Summaries of Wildlife Research Findings

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

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SUMMARIES OF WILDLIFE RESEARCH FINDINGS 2008

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ESTIMATING WHITE-TAILED DEER ABUNDANCE USING AERIAL QUADRAT SURVEYS

Brian S. Haroldson

SUMMARY OF FINDINGS

I estimated white-tailed deer (*Odocoileus virginianus)* abundance in select permit areas (PA) using quadrat surveys to recalibrate deer population models and evaluate the impact of deer season regulation changes on population size. With rare exception, precision of population estimates was similar among permit areas. However, because population estimates were not corrected for sightability, estimates represent minimum counts and are biased low. In 2009, I began testing a sightability estimator to adjust estimates for animals missed during surveys. Undocumented deer movement made analysis difficult. Additional trials are needed to develop sightability estimates.

INTRODUCTION

Management goals for animal populations are frequently expressed in terms of population size (Lancia et al. 1994). Accurate estimates of animal abundance allow for documentation of population trends, provide the basis for setting harvest quotas (Miller et al. 1997), and permit assessment of population and habitat management programs (Storm et al. 1992).

The Minnesota Department of Natural Resources (MNDNR) uses simulation modeling to estimate and track changes in deer abundance and, subsequently, to develop harvest recommendations to keep deer populations within goal levels. In general, model inputs include estimates of initial population size and spatial/temporal estimates of survival and reproduction for various age and sex cohorts. Because simulated population estimates are subject to drift as model input errors accumulate over time, it is imperative to periodically recalibrate the starting population within these models with independent deer population estimates (Grund and Woolf 2004).

Minnesota's deer numbers are managed according to numeric population goals within 125 PAs. MNDNR recently revised deer population goals within each PA using a consensus-based, roundtable approach consisting of 15-20 citizens representing varied interest groups (e.g. deer hunters, farmers, foresters, environmental groups, etc.; Stout et al. 1996). Revised goals are used to guide deer-harvest recommendations. Currently, deer populations exceed management goals in many PAs. A conventional approach of increasing the bag limit within the established hunting season framework has failed to reduce deer densities. As a result, MNDNR began testing the effectiveness of 3 non-traditional harvest regulations to increase the harvest of antlerless deer and reduce overall population levels (Grund et al. 2005). Accurate estimates of deer abundance are needed to evaluate these regulations.

My objective in this investigation is to provide independent estimates of deer abundance in select PAs that are within 20% of the true mean with 90% confidence (Lancia et al. 1994). Abundance data will be used to recalibrate population models to improve population management and to evaluate impacts of deer season regulation changes on deer abundance.

METHODS

I estimated deer populations in selected PAs using a quadrat-based, aerial survey design. Quadrat surveys have been used to estimate populations of caribou (*Rangifer tarandus*; Siniff and Skoog 1964), moose (*Alces alces*; Evans et al. 1966), and mule deer (*O. heimonus*; Bartmann et al. 1986) in a variety of habitat types. Quadrats were selected using 1 of 3 sampling designs: (1) stratified random (StRS; Cochran 1977); (2) 2-dimensional (2-D) systematic (Cressie 1993, D'Orazio 2003); or (3) generalized random-tessellation stratified (GRTS; Stevens and Olsen 2004). I used a StRS sampling design in PAs where the local wildlife manager had prior knowledge about deer abundance and distribution. Quadrats were stratified into 2 abundance classes (low, high) based on relative deer densities. Occasionally, additional strata were constructed to encompass management boundaries (e.g., park boundaries). I used a 2-D systematic sampling design in other areas. Systematic designs are typically easier to implement and maximize spatial distribution of the sample. Beginning in 2008, I used the GRTS design to obtain spatially balanced stratified and random samples. This design improves the spatial distribution of StRS and permits replacement of sample quadrats that are lost due to navigation hazard or high human development. Previously, replacement quadrats were unavailable in systematic PAs because of the rigid, 2-D design.

