higher levels than benthivorous species in Itasca lakes. Piscivore abundance was also relatively high in Chippewa lakes. Dense populations of planktivores (shiners, yellow perch) and benthivores (bullheads) were evident in Chippewa lakes, although conventional wisdom suggests that these species are more abundant in prairie regions.

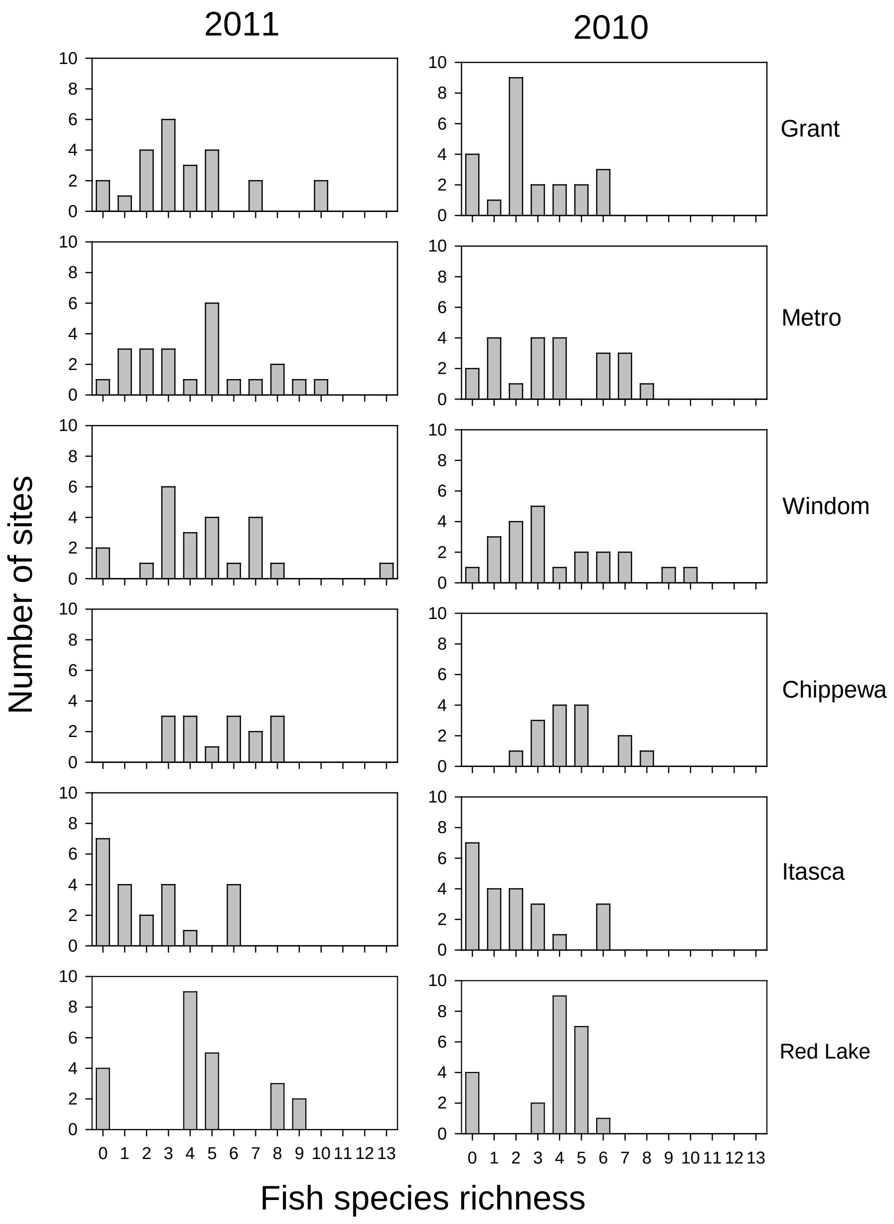

Figure 5. Fish species richness for shallow lakes in 6 study regions during 2010 and 2011, respectively. Height of bars on x-axis depicts number of lakes in which corresponding number of fish species were collected.

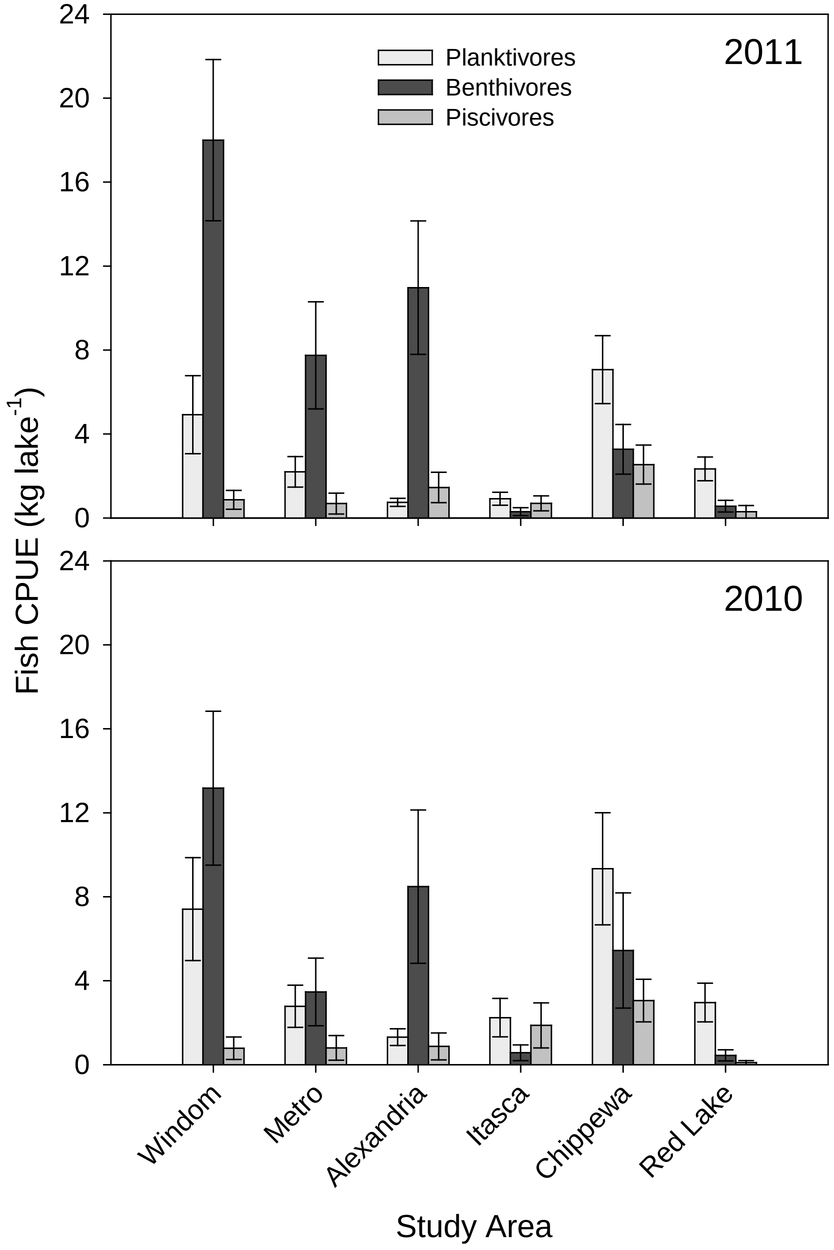

Figure 6 Summary of fish relative abundance for shallow lakes in 6 study regions during 2010 and 2011, respectively. Height of bars on x-axis depicts average weight (mass) for each of 3 major fish feeding guilds common in these lakes (planktivores, benthivores, piscivores).

Earthworm Sampling in Itasca study area - We continued investigating the influence of earthworms on lake productivity and soil properties in ten small watersheds within or near Itasca State Park, Minnesota. In addition to field work, we conducted experiments where we manipulated worm biomass and examined effects on dissolved organic phosphorus (DOP), dissolved organic carbon (DOC), and total nitrogen (TN). The results of these experiments suggested that the presence of earthworms increased the availability of organic carbon and nitrogen in soils. One of the important implications of these results is that watersheds with abundant anecic worms are likely to have depleted organic carbon and nitrogen pools due to increased lability of organic matter and increased leaching into surrounding water bodies.