Within each PA, quadrats were delineated by Public Land Survey section boundaries and a 20% sample was selected for surveying. Sample size calculations indicated this sampling rate was needed to meet accuracy and precision objectives. I used OH-58 helicopters during most surveys and attempted to maintain a flight altitude of 60 m above ground level and an airspeed of 64-80 km/hr. A Cessna 182 airplane was used in 3 PAs dominated by intensive row-crop agriculture. To increase visibility, I completed surveys after leaf-drop and when snow cover measured at least 15 cm. A pilot and 2 observers searched for deer along transects spaced at 270-m intervals until they were confident all deer were observed. When animals fled the helicopter, direction of movement was noted to avoid double counting. I used a real-time, moving-map software program (DNR Survey; MNDNR 2005), coupled to a global positioning system receiver and a tablet-style computer, to guide transect navigation and record deer locations and aircraft flight paths directly to ArcView GIS (Environmental Systems Research Institute 1996) shapefiles. I estimated deer abundance from StRS surveys using PROC SURVEYMEANS (SAS 1999). I used the R programming language (RDCT 2009) and formulas developed by D'Orazio (2003) for 2-D systematic surveys and the R package SPSURVEY (ver. 2.0; RDCT 2009) for GRTS surveys. I evaluated precision using coefficient of variation (CV), defined as standard deviation of the population estimate divided by the population estimate, and relative error (RE), defined as the 90% confidence interval bound divided by the population estimate (Krebs 1999).

During 2009, I conducted a pilot study in 2 PAs (240, 345) to test the repeatability (temporal variation) of our survey protocol and to evaluate if the Gasaway et al. (1986) resurvey method could be used to adjust population estimates for sightability. Deteriorating snow conditions precluded data collection in additional PAs. After the initial survey was completed in each PA, I selected 10 quadrats where at least 20 deer had been previously observed. I then resurveyed each quadrat twice in succession before moving to subsequent quadrats. In the first replicate (temporal survey), I followed the same survey protocol as the initial survey. However, in the second replicate (sightability survey), flight speed was reduced to 48-64 km/hr to allow observers additional time to locate deer. Replicate surveys were flown 1-4 days and 11 days after completion of initial surveys in PA 345 and 240, respectively. The pilot and observers were the same for all surveys within each PA. I compared deer counts during initial and temporal surveys to evaluate repeatability. To evaluate sightability, I examined deer locations from survey-specific shapefiles and compared deer counts during temporal and sightability surveys. I defined sightability as number of deer observed / number of deer available. Animals observed departing a quadrat were excluded from analysis if they were only observed during 1 survey. Deer groups (*n*=1, 2, 3, etc.) observed during both surveys and located in close proximity were assumed to be replicate sightings. If group size varied between surveys, I used the higher value as the "available" group size in computations. When deer groups had no obvious match between surveys, I assigned each group to 2 classes: (1) minimum estimate - groups were treated as independent observations, available during both surveys, but not observed; and (2) maximum estimate - groups were pooled together and group size was summed by survey; the higher count value between surveys defined available group size and the lower value defined observed group size. Once all deer groups were classified, I calculated minimum and maximum sightability estimates by quadrat and mean sightability by PA.

RESULTS AND DISCUSSION

I completed 4-8 surveys each winter (December-March, 2005-2009; Table 1). Stratified fixed-wing surveys were conducted in PAs 270 and 272. Based on long-term deer harvest metrics, population estimates in these areas were biased low. Several possibilities may explain this result: (1) deer were clustered in unsampled quadrats; (2) deer were wintering outside PA boundaries; (3) sightability was biased using fixed-wing aircraft; and/or (4) kill locations from hunter-killed deer were reported incorrectly. Land cover in these PAs was dominated by intensive row-crop agriculture. After crops were harvested each fall, deer habitat was limited to riparian areas, wetlands, abandoned farm groves, and undisturbed grasslands, including those enrolled in state and federal conservation programs. Although recreational feeding of deer could influence distribution, wildlife managers believed it was not a common practice in these PAs. Thus, I had no evidence to support non-traditional deer distribution in these units. I also had no reason to believe hunter registration errors had greater bias in these units than in other PAs. Although it was possible that deer occupied unsampled quadrats by chance, the use of optimal allocation to increase sampling effort in high strata quadrats because of expected higher deer densities should minimize this possibility. Furthermore, we surveyed 100% of the high-strata quadrats in PA 421, resulting in no unsampled quadrats. Sightability bias, however, is greater in fixed-wing aircraft than helicopters (LeResche and Rausch 1974, Kufeld et al. 1980, Ludwig 1981) and likely explained much of the bias I observed in these PAs. Consequently, all surveys have subsequently been conducted using a helicopter.

With the exception of PAs 270, 272, and 201, precision (CV, RE) of the population estimates was similar among PAs (Table 1). High precision in PA 270 was, in part, an artifact of sample design. Based on optimal allocation formulas, we selected and surveyed all high strata quadrats. Thus, because no sampling occurred within the high stratum (100% surveyed), sampling variance was calculated only from low strata quadrats. We observed few deer in these low strata quadrats, which resulted in low sampling variance and high precision of the population estimate. It is unlikely that this design (i.e., sampling 100% of high strata quadrats) will be feasible in all areas, especially if deer are more uniformly distributed throughout the landscape.

In contrast, survey precision in PAs 272 and 201 was poor. We observed few deer during either survey (n=144 and 56, respectively) and nearly all observations occurred within 1 or 2 quadrats. As a result, associated confidence intervals exceeded 60% of the population estimate (Table 1). Kufeld et al. (1980) described similar challenges with precision due to nonuniformity of mule deer distribution within strata in Colorado.

The number of deer observed between initial and temporal surveys varied by 2% and 18% in PAs 240 and 345, respectively (Table 2). Although Potvin et al. (2002) suggested values <30% were not biologically significant, my assessment of repeatability was derived from small sample sizes (sample rate=1-3%) and needs further evaluation. Repeatability of surveys is important if deer distribution varies significantly during static weather conditions. Random movement of deer across quadrat boundaries likely has little impact on population estimates if the sampling design is robust and incorporates habitat stratification and randomization. However, in situations where significant non-random movements are likely to occur in response to changing conditions (e.g., weather events), replicate surveys should be considered.

I did not correct population estimates for sightability. Thus, estimates represent minimum counts and are biased low. However, a sightability survey protocol was examined in sample quadrats during 2009. Mean sightability ranged from 0.68 to 0.72 in PAs 345 and 240, respectively. Estimated sightability per quadrat was also similar between PAs (Table 2). Despite greater search intensity during sightability surveys, temporal survey counts were higher in 8 of 19 quadrats (Table 2). Sightability data were excluded from 1 quadrat because survey protocol was not maintained during inclement weather. This count discrepancy between surveys may be the result of undocumented deer movement across quadrat boundaries. Deer response to aerial surveys ranged from apparent disregard to escape (running away). Although observers were instructed to make mental note of directional movement of deer groups to minimize double counting, implementation of consecutive surveys will mandate documentation of movement because of observer memory-bias between surveys and the unknown fate of moving deer. When deer are sedentary during surveys, the resurvey method (Gasaway et al. 1986) shows promise. However, as deer movement increases, confidence in count data decreases.

Additional sightability trials are needed to improve our understanding of the applicability and limitations of the resurvey method for adjusting population estimates for sightability. Future analysis will also include *post-hoc* evaluation of habitat features present in quadrats containing deer. This will provide additional empirical data for use in quadrat stratification. In addition, the impact of winter feeding on deer distribution will be examined to determine if pre-survey stratification flights (Gasaway et al. 1986) are warranted.

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Table 1. Deer population and density estimates derived from aerial surveys in Minnesota, 2005-2009.

Sampling design	Year	Permit area	Population estimate N	Population estimate 90% CI	CV (%)	Relative error (%)[a]	Density estimate (deer/mi^2) Mean	Density estimate (deer/mi^2) 90% CI	Model estimate (deer/mi^2)
Systematic	2005	252	2,999	2,034 – 3,969	19.5	32.2	2.9	2.0 – 3.9	2
		257	2,575	1,851 – 3,299	16.9	28.1	6.2	4.4 – 7.9	7
	2006	204	3,432	2,464 – 4,401	17.0	28.2	4.6	3.3 – 5.9	5
		209	6,205	5,033 – 7,383	11.4	18.9	9.7	7.9 – 11.5	5
		210	3,976	3,150 – 4,803	12.5	20.8	6.3	5.0 – 7.6	7
		256	4,670	3,441 – 5,899	15.9	26.3	7.1	5.3 – 9.0	5
		236	6,774	5,406 – 8,140	12.1	20.2	16.8	13.4 – 20.2	37
	2007	225	5,341	4,038 – 6,645	14.7	24.4	8.0	6.0 – 9.9	24
		227	5,101	4,245 – 5,960	10.1	16.8	9.8	8.2 – 11.5	13
		346	7,896	5,736 – 10,062	16.4	27.4	22.7	16.5 – 29.0	31
	2008	266	3,853	2,733 – 4,977	17.5	29.1	6.2	4.4 – 8.0	n/a[b]
Stratified	2005	206	2,486	1,921 – 3,051	13.7	22.5	5.2	4.0 – 6.4	5
		270	631	599 – 663	3.0	5.0	0.8	0.8 – 0.9	5
		342	3,322	2,726 – 3,918	10.8	17.7	9.1	7.5 –10.7	10
	2006	201	274	100 – 449	37.6	61.9	1.6	0.6 – 2.7	6
		269	1,740	1,301 – 2,180	15.2	25.1	2.6	2.0 – 3.3	3
		272	472	179 – 764	37.4	61.5	0.9	0.3 – 1.4	5
	2007	343	6,982	5,957 – 8,006	8.9	14.6	10.1	8.6 – 11.6	29
		344	4,116	3,375 – 4,857	10.7	17.7	19.7	16.1 – 23.2	49
		347	5,482	4,472 – 6,492	11.1	18.2	12.6	10.3 – 14.9	13
		349	10,103	8,573 – 11,633	9.1	15.0	20.4	17.3 – 23.5	35
	2008	262	2,065	1,692 – 2,437	10.9	17.9	3.0	2.5 – 3.6	n/a[b]
		271	1,019	848 – 1,189	10.1	16.6	1.6	1.3 – 1.8	8
GRTS[c]	2008	265	4,575	3,766 – 5,384	10.7	17.7	9.2	7.6 – 10.9	n/a[b]
	2009	240	11,041	9,799 – 13,003	8.5	14.1	16.7	14.4 – 19.1	28
		261	1,721	1,450 – 1,992	9.6	15.7	2.2	1.8 – 2.5	4
		345	4,247	3,678 – 4,806	8.0	13.2	12.8	11.1 – 14.5	21
		348	5,717	4,953 – 6,480	8.1	13.4	17.8	15.4 – 20.1	13