Evaluation of Intensive Lakes - We have begun compiling data on aquatic plants, fish communities, and other characteristics of intensive sites so lake responses can be assessed.

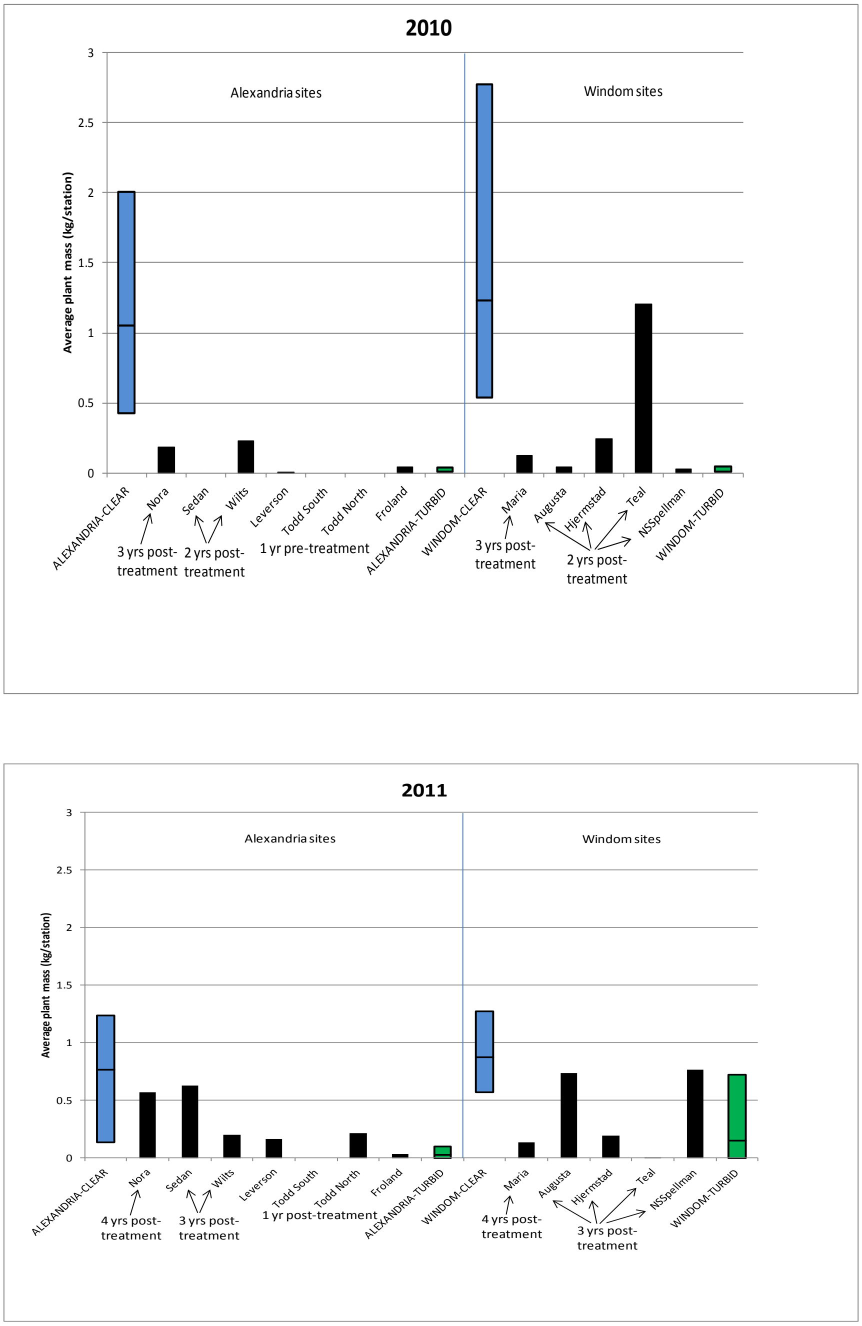

Figure 7. Black bars indicate relative abundance of submerged aquatic plants (average mass per sampling station) in Intensive Lakes in Alexandria and Windom study regions during 2010 (top panel) and 2011 (bottom panel). Blue bars indicate means and ranges of regional Extensive Lakes in clear-water regime; green bars indicate means and ranges of regional Extensive sites in turbid-water regime (based on threshold values of Zimmer et al. 2009).

We depicted relative abundances of plants and fish to allow comparisons between individual Intensive Lakes and Extensive Lakes within the same study region (Windom or Alexandria). We included Extensive Lake data averages and ranges of clear and turbid sites as a basis for comparisons. Resulting patterns indicated that plant communities were less extensive in Alexandria Intensive Lakes than in most regional Extensive sites in 2010 (Figure 7). By 2011, 2 of Alexandria 3 lakes rehabilitated 3 or more years earlier showed improvement in submerged macrophytes, while ½ of the sites treated the year previous with rotenone application showed improvement. Plant communities in Windom Intensive Lakes showed variable improvements relative to regional data ranges. Initial improvements in one lake (Teal) faded between 2010 and 2011; 2 lakes improved with an additional year post-treatment (cf. Augusta and NS Spellman in 2010 and 2011), and one lake (Maria) had plant biomass similar to regional turbid lakes throughout 2010-11 (Figure 7). Overall, these results seem to indicate both initial improvements and extreme variability in responses following rehabilitation efforts. For example, Teal Lake (Windom) showed high submerged plant abundance during 2010, but plants disappeared by 2011, just 3 years following rotenone treatment.

Relative abundance of planktivorous and benthivorous fishes was variable, but often high in Intensive Lakes, especially in Windom (Figure 8). Fish abundance was also extremely variable in Extensive Lakes in both regions, so comparisons between Intensive and Extensive Lakes may be less informative than expected. Still, Windom sites showed a trend toward higher abundance of planktivorous and benthivorous fishes than did Alexandria lakes and this was evident during 2010 and 2011. High fish abundance in 3 or 4 Windom sites was well within the range indicated for regional turbid lakes, probably indicating that fish successfully repopulated these lakes during 2 or 3 years following drawdown and reflooding. Interestingly, one of the Windom sites (Teal) had consistently low fish abundance, and yet water quality (as measured by plants – Figure 7) remains low, suggesting something other than fish is influencing the regime status of this lake, perhaps watershed land use, internal nutrient loading, or both.

Relatively low plant and high fish abundance indicated that 3 or 4 Windom Intensive Lakes showed characteristics of turbid regimes. Using our data, is not possible to determine whether these sites failed to respond to rehabilitation (via drawdown and reflooding) or if fish quickly recolonized and caused trends back towards turbid conditions prior to onset of our study. It is possible that abundance of submerged plants and fishes may not reflect regime characteristics of Intensive sites. This is because shallow lakes exhibit stability in both turbid and clear regime conditions, so fish or plants may not be closely correlated with regime conditions in any single year.