[a]Relative precision of population estimate. Calculate as 90% CI bound/*N*.
[b]Permit area boundaries were recently modified. No model estimate is available.
[c]Generalized Random-Tessellation Stratified sample design.

Table 2. Deer observed and estimated sightability during multiple aerial surveys in Minnesota, 2009.

Permit area	Quadrat	Survey[a]			Minimum estimated value			Maximum estimated value		
		1	2	3	Observed	Available	Sightability	Observed	Available	Sightability
345	45	27	23	27	23	27	0.85	23	27	0.85
	48	42	54	72	49	77	0.64	49	77	0.64
	57	24	27	26	23	30	0.77	23	26	0.88
	16	20	20	4	7	14	0.50	7	12	0.58
	18	18	14	14	12	16	0.75	12	16	0.75
	19	66	36	41	27	48	0.56	30	46	0.65
	23	29	22	20	18	24	0.75	18	21	0.86
	22	19	24	29	21	32	0.66	23	30	0.77
	34	48	32	22	17	37	0.46	21	33	0.64
	65	38	18	—[b]	—[b]	—[b]	—[b]	—[b]	—[b]	—[b]
	Total	331	270	255	197	305	0.65	206	288	0.72
240	11	51	45	48	40	53	0.75	44	49	0.90
	42	33	60	56	52	65	0.80	52	61	0.85
	81	31	11	10	7	10	0.70	7	8	0.88
	18	45	33	49	33	49	0.67	33	49	0.67
	97	49	57	47	39	66	0.59	41	52	0.79
	128	22	27	33	19	33	0.58	19	33	0.58
	15	50	60	58	51	67	0.76	51	61	0.84
	32	26	26	28	17	35	0.49	21	27	0.78
	52	23	27	28	20	35	0.57	20	35	0.57
	106	54	45	57	42	60	0.70	45	57	0.79
	Total	384	391	414	320	473	0.68	333	432	0.77

[a]Survey 1 = initial survey; Survey 2 = temporal survey; Survey 3 = sightability survey.
[b]Data excluded because deteriorating weather compromised survey protocol.

ESTIMATING WHITE-TAILED DEER DENSITY USING TRAIL CAMERAS

Emily J. Dunbar and Marrett D. Grund

SUMMARY OF FINDINGS

White-tailed deer (*Odocoileus virginianus*) densities in the farmland zone of Minnesota are estimated using simulation modeling and aerial surveys. Simulation modeling is not well suited for modeling population dynamics in small areas, such as Itasca State Park (Permit Area 287). In 2005, Itasca State Park was chosen as a study area to test alternative deer hunting regulations. Deer density estimates were needed to evaluate the effect of antler-point restriction regulations (>3-points-on-a-side) on the deer population in the park. A trail camera study was initiated in 2006 to monitor the population. Forty-two cameras were systematically placed at a density of 1 camera/130 ha. The ratio of legal bucks to sub-legal bucks (fork and spike bucks) and buck:antlerless deer was calculated for 2, 3-week sampling periods before and after the hunting season. During 2006, cameras captured 12,486 images of deer, and in 2007 cameras captured 11,326 images of deer over the 6-week sampling period. The study was continued in 2008; data entry was not complete at the time this report was written.

INTRODUCTION

In 2005, Itasca State Park was chosen as a study area to test a 3-points-on-a-side antler-point restriction regulation for deer hunting. Deer density estimates were needed to evaluate the effect of the antler-point restriction on the density and demographics of the deer population. The primary management objective associated with the antler-point restriction was to reduce deer density by increasing the antlerless harvest via reductions in the antlered harvest (Grund et al. 2005).