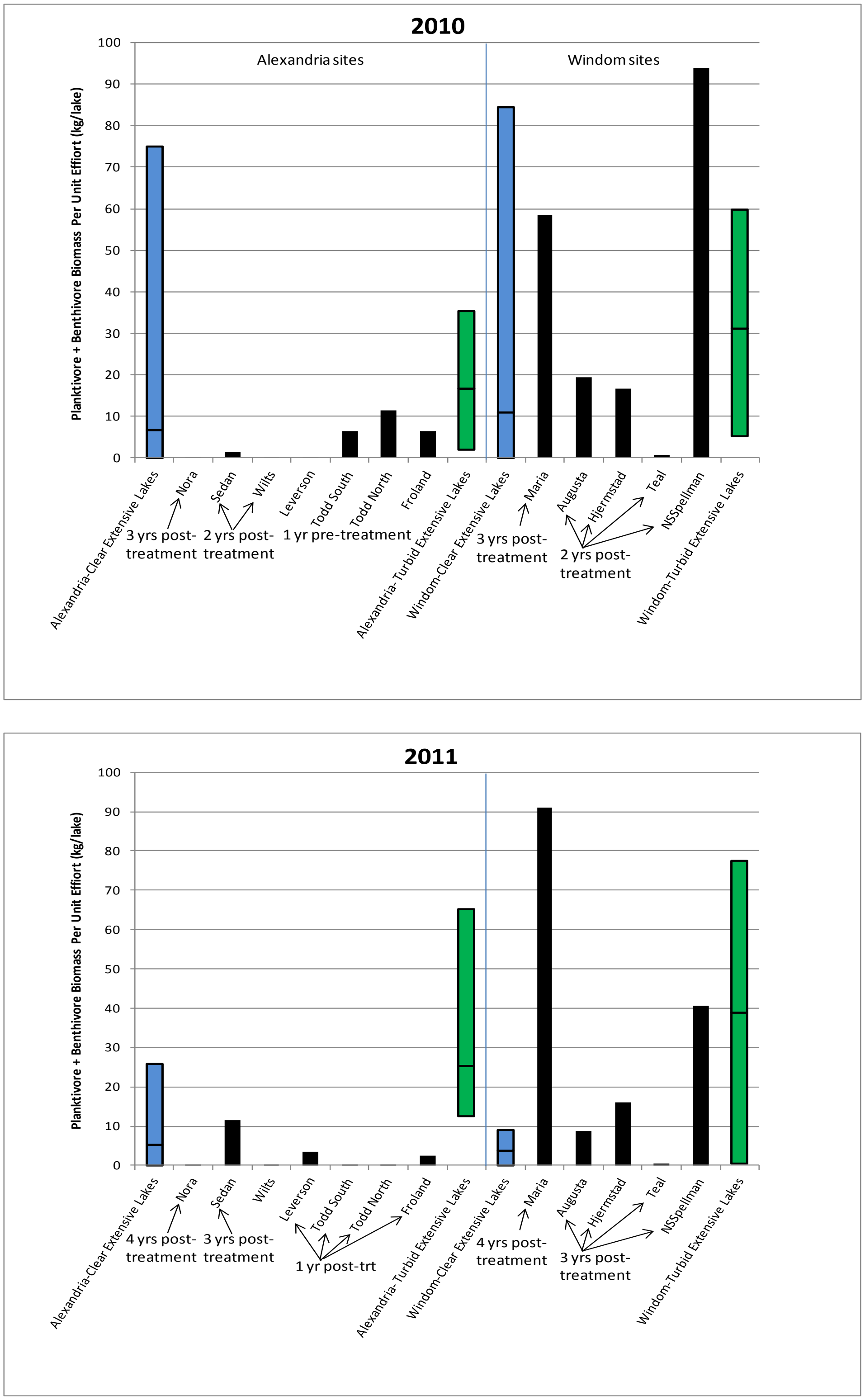

Figure 8. Black bars indicate relative abundance of planktivorous and benthivorous fishes (average mass per lake) in Intensive Lakes in Alexandria and Windom study regions during 2010 (top panel) and 2011 (bottom panel). Blue bars indicate means and ranges of regional Extensive Lakes in clear-water regime; green bars indicate means and ranges of regional Extensive sites in turbid-water regime (based on threshold values of Zimmer et al. 2009).

Cost-effectiveness – A cost-effectiveness assessment is being developed in two stages. First, using data from 2010, we identified high and low fish mass levels (planktivores and benthivores), and high and low agriculture sites (as a % of watershed area), by selecting 75th and 25th percentile values from Extensive Lakes in Alexandria and Windom study regions. We assumed in-lake rehabilitation would reduce fish abundance by 50% and 90% (lakes with low fish abundance are unlikely to be rehabilitated). Next, we used regression models to predict chla from combinations of fish mass (planktivores + benthivores) or extent (%) of watershed agriculture for these Extensive Lakes. Finally, we simulated likelihoods that lake rehabilitation by fish removal would reduce fish mass and chla to levels associated with clear-water regimes in our Minnesota study lakes (37 chla ug L^{-1}, updated threshold value from most recent analysis of present study data). Simulations identify the proportion of outcomes for which ending chla values (following lake management) were $\leq$ thresholds known for clear-water shallow lakes in Minnesota (Figure 9).

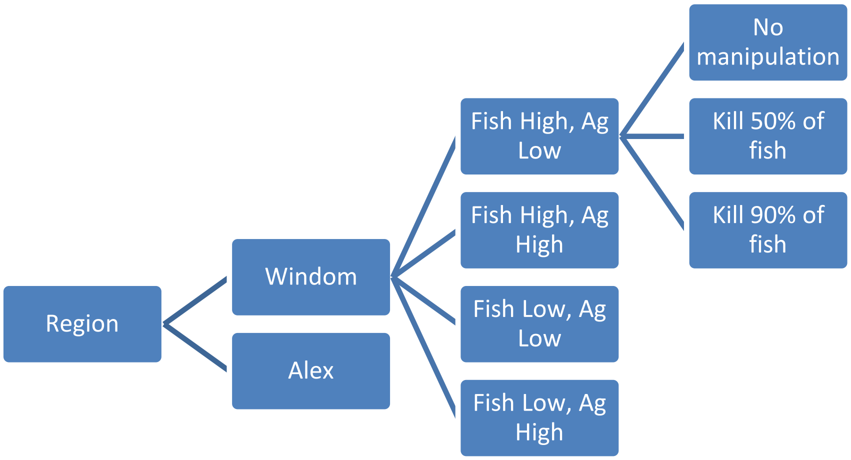

Figure 9: We considered a total of 24 scenarios (12 for each Region showing high fish mass and high proportions of agriculture in study landscapes). These scenarios capture 2 different values of %Ag (25th and 75th percentiles from within each region), 2 different starting levels of fish biomass, and 3 possible management scenarios (no reduction in fish biomass, 50% reduction, 90% reduction).

Results allow comparisons of likely water quality improvements (chla reductions) following lake rehabilitation via drawdown or rotenone (reducing benthivorous and planktivorous fish) or by restoration of upland cover (such as replacing row crops with grass). For example, results indicate that fish removal to 10 kg has approximately 70% likelihood of reducing chla levels to 37 ug L^{-1}, a level observed to be associated with clear-water regimes in Extensive Lakes (Figure 10). Results also indicate that a wide range of chla responses are likely in response to reductions in watershed agriculture. Comparing such patterns seems to suggest that within-lake rehabilitation has higher potential for reducing chla in shallow lakes than does conversion of watershed cover types. Still, we caution that presents results reflect short-term reductions in chla. Lake rehabilitation by drawdown or rotenone application often induces only brief increases in lake water quality. Most likely, restoration of grass in lake watersheds will ultimately favor long-term, sustainable reductions in chla and other water quality improvements, but improvements following watershed-scale enhancements may be delayed due to nutrient accumulation in lake sediments or other factors (Hobbs et al. 2012).

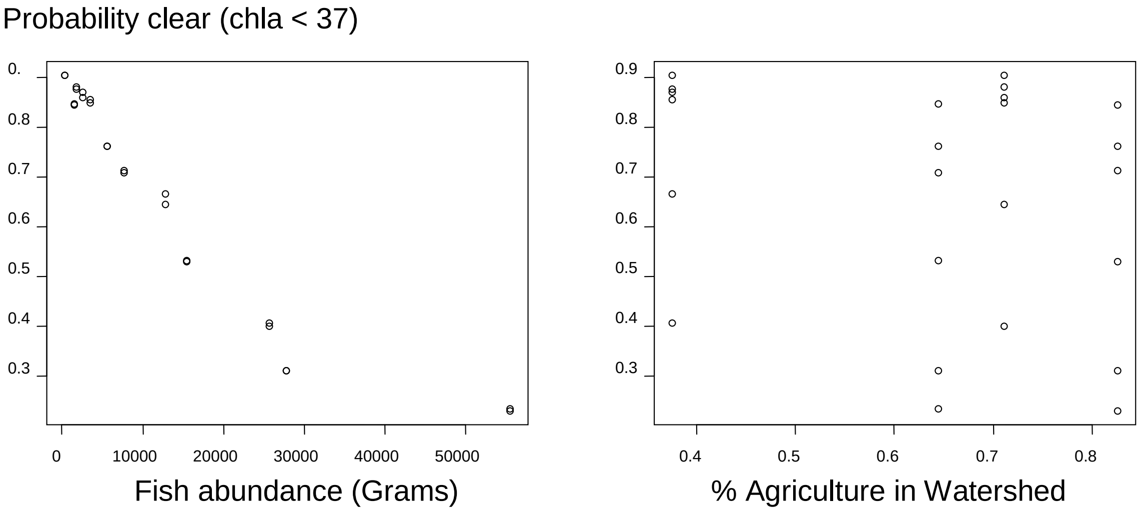

Figure 10. Summary of simulation results (probability of clear state [chla < 37] as a function of ending fish abundance [planktivore + benthivore fish mass] and % Agriculture in Lake Watersheds.

A second stage involves summarizing physical costs and results associated with each of our Intensive lakes and using these to valuate benefits to lakes. Through assistance of project collaborators (Ducks Unlimited, Minnesota DNR Shallow Lakes Program, USFWS, and others), we have secured data to support this larger cost-effectiveness analysis. Collaborators are also providing lake-specific costs of fish removal methods, installation and maintenance of water level control structures, seasonal water level drawdown, conservation easements, staff time, and other various physical and operational needs required for completion of these rehabilitation projects. Costs and results of of lake rehabilitation will be evaluated in case study manner. We believe this combined approach (results from Extensive and Intensive lakes) has the best potential for reflecting cost details associated with lake rehabilitation, and should ultimately allow cost-effectiveness comparisons of lake- vs. watershed-scale approaches.

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MODELING AND ESTIMATION OF HARVEST PARAMETERS AND ANNUAL SURVIVAL RATES OF WOOD DUCKS IN MINNESOTA, 1996–2010

James B. Berdeen

SUMMARY OF FINDINGS

Knowledge of the variables that influence harvest parameters and demographic vital rates is important for the management of game species. Harvest regulations are thought to have influenced the harvest parameters (e.g., Conroy et al. 2005) and demographic vital rates (e.g., Sheaffer et al. 2005) of some gamebird species. Consequently, there has been some concern regarding the increase in the daily bag limit of wood ducks (*Aix sponsa*) from 2 to 3 in the Mississippi Flyway (2008) and Minnesota (2011). The predicted harvest rates of adult males when the daily bag limits were 2 (0.087 [95% CI: 0.078–0.096]) and 3 (0.093 [95% CI: 0.077–0.109]) during a 60-day hunting season (Balkcom et al. 2010) suggest that the increase in hunting mortality would be slight, but this prediction has not been examined with empirical data.

To better understand the population biology and harvest dynamics of wood ducks, I conducted preliminary analyses on wood duck band and recovery data collected in Minnesota during 1996–2010. The specific objectives of these analyses were to (1) develop and evaluate band and recovery models of direct-recovery, recovery, and annual survival rates, (2) generate estimates of these parameters from the best approximating models, (3) convert estimates of direct-recovery rates to estimates of harvest rates, and (4) estimate the cohort-specific sample sizes of marked individuals needed to generate precise estimates of parameters of interest.

I developed and evaluated 44 approximating models to determine the influence of the predictors age, sex, time (year, aggregations of years corresponding to the periods during which there were 2- and 3-bird daily bag limits), and geographic strata of capture (ecological provinces, statewide) on recovery rates and annual survival rates. I conducted analyses using the Brownie et al. Recoveries option in Program MARK (Brownie et al. 1985, White and Burnham 1999) and evaluated the support for each model based on QAICc–values (Burnham and Anderson 2002).

I also developed and evaluated 39 logistic regression models to ascertain the influence of age, sex, time (year, aggregations of years corresponding to the periods during which there were 2- and 3-bird daily bag limits), and geographic strata of capture (ecological provinces, statewide) on direct-recovery rates (see Otis and White 2002). I analyzed data using the Known Fate option in Program MARK and evaluated the support for each model based on AICc–values (Burnham and Anderson 2002). I converted the direct-recovery rate estimates associated with the best approximating logistic regression model to harvest rate estimates using the equation of Henny and Burnham (1976).

To guide future banding efforts, I estimated the sample size of banded individuals in each age-sex cohort needed to generate precise estimates of harvest rates. I first fitted exponential curves to the relationship between the sample size of banded wood ducks in each age-sex-year cohort and the corresponding estimated coefficients of variation of harvest rates $C\hat{V}(\hat{H})$ associated with the best approximating model. I then solved these equations to estimate sample sizes necessary to achieve $CV(H)$–values of 0.05 and 0.10.

The best approximating Brownie et al. (1985) model suggested that recovery rates and annual survival rates varied by age, sex, and ecological provinces in additive relationships. This model had 76% of total weight. The parameter estimates associated with this model indicate

that within each ecological province, the AHY-M cohort had the greatest annual survival rate estimate, followed by the HY-M, AHY-F, and HY-F cohorts. Further, the greatest annual survival rate estimates were associated with the Forest ecological province, followed by the Prairie, Aspen Parkland, and Transition ecological provinces.

The best approximating logistic regression model indicated that direct-recovery rates of wood ducks varied by age, sex, and time (year) in an additive relationship, and had 52% of model weight. The direct-recovery rate estimates associated this model are greatest for the HY-M cohort, followed by the HY-F, AHY-M, and AHY-F cohorts. There is little evidence that the temporal variation of direct-recovery rates varied is attributable to the change in daily bag limit in the Mississippi Flyway. Further, the harvest rate estimates of the AHY-M cohort during the years in which daily bag limits were 2 and 3 were not significantly different than those predicted by Balkcom et al. (2010) during 13 of 15 years, as indicated by overlapping 95% CIs.

These best approximating models suggest that mortality rate estimates (1 – annual survival rate estimate) vary at the spatial scale of ecological provinces but direct-recovery rates vary at a statewide scale. These models also suggest that annual survival was constant but direct-recovery rates varied annually during 1996–2010. Thus, the spatiotemporal characteristics of mortality of wood ducks in Minnesota appear complex. Further research is needed to develop a better understanding of the influence of harvest regulations and habitat characteristics on harvest parameters and demographic vital rates.

Preliminary estimation of sample size needs indicate that 952 AHY-M, 209 AHY-F, 259 HY-M, and 276 HY-F should be banded to achieve $CV(H)$–values of 0.10. If a $CV(H)$–value of 0.05 is desired, 2107 adult males, 440 AHY-F, 605 HY-M, and 622 HY-F should be banded. The cohort-specific sample sizes seem attainable, especially if $CV(H)$–values of 0.10 are desired.

INTRODUCTION

Reliable knowledge of harvest parameters and demographic vital rates are necessary for the management of populations of game species. Harvest regulations are thought to have influenced the harvest parameters (e.g., Conroy et al. 2005) and demographic vital rates (e.g., Sheaffer et al. 2005) of some gamebird species. Data from reencounters of banded gamebirds can be used to estimate to estimate several of these parameters, including direct-recovery, recovery, and annual survival rates (Brownie et al. 1985). Harvest rates can be estimated if both direct-recovery rate and band-reporting rate estimates are available (Henny and Burnham 1976).

There has been some concern regarding the increase in the daily bag limit from 2 to 3 for wood ducks (*Aix sponsa*) in the Mississippi Flyway (2008) and Minnesota (2011). During a 60-day hunting season, the predicted harvest rates of adult males are 0.087 (95% confidence interval [CI]: 0.078–0.096) and 0.093 (95% CI: 0.077–0.109) when the daily bag limits are 2 and 3, respectively (Balkcom et al. 2010). Unfortunately, an examination of these predictions have not been performed on empirical data and published. Further, harvest rates for other age-sex cohorts were not predicted by Balkcom et al. (2010) and changes in cohort-specific annual survival rates in response to this regulatory change have not been examined.

To better understand the population biology and harvest dynamics of wood ducks, I performed analyses on band and recovery data from wood ducks captured in Minnesota during 1996–2010. The results of these analyses can be used to improve the understanding of the sources of variation of harvest parameters and annual survival rates, identify information gaps, and provide the information needed to improve the existing Minnesota Department of Natural Resources (MNDNR) banding program.