Deer densities in Minnesota have traditionally been estimated using simulation modeling (Grund 2007, Lenarz 2007). Aerial surveys have been used in some farmland permit areas to provide an independent field estimate for correcting population models (Haroldson and Giudice 2006). However, due to errors caused by demographic stochasticity and seasonal movement patterns, simulation modeling is not recommended for small areas (Grund 2001). The small size of Itasca State Park (approximately 130 km^2) made population modeling impractical. Also, aerial surveys were not feasible due to dense coniferous cover that exists in parts of the park. While deer density estimates were not available for the park, the simulated deer density immediately north of the park was estimated at 65 deer/km^2 (25 deer/mi^2) in spring 2007 (Lenarz 2007).

Infrared-triggered cameras have been used to estimate deer populations in a variety of habitat types and study area sizes (Moore 1995, Jacobson et al. 1997, Koerth et al. 1997, Warlock et al. 1997, and Roberts et al. 2006). Jacobson et al. (1997) developed a camera technique to estimate deer density using known numbers of individually identifiable mature bucks and associated age and sex ratios from the deer herd. In Texas, Koerth et al. (1997) compared camera population estimates to helicopter counts and concluded that both techniques provided reliable deer density estimates.

In Fall 2005, a pilot study, initiated at Itasca State Park using infrared-triggered cameras, determined that: (1) more sampling effort was needed, (2) a systematic sampling design should be used, and (3) pre-baiting of sites was needed. In 2006, the study was adjusted to accommodate the pilot study findings. The study was continued in 2007 and in 2008; data entry from 2008 was not complete at the time this report was written.

OBJECTIVE

1. To determine the density and demographics of the deer herd to assess effects associated with the antler-point restriction regulation at Itasca State Park.

METHODS

The trail camera study was conducted at Itasca State Park, located in northwestern Minnesota in 2006-2008 from September to December. The park is approximately 130 km^2. The study area we used was approximately 6,400 ha located in the interior portion of the park in order to minimize effects that movement patterns would have on deer observations along the perimeter of the park. Following the protocol developed by Jacobson et al. (1997), 42 trail cameras were systematically placed at a density of 1 camera/130 ha throughout the study area using the Systematic Point Sample tool in ArcView 3.3. Minor adjustments were needed to avoid wetland areas (Figure 1).

Each site was located using a global positioning system unit and marked using flagging material. Cameras were in the field for 3 weeks before and after the regular firearms season for a total of 6 weeks each year. For the first sampling period (before the regular firearms deer hunting season), sites were baited with 23 kg (50 lbs) of shelled corn 3 weeks prior to placing the cameras in the field. An additional 11 kg (25 lbs) of corn was added to each site 1 week before camera sampling began. Sites were baited with 23 kg (50 lbs) of shelled corn 1 week before the second sampling period (after the regular firearms season).

A Bushnell TrailScout Pro 2.1 Mega Pixel (MP) or 3.0 MP trail camera was used at each site. Cameras were attached to a nearby tree at a height of 1.5 m. Each camera faced north and was 4-6 m from the established bait pile. Cameras were angled slightly downward to aim the infrared beam to a height approximately 1 m above the bait pile. In 2007, wooden boxes were constructed to house cameras to protect them from precipitation and damage by bears. Cameras were programmed to take pictures day and night with a 1-minute delay between pictures in 2006 and a 30-second delay between pictures in 2007. Batteries and memory cards were replaced on a weekly basis. Corn (11 kg) was added to the baited area on a weekly basis for both sampling periods in 2006, the first sampling period in 2007, and 23 kg was added to each site on a weekly basis for the second sampling period in 2007. These adjustments were made based on observed feeding patterns each season.

Each image was examined using Adobe Photoshop 3.0 or Microsoft Photo Editor, and only images of deer within the sampling time frame were used in the analysis. We classified each deer as legal buck (>3-points-to-a-side), sub-legal buck, or antlerless deer. Legal bucks were individually identified using number, size, and arrangement of points. We excluded images if we were unable to classify a deer to an appropriate category.