The specific objectives of these analyses were to:

(1) Develop and evaluate band and recovery models of direct-recovery, recovery, and annual survival rates,

(2) Generate estimates of these parameters from the best approximating models,

(3) Convert estimates of direct-recovery rates to estimates of harvest rates, and

(4) Estimate the cohort-specific sample sizes of marked individuals needed to generate precise estimates of parameters of interest.

METHODS

I used the banding data from (1) normal, wild, and flighted wood ducks of known age and sex that were captured in Minnesota during July – mid-September (i.e., prehunting season) 1996–2010, (2) marked with a standard U.S. Geological Survey (USGS) leg band on which a toll-free telephone number or website at which reencountered bands could be reported, and (3) captured via standard methods (e.g., rocketnetting [Wunz 1984], baited swim-in traps [Mauser and Mensick 1992]) or nightlighting (Lindmeier and Jessen 1961) and held <24 h. I used recovery data from (4) unsolicited bands that were reencountered during the 1996–1997 to 2010–2011 hunting seasons. There were 12,220 bandings, 994 direct recoveries, and 860 indirect recoveries that met these criteria.

I initially stratified capture and recovery data by age class (2 levels: after hatch-year [AHY] or hatch-year [HY]), sex (M or F), time (years of capture and recovery), and ecological province of capture (4 levels: Tallgrass Aspen Parkland [Aspen Parkland], Prairie Parkland [Prairie], Eastern Broadleaf Forest [Transition], Laurentian Mixed Forest [Forest]; Hanson and Hargrave 1996). Unfortunately, birds were banded in the Forest ecological province during only 2004–2010.

Estimation of Recovery and Annual Survival Rates

I used maximum likelihood methods to model and estimate the recovery rate (f) and annual survival rate (S) of wood ducks. Recovery rate is defined as the probability that a banded duck is shot or found dead during the hunting season, retrieved, and reported (Williams et al. 2001). Annual survival rate is defined as the probability that a banded duck alive at the time of banding in year *t* survives until the banding period in year *t* + 1 (Williams et al. 2001).

I developed and examined the support for the global model, $f_{age*sex*time*ecological\ province}$ $S_{age*sex*time*ecological\ province}$, in which f and S varied by age, sex, time (year), and ecological province of capture in an interactive relationship. I also developed 43 reduced models in which these variables in different combinations and structural relationships were used as predictors. To ascertain whether a regulatory change influenced f and S , I developed some reduced models in which time (years of capture and recovery) was aggregated into 2 time periods that corresponded with the years during which there were 2- and 3-bird daily bag limits. I also developed a subset of models in which there were 2 different spatial aggregations of the capture locations (i.e.; ecological provinces, statewide). The relative support for models with spatially aggregated capture data was used to make inferences about the sources of spatial variation of f and S.

I used the Brownie et al. Recoveries option in Program MARK (Brownie et al. 1985, White and Burnham 1999) to fit each model to the band and recovery data, generate parameter estimates and associated 95% CIs, calculate the Akaike Information Criterion (QAICc) adjusted for overdispersion and small sample size, and perform a bootstrap simulation to estimate a median $\hat{c}$–value associated with the global model. I evaluated the relative support for each model based on the QAICc–values (Burnham and Anderson 2002).

Recovery rate estimates generated in this analyses could be used as an index of harvest rates, but (1) there is covariance between this parameter and S (Brownie et al. 1985), and (2) the parameterization of both S and f have some influence on the likelihood function associated with each model (see Brownie et al. 1985:12). Thus, the parameterization of S has some influence on maximum likelihood estimates and consequently f in Brownie models. The parameterization and estimation of S is of primary interest in this analysis because of the need to understand the sources of variation of this parameter; specifically whether the change in the daily bag limit in the Mississippi Flyway had some effect on the annual survival of wood ducks. Consequently, I report the values of $\hat{S}$ but not $\hat{f}$ associated with the best approximating Brownie model. I estimate direct-recovery rates ($f*$) in a separate analysis to avoid the influence of S.

Estimation of Direct-Recovery and Harvest Rates

I modeled and estimate $f*$ using logistic regression methods (Otis and White 2002). This parameter is defined as the proportion of wood ducks banded in year *t* that is shot and retrieved during the first hunting season after banding, and reported to the BBL (Williams et al. 2001). I used the Known Fate option of Program MARK (White and Burnham 1999) to fit candidate models to the data, generate parameter estimates, calculate AICc–values, and rank the models based on these values (Burnham and Anderson 2002). The global model, $f*_{age*sex*ecological\ province*time}$, used the variables age, sex, year, ecological province of capture, and time in an interactive relationship as predictors of $f*$. The 38 reduced models used different combinations of these predictors in interactive and additive relationships.

I converted $\hat{f}*$–values from my best approximating logistic regression model to $\hat{H}$ with the equation of Henny and Burnham (1976):

$$f* = H\lambda,$$

in which the value of band-reporting rate estimate ($\hat{\lambda}$) was 0.73 (1 $S\hat{E}$ = 0.026; Zimmerman et al. 2010). I used the delta method (Seber 1982) to estimate the variance ($\hat{\text{var}}$) of $\hat{H}$.

Estimation of Sample Size

Although Program Band2 (Wilson et al. 1989) can be used to estimate cohort-specific sample sizes necessary to precisely estimate annual survival rates, a sample size of 300 individuals per cohort generally is required to estimate this parameter (Brownie et al. 1985). However, this sample size guideline now may be <300 because (1) modern analytical software has greater capabilities than the program used when the original sample size recommendation was made, and (2) there has been an increase in $\hat{\lambda}$ from 0.32 (1 $S\hat{E}$ = 0.020) when the original guidelines were established to 0.73 (1 $S\hat{E}$ = 0.026) during the contemporary period (Zimmerman et al. 2010).

To ascertain the sample size of each age-sex cohort that must be banded to generate precise estimates of harvest rates, I fit an exponential curve to the relationship between the sample size of banded wood ducks of each age-sex-year cohort and the corresponding $C\hat{V}(\hat{H})$ value. I solved the exponential equation associated with the curve of each age-sex cohort in a manner that the sample sizes necessary to achieve $CV(H)$–values of 0.05 and 0.10 were estimated.

RESULTS

Recovery and Annual Survival Rate Estimates

The values of AICc associated with all models and variances associated with parameter estimates were adjusted with the median $\hat{c}$–value of the global model, 1.065. Four Brownie et al. (1985) models have weights of $\geq$0.5% (Table 1). Models in which annual survival varies by age, sex, ecological province, time (year or Flyway bag limit) have 81.8, 99.8, 76.1, 0, and 0%, respectively, of model weight.

The best approximating Brownie et al. (1985) model of recovery and annual survival rates, $f_{\,age+sex+ecological\ province}\ S_{\,age+sex+ecological\ province}$, uses age, sex, and ecological province in an additive relationship as predictors of f and S, and has 76.1% of model weight (Table 1). The parameter estimates associated with this model indicate that in general, the AHY-M cohort had the greatest annual survival rates, followed by the HY-M, AHY-F, and HY-F cohorts in each ecological province (Table 2). Within each age-sex cohort, the annual survival rate estimates associated with the Forest ecological province are greatest, followed by Prairie, Aspen Parkland, and Transition ecological provinces (Table 2). Overlapping 95% CIs associated with some pairs of age-sex-ecological province cohorts suggest that differences are not significant. However, nonoverlapping 95% CIs indicate that the annual survival rate estimates of AHY-M birds captured in the Prairie, Transition, and Aspen Parkland ecological provinces are significantly greater than the AHY-F and HY-F cohorts captured in the same geographical strata (Table 2).

Direct-recovery and Harvest Rate Estimates

Nine logistic regression models have weights of $\geq$0.5% (Table 3). Approximating models in which age, sex, time (year), time (Flyway bag limit), and ecological province of capture were predictors of direct-recovery rates have 98.5, 78.3, 68.7, 3.8, and 26.0% of model weight, respectively (Table 3). The best approximating model, $f^*_{\,age+sex+time}$, includes age, sex, and time (year) in an additive relationship as predictors and has 52.1% of the model weight.