RESULTS AND DISCUSSION

In 2006, trail cameras captured 16,682 images during the 2, 3-week sampling periods. More images were captured during the postseason (9,346) than during the preseason period (7,336). Approximately 75% of the images contained a photo of a deer. Other species we observed included black bear (*Ursus americanus*), raccoon (*Procyon lotor*), bobcat (*Lynx rufus*), snowshoe hare (*Lepus americanus*), a variety of avian species, gray wolf (*Canis lupus*), mice (*Peromyscus* spp.), squirrels (*Sciurus* spp. and *Tamiasciurus hudsonicus*), chipmunks (*Tamias striatus*), fisher (*Martes pennanti*) and humans. Some images (16%) were eliminated due to: (1) images contained no visible animal, (2) the image was distorted, (3) the distance from the camera to the deer was too great, and (4) vegetation obstructed the view of the deer. Thus, 11,526 images containing 14,115 deer observations were useable for project purposes in 2006. During the preseason period in 2006, we observed 1,507 legal bucks, 811 sub-legal bucks, and

3,430 antlerless deer. During the postseason period in 2006, we observed 1,772 legal bucks, 1,519 sub-legal bucks, and 5,078 antlerless deer.

In 2007, trail cameras captured 21,486 images during the 2, 3-week sampling periods. Nearly equal numbers of images were captured during the postseason (10,754) and the preseason period (10,732). Approximately 53% of the images contained a photo of a deer. Species other than deer that were observed in 2007 included those seen in 2006 (except for fisher) and red fox (*Vulpes vulpes*) and porcupine (*Erethizon dorsatum*). Some images (27%) contained no visible animal, and distortion of the image, distance from the camera, or vegetation obstructing the view also caused some deer to be unidentifiable (6%). Thus, 11,326 images containing 13,380 deer observations were useable for project purposes in 2007. During the preseason period in 2007, we observed 440 legal bucks, 41 sub-legal bucks, and 3,294 antlerless deer. During the postseason period in 2007, we observed 2,218 legal bucks, 1,159 sub-legal bucks, and 6,228 antlerless deer.

Future work includes finishing data entry for 2008 and comparing population estimates for all three years (2006-2008) using Jacobson's (1997) mark-recapture technique and change-in-ratio formula (Paulik and Robson 1969).

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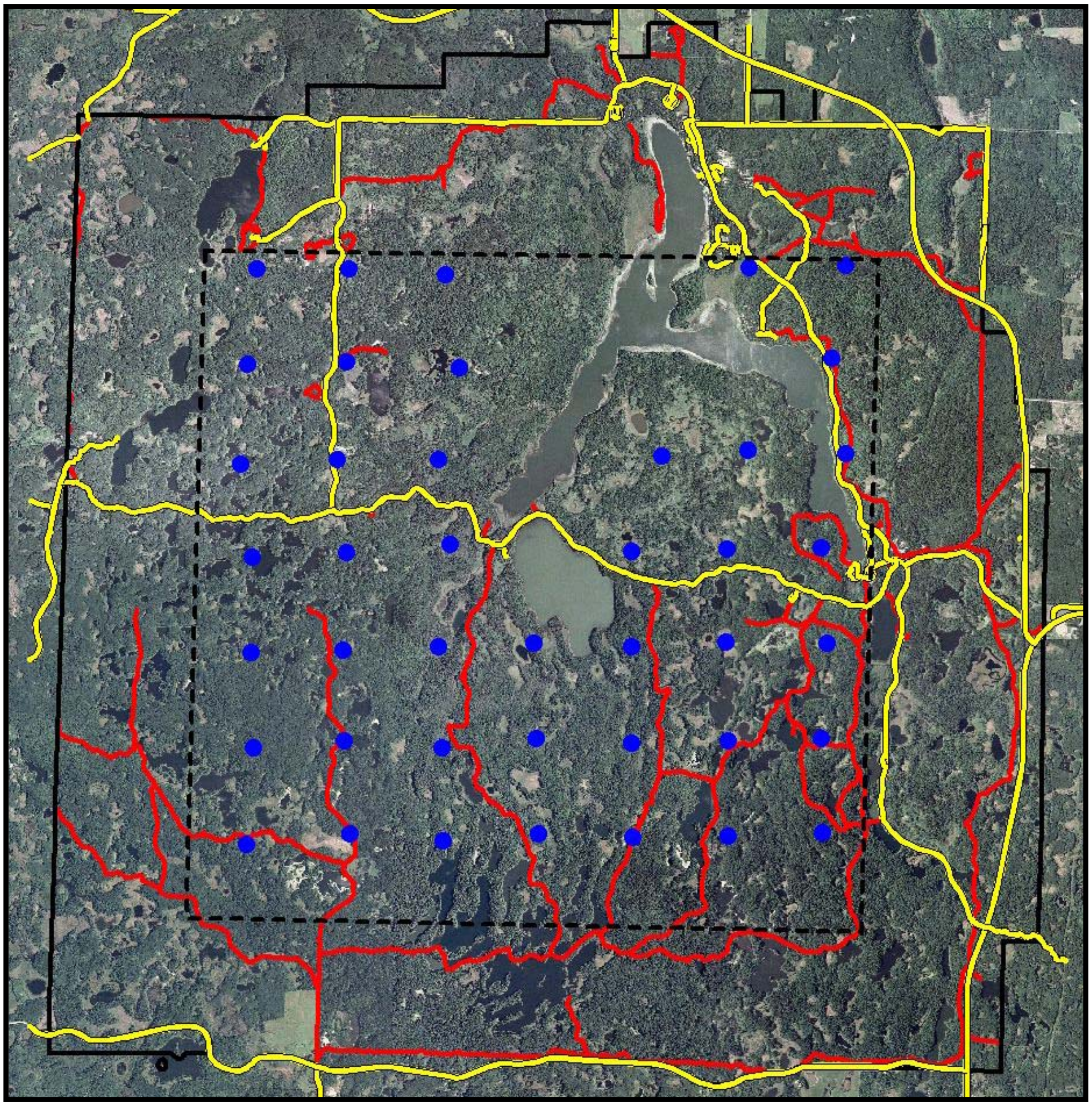