I used a band-reporting rate of a single value to convert $\hat{f}^*$ from the best approximating logistic regression model to $\hat{H}$ (Figure 1a–d). Consequently, the relationships among $\hat{H}$ associated with each age-sex-year cohort are the same as those of $\hat{f}^*$ from the best approximating model.

Sample Size Estimates

Only 7 of 240 age-sex-year-ecological province cohorts have sample sizes of banded birds that met the recommended guideline of 300 marked individuals (see Brownie et al. 1985). However, 11 of 16 estimated coefficients of variation associated with annual survival rate estimates ($\hat{CV}[\hat{S}]$) of age-sex-ecological province cohorts generated in the best approximating Brownie model are <0.10, and 3 of 16 are <0.05. All age-sex cohorts captured in the Forest ecological province and HY-F cohort captured in the Aspen Parkland ecological province have associated $\hat{CV}(\hat{S})$–values >0.10. In contrast, only AHY-M captured in the Prairie, Transition, and Aspen Parkland ecological provinces are the only cohorts with $\hat{CV}(\hat{S})$–values <0.05.

Generally, the $\hat{CV}(\hat{H})$–values are greatest for the AHY-F cohort and lowest for the AHY-M cohort. However, only 1 of 60 age-sex-year cohorts has a $\hat{CV}(\hat{H})$–value of <0.10. I used the equations in Figure 2a–d to estimate sample sizes of each age-sex cohort that are necessary to generate $CV(H)$–values of 0.05 and 0.10. I present these estimates in Table 4.

DISCUSSION

Modeling and Estimation of Annual Survival Rates

Age, sex, and ecological province of capture appear to be important predictors of the annual survival rates of wood ducks banded in Minnesota during 1996–2010. Variation among age-sex cohorts is somewhat consistent to those of other analyses of wood ducks banded in the northern portion of the Mississippi Flyway during earlier periods (Nichols and Johnson 1990, Bellrose et al. 1994). In each of these studies, the AHY-M cohort had the greatest annual survival rate estimates, followed by HY-M, AHY-F, and HY-F (Table 2).

This and other studies have detected geographic variation of annual survival rates, but the different spatial scale at which these analyses were conducted restricts the ability to make comparisons. Previous studies indicated that wood ducks banded in the southern U.S. generally had greater annual survival rate estimates than did those banded in northern states (Nichols and Johnson 1990, Bellrose et al. 1994). My preliminary results suggest that variation of annual survival is better explained at the spatial scale of ecological provinces in Minnesota than at that of this state. It may be that habitat characteristics associated with ecological provinces influenced survival. Alternatively, annual survival rates may have been influenced by some characteristic unrelated to habitat but associated with capture areas.

There was a lack of support for any Brownie et al. (1985) models in which annual survival varied either annually or by groups of years that corresponded to the daily bag limits in the Mississippi Flyway. These results suggest that annual survival was relatively constant during 1996–2010, and was not influenced by the liberalization of this hunting regulation in the Mississippi Flyway. Temporal variation of this parameter also was not observed in either the northcentral U.S. during 1966–1985 (Nichols and Johnson 1990) or the northern Mississippi Flyway during 1960–1985 (Bellrose et al. 1994). However, increasing trends in annual survival rates were detected in 9 of 10 sex-geographic area cohorts during 1959–1996 (Franklin et al. 2002). It may be that the sample sizes of banded wood ducks were insufficient to detect temporal changes in annual survival rates in some of these studies. Thus, it is important to mark an adequate sample size of individuals in each cohort if it is a priority to monitor the demographic response to changes in hunting regulations.

Although it is difficult to compare results of studies conducted at different spatiotemporal scales, an examination of the annual survival rate estimates of each age-sex cohort could provide direction for future investigations and population management. The annual survival rate estimates of the AHY-M, HY-M, and HY-F birds banded in Minnesota during 1996–2010 were at least slightly greater than the corresponding cohorts banded in Illinois, Iowa, Minnesota, Missouri, and Wisconsin during 1966–1985 (hereafter, northcentral U.S.; Nichols and Johnson 1990) and in portions of the eastern Dakotas, Manitoba, Michigan, Minnesota, Ontario, and Wisconsin (hereafter, northern Mississippi Flyway) during 1960–1985 (Bellrose et al. 1994, Table 2). The annual survival rate estimate of the AHY-M cohort captured in the Prairie ecological province during 1996–2010 is significantly greater than this age-sex cohort banded in both the northcentral U.S. during 1966–1985 and the northern Mississippi Flyway during 1960–1985 (Table 2). Similarly, estimates of the annual survival rate of HY-M birds captured in the Prairie, Transition, and Aspen Parklands ecological province during 1996–2010 are significantly greater than the same age-sex cohort in both the northcentral U.S. during 1966–1985 and the northern Mississippi Flyway during 1960–1985 (Table 2). The apparent spatiotemporal variation of the annual survival rates in 3 investigations (Nichols and Johnson 1990, Bellrose et al. 1994, this study) indicates a need to better understand why such differences existed.

Demographic vital rates of females often are used to estimate rates of population change in population projection matrices (e.g., Case 2000). In some species, survival especially influences the finite rate of population growth (Crone 2001). Juvenile survival may have the greatest value of elasticity in increasing populations and short-lived species, but the survival rates of the oldest age class generally have the greatest elasticity value in decreasing populations and long-lived species (see Sandercock 2006). Thus, it is particularly important to ascertain whether annual survival rates of the AHY-F or HY-F cohorts have the greatest influence on the rate of population growth.

Estimation of Direct-Recovery Rates and Harvest Rates

Age, sex, and time (year) appear to be important predictors of the direct-recovery rates of wood ducks in Minnesota during 1996–2010. Based on estimates of direct-recovery rates, it appears that the HY-M cohort was most susceptible to hunting mortality, followed by the HY-F, AHY-M, and AHY-F cohorts. This cohort-specific order of susceptibility is the same as that observed in the northern Mississippi Flyway during 1960–1985 (Bellrose et al. 1994) and in the northcentral U.S. during 1966–1985 (Nichols and Johnson 1990).

Models in which time (year) was a predictor of direct-recovery rates had 68.7% of model weight, but models in which time (Flyway bag limit) was a predictor had 3.8% of model weight. Thus, there is little evidence that the temporal variation of harvest parameters was attributable to a change in the daily bag limit in the Mississippi Flyway. Unexamined variables (e.g., hunter effort) probably had a greater influence on the likelihood of a banded wood duck being harvested than did the change in daily bag limit. The harvest rate estimates of AHY-M banded in Minnesota during periods in which there were a 2- and 3-bird daily bag limits 1996–2010 were not significantly different than those predicted by Balkcom et al. (2010) during 13 of 15 years, as indicated by overlapping 95% CIs. However, the effects of the 3-bird daily bag limit on harvest rates should continue to be monitored in both Minnesota and the Mississippi Flyway, given that (1) Minnesota enacted a 3-bird daily bag limit after the period of analyses, (2) the majority of direct recoveries of the female age-sex cohorts occurred in-state during the study (MNDNR, unpublished data), and (3) the observed temporal variation of this parameter estimate.

The best approximating models suggest that annual mortality rates ($M = 1 - S$) varied at the spatial scale of ecological provinces within Minnesota, but direct-recovery rates varied at a statewide scale. Thus, habitat characteristics associated with the ecological province of capture appeared to have influenced overall mortality rates but not one source of mortality (i.e., hunting). Alternatively, it may be that the spatial variation of direct-recovery rates is better explained by another habitat classification system, or that the sample size of banded individuals was insufficient to detect geographic variation. These approximating models also suggest that annual mortality rates were constant but hunting mortality rates varied annually during 1996–2010. Such results suggest that the spatiotemporal characteristics of mortality of wood ducks in Minnesota are complex. Further research is needed to develop a better understanding of the variables (e.g., harvest regulations, habitat characteristics) that influence harvest parameters and demographic vital rates.

Relationship between Harvest and Annual Survival Rates

The differences among age-sex cohorts with regard to estimates of harvest rates and overall mortality rates suggest that these cohorts are differently affected by hunting and other sources of mortality. I examined the associated harvest rate estimates in relation to mortality rate estimates ($\hat{H} / \hat{M}$, with parameter estimates generated in the best approximating models) to better understand the magnitude of hunting mortality of these cohorts. The values of these proportions are 13.7 – 39.7, 9.3 – 25.9, 19.4 – 53.3, and 13.3 – 35.5% for the AHY-M, AHY-F, HY-M, and HY-F cohorts, respectively. These estimates suggest that there was not an inverse relationship between harvest rate estimates and annual survival rate estimates among cohorts (e.g., the cohort with the lowest survival rate estimate [HY-F] did not have the greatest harvest rate estimate and the cohort with the greatest survival rate estimate [AHY-M] did not have the lowest harvest rate estimate) in Minnesota during 1996–2010. However, an inverse relationship detected between the recovery and annual survival rates of wood ducks during 1966–1984 suggests that hunting mortality had a negative influence on survival during that period (Trost 1990).

The spatiotemporal variability of annual survival in the northern Mississippi Flyway, significant temporal variation of contemporary harvest rate estimates of Minnesota-banded birds (Figure 1a–d), substantial variability of harvest rate estimates in relation to the overall mortality rate estimates, and lack of knowledge about both the point at which hunting mortality becomes additive and the contemporary relationship between harvest and annual survival rates indicate that a better understanding of the population ecology of wood ducks is needed. These topics should be addressed formally in well-designed studies using modern analytical methods; e.g., the Pollock et al. (1994) method to apportion overall mortality rates into hunting and natural mortality rates, and the Otis and White (2004) or Sedinger et al. (2010) method to examine the relationship between the harvest and annual survival rates of each cohort.

Sample Size Estimates

Future studies that address gaps in current knowledge of the population dynamics of wood ducks will require a representative sample of a sufficient number of banded individuals in each cohort of interest (e.g., age, sex, geographic area, time). An adequate sample size of banded individuals will permit the development of reliable models, generation of precise estimates of harvest parameters and annual survival rates, increase the likelihood of detecting significant differences among parameters estimates, and facilitate the evaluation of the effects of regulatory change on harvest parameters and annual survival rates. It is especially important to band an adequate sample size of those cohorts which (1) may be most important in influencing the rate of population change (e.g., females), (2) have relatively great $C\hat{V}$–values, and (3) traditionally have had relatively small samples sizes of banded individuals.

$C\hat{V}$–values >0.10 may indicate that the sample size of marked birds is too small or that the model structure does not describe the system well. Generally, the $C\hat{V}(\hat{S})$–values associated with the HY-F cohort were greatest and those associated with the AHY-M cohort were lowest, and those associated with the Forest ecological province were greatest and those captured in the Prairie ecological province were lowest in this study. Further, 11 of 16 $C\hat{V}(\hat{S})$–values associated with the best approximating Brownie model are <0.10. Thus, it appears that the sample size of banded birds and fit of the best approximating Brownie et al. (1985) model were sufficient to generate precise estimates of annual survival rates for all cohorts except HY-F captured in the Aspen Parkland ecological province and all age-sex cohorts captured in the Forest ecological province.

Unfortunately, 59 of 60 $C\hat{V}(\hat{H})$–values were >0.10. The lack of precision of these estimates may be attributed to (1) the use of variances associated with both direct-recovery rates and band-reporting rates in the estimation of $CV(H)$, (2) the likelihood that another logistic regression model would better describe the system of interest, and (3) an insufficient sample size of banded individuals in most cohorts. More specifically, the mean sample size of each age-sex cohort banded annually in Minnesota during 1996–2010 was: AHY-M = 478.6 (range: 238–952), AHY-F = 75.3 (range: 42–142), HY-M = 146.9 (range: 28–297), and HY-F = 113.7 (range: 26–237). Thus, it appears necessary to increase the sample size of banded individuals in most cohorts.

The cohort-specific sample size estimates necessary to generate precise harvest rate estimates (Table 4) and annual survival rate estimates (300 per cohort, Brownie et al. 1985) in general seem attainable. However, the low sample sizes associated with some cohorts banded during 1996–2010 suggest that it has been difficult to obtain adequate samples via standard capture methods. It may be necessary to use alternative capture methods in a more focused effort than the MNDNR has previously employed in their general banding program.

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Table 1. Model selection statistics for Brownie et al. (1985) models used to estimate the recovery (f) and annual survival rates (S) of wood ducks banded in Minnesota, 1996–2010.

Model	K	QAICc	ΔQAICc	ω_i
$f_{age+sex+ecological\ province}$ $S_{age+sex+ecological\ province}$	13	14,093.684	0	0.761
$f_{age+sex}$ S_{sex}	5	14,097.251	3.567	0.128
$f_{age+sex}$ $S_{age+sex}$	6	14,098.876	5.192	0.057
$f_{age\ x\ sex}$ S_{sex}	6	14,099.035	5.350	0.052

Table 2. Annual survival rate estimates of wood ducks banded in the northern Mississippi Flyway generated in 3 studies.

Period [a]	Geographic area [b]	Annual survival rate estimates (95% CI) of each age-sex cohort [c]			
		AHY-M	AHY-F	HY-M	HY-F
1996–2010	Prairie	0.607 (0.579– 0.634)	0.494 (0.439–0.548)	0.572 (0.492–0.649)	0.458 (0.376–0.542)
	Transition	0.556 (0.510–0.600)	0.441 (0.380–0.504)	0.520 (0.440–0.599)	0.406 (0.330–0.487)
	Aspen Parkland	0.567 (0.529–0.605)	0.453 (0.390–0.518)	0.532 (0.444–0.617)	0.418 (0.330–0.511)
	Forest	0.640 (0.469–0.781)	0.529 (0.356–0.695)	0.606 (0.420–0.767)	0.493 (0.315–0.673)
1960–1985	Northern Mississippi Flyway	0.543 (0.529–0.557)	0.472 (0.440–0.504)	0.411 (0.383–0.439)	0.398 (0.355–0.441)
1966–1985	Northcentral U.S. [j]	0.544 (0.530–0.558)	0.496 (0.469–0.523)	0.416 (0.392–0.440)	0.363 (0.332–0.394)

[a] Estimates from 1996–2010 were generated in this study, 1960–1985 estimates are from Bellrose et al. (1994), and 1966–1985 estimates are from Nichols and Johnson (1990).

[b] Geographic areas are as follows: Prairie = Prairie Parkland ecological province, Transition = Eastern Broadleaf Forest ecological province, Aspen Parkland = Tallgrass Aspen Parkland ecological province, Forest = Laurentian Mixed Forest ecological province (Hanson and Hargrave 1996) all within Minnesota; Northern Mississippi Flyway = portions of the eastern Dakotas, Manitoba, Michigan, Minnesota, Ontario, and Wisconsin; Northcentral U.S. = Illinois, Iowa, Minnesota, Missouri, and Wisconsin.

[c] Age-sex cohorts are AHY-M = adult male, AHY-F = adult female, HY-M = juvenile male, HY-F = juvenile female.

Table 3. Model selection statistics for logistic regression models used to estimate the direct-recovery rates ($f*$) of wood ducks banded in Minnesota, 1996–2010.

Model	K	AICc	ΔAICc	ω_i
$f*_{age+sex+time}$	17	6848.973	0	0.521
$f*_{age\ x\ sex\ x\ ecological\ province}$	16	6851.389	2.416	0.156
$f*_{age+time}$	16	6851.422	2.449	0.153
$f*_{age+sex+ecological\ province}$	6	6852.828	3.854	0.076
$f*_{age+sex+flyway\ bag\ limit}$	4	6854.690	5.717	0.030
$f*_{age+ecological\ province}$	5	6856.002	7.029	0.016
$f*_{age\ x\ time}$	30	6856.309	7.336	0.013
$f*_{age\ x\ ecological\ province}$	8	6856.463	7.490	0.012
$f*_{age+flyway\ bag\ limit}$	3	6857.426	8.453	0.008

Table 4. Estimates of the sample size of banded wood ducks needed to generate harvest rate (H) estimates with coefficients of variation (CV)–values of 0.05 and 0.10.