Figure 1. Locations of trail cameras (dots) in the study area (dashed line) at Itasca State Park, Minnesota in 2006-2008.

ECOLOGY, HUNTING SEASONS, AND MANAGEMENT OF GRAY AND FOX SQUIRRELS IN MINNESOTA

Emily J. Dunbar, Eric M. Walberg, and Richard O. Kimmel

SUMMARY OF FINDINGS

Southeast Asian Hmong hunters expressed concern about perceived low populations of squirrels (*Sciurus* spp.) on public hunting lands near population centers. The Section of Wildlife initiated a pilot study to identify potential problems for squirrel hunters and to determine if squirrel-hunting opportunities would benefit from harvest regulation changes or management activities. A survey of Minnesota squirrel hunters was conducted April-May 2009 to provide an understanding of how hunting opportunities could be improved and to determine if perception of squirrel hunting problems differed between hunter groups. A survey of state/provincial wildlife agencies was conducted April-May 2009 to gain knowledge about squirrel hunting seasons and management in other jurisdictions. Data collection is currently ongoing and results will be reported in a future report.

INTRODUCTION

Gray and fox squirrel (*Sciurus carolinensis* and *S. niger*) hunting provides recreational opportunities for an estimated 26,000 hunters in Minnesota (Dexter 2008). Hunters are not required to report harvest, but numbers of squirrels harvested are estimated through annual small game hunter surveys (Dexter 2008). Recently, southeast Asian Hmong hunters expressed concern about perceived low populations of squirrels on public hunting land near population centers. The Minnesota Department of Natural Resources (MNDNR) does not estimate squirrel populations, which are only managed through harvest regulations. MNDNR Section of Wildlife initiated a pilot study to determine if squirrel hunting opportunities would benefit from harvest regulation changes and/or management activities and which changes could potentially increase the huntable squirrel population on public land. We conducted a literature review of fox and gray squirrel ecology and management, a survey of squirrel hunters, and a survey of squirrel hunting season and management programs of state and provincial wildlife agencies. Data collection is currently ongoing and results will be provided in a future report.

OBJECTIVES

1. Conduct a literature review of squirrel ecology and management.
2. Survey squirrel hunters to provide an understanding of their experiences and thoughts on how squirrel hunting and management can be improved.
3. Survey other state and provincial wildlife agencies regarding squirrel hunting seasons and management programs.
4. Provide recommendations for squirrel hunting season management for Minnesota and potential research and/or management activities.

METHODS AND RESULTS

Manuscripts relating to the ecology, management, and hunting mortality of gray and fox squirrels are being compiled and selected manuscripts summarized.

A survey of Minnesota squirrel hunters was conducted April-May 2009 to provide an understanding of how hunting opportunities can be improved to determine if perceptions about squirrel populations and hunter satisfaction differ between hunter groups. We collected names and addresses of hunters that had indicated they had harvested squirrels on small game hunter surveys from 2005-2008. Additional names and addresses of assumed Hmong hunters were included from the 2008-2009 small game surveys. (Note – Hmong hunters had requested and provided funding for this project.) Surveys were mailed to 100 randomly selected assumed non-Hmong hunters from the 7-county metro region (Anoka, Carver, Dakota, Hennepin, Ramsey, Scott, and Washington), 100 randomly selected assumed Hmong hunters from the metro region, and 200 randomly selected assumed non-Hmong hunters from non-Metro regions of Minnesota. The survey instrument consisted of questions relating to harvest, counties and land ownership of properties hunted, hunter experiences, hunter access, hunter perception of squirrel populations, and suggestions for improving squirrel hunting experiences (Appendix 1). Surveys were mailed on 10 April 2009 with a second mailing to non-respondents on 1 May 2009. At this writing, 300 completed surveys have been returned from the first 2 mailings for an initial response rate of 79%.