	$CV(H)$	
Age-sex cohort	0.05	0.10
AHY-M	2107	952
AHY-F	440	209
HY-M	605	259
HY-F	622	276

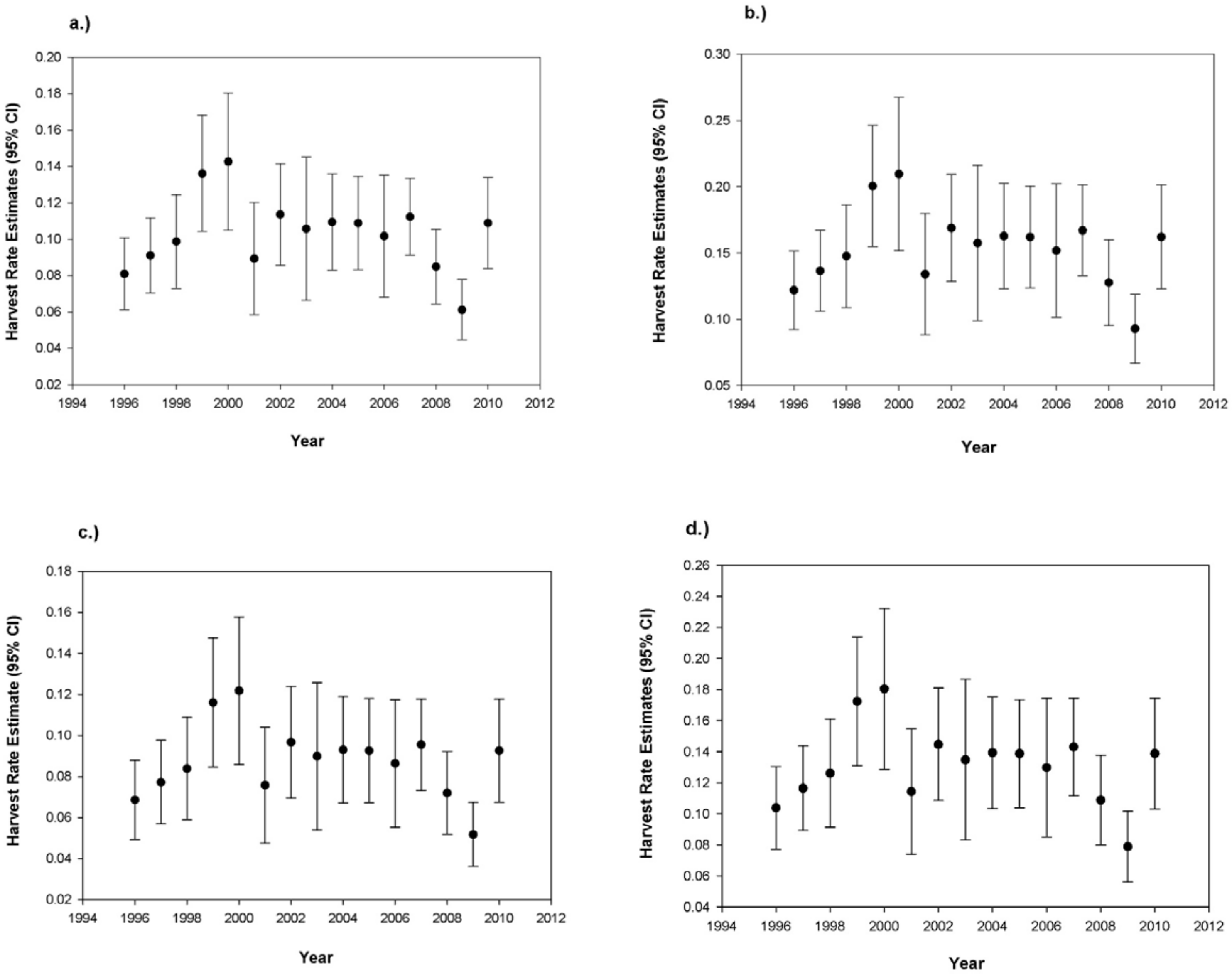

Figure 1a–d. Harvest rate estimates of a.) AHY-M, b.) HY-M, c.) AHY-F, and d.) HY-F wood ducks banded in Minnesota, 1996–2010.

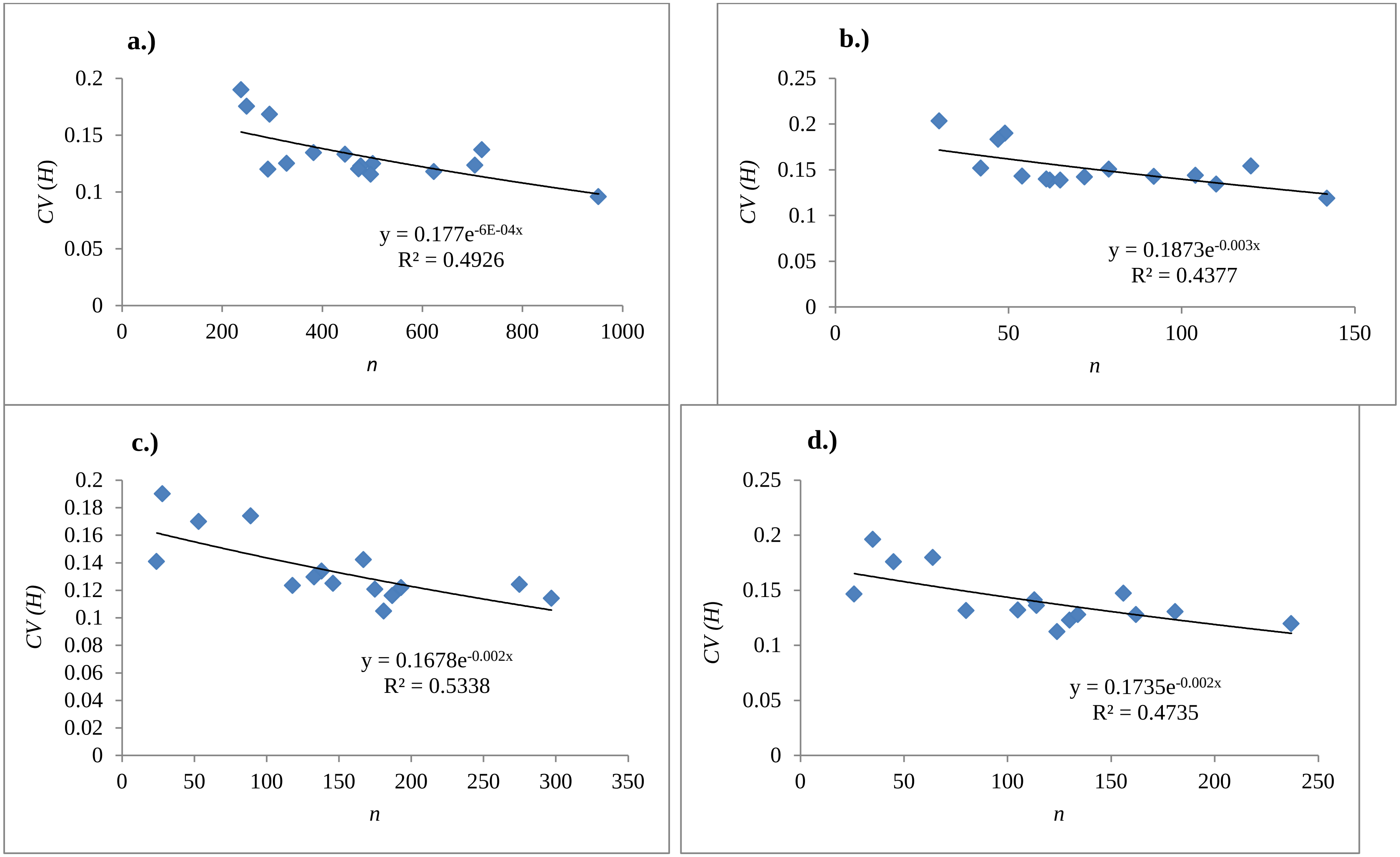

Figure 2a–d. The exponential relationship between the estimated coefficient of variation ($C\hat{V}[\hat{H}]$) and sample size (*n*) of a.) AHY-M, b.) AHY-F, c.) HY-M, and d.) HY-F wood ducks banded in Minnesota, 1996–2010.

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