A survey of state/provincial wildlife agencies was conducted April-May 2009 to gain knowledge about squirrel hunting regulations and management. An email survey was sent to 47 state wildlife agencies (Alaska, Hawaii, and Minnesota were excluded) and 4 Canadian provincial wildlife agencies (Quebec, Ontario, Manitoba, and Saskatchewan). The survey instrument consisted of questions relating to season opening/closing dates, bag and possession limits, management, research, population estimation activities, and issues concerning squirrel hunting (Appendix 2). Surveys were emailed on 27 April 2009 with a second emailing to non-respondents on 15 May 2009. At this writing, 41 completed surveys have been returned for an initial response rate of 80%.

ACKNOWLEDGMENTS

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Appendix 1.

Minnesota Fox and Gray Squirrel Hunter Survey

You have been selected from a group of hunters that harvested squirrels as indicated on past Small Game Hunter surveys. Because this survey is only being sent to a small number of hunters, your input is extremely valuable, please complete and return the following survey as soon as possible. Your identity will be kept confidential.

1. Did you hunt fox and/or gray squirrels last season (September 2008-February 2009)?

   Yes ___ No* ___

   *If No, then you do not need to continue the survey; please answer question 1 and return the survey.

2. In which county/counties did you hunt squirrels? ______________________________

3. Approximately, how many squirrels did you harvest last season?

   1-5 ____ 6-10 ____ 11-15 ____ 16-20 ____ 21-25 ____ 26-30 ____ 31+_____

4. Approximately, how many days did you spend hunting squirrels last season?

   1-7 days ____ 8-14 days ____ 15-21 days _____ 22-28 days ____ 29+ days ____

5. How long have you hunted squirrels in Minnesota?

   1-5 years ____ 6-10 years ____ 11-15 years _____ 16-20 years ____ 21+ years ____

6. Do you hunt squirrels on public land ___, private land ___, or both ___?

7. In the past 5 years, have you hunted squirrels:

   on the same properties ___, on new properties ___, or both ___?

(Survey continues on next side)

8. What are the main obstacles for gaining access to property for squirrel hunting? (check all that apply)

____ I have not encountered obstacles

____ Land is posted as no hunting or trespassing

____ Not sure how to find additional public hunting lands

____ Not comfortable asking for permission to access private land from landowner

____ Denied access by landowner(s) in the past

____ Difficulty finding owners of private land

____ Other (please specify) ____

9. Over the past 5 years, do you think squirrel populations in areas where you hunt are: decreasing____, about the same____, or increasing_____?

10. Based on your perception of squirrel population trends, would you recommend changes to
(check all that apply):
____ Hunting regulations
____ Habitat management
____ Enforcement of regulations
____ Other (Please Specify) ____
____ I don't recommend any changes

11. How could the DNR improve your squirrel hunting experience?

____

____

____

Thank you for completing the survey! Please return the survey in the enclosed, postage-paid envelope.

Appendix 2.

Wildlife Agency Squirrel-Hunting Survey

1. Your name________________________
   Your position title __________________________
   Your email address_________________________________
   Your phone number _________________________________

2. Does your state/province have a fox and/or gray squirrel hunting season?
   Yes ____ No____
   If No, then the survey is complete. Please send this survey back.

3. When does the hunting season open?

4. When does the season close?

5. What is the daily bag limit and possession limit?

6. What are the shooting/hunting hours?

7. Does your state/province manage specifically for fox and/or gray squirrels ?
   Yes ____ No____
   If Yes, what are the management activities?

8. Does your state/province estimate fox and/or gray squirrel populations?
   Yes ____ No____
   If Yes, what techniques is used?

9. Is there currently any research being conducted by your agency on fox and/or gray squirrels?
   Yes _____ No______
   If so, please describe the study/studies:

10. Are there any issues surrounding squirrel populations or squirrel hunting in your state/province?

11. Other things we should know about your squirrel season:

CONDITIONAL PROBABILITY OF DETECTION OF RING-NECKED PHEASANTS IN CROWING MALE SURVEYS

Alison L. Harwood[1], Brock R. McMillan[1], Kurt J. Haroldson, and John H. Giudice.

SUMMARY OF FINDINGS

Because population-estimation methods are not practical for ring-necked pheasants (*Phasianus colchicus*), managers typically use indices of population size. A common criticism of indices is that constant probability of detection is assumed but not estimated. We used an auditory, mark-recapture (MR) method to estimate the conditional probability of detection ($\hat{\rho}$) in replicate crowing surveys of male pheasants on 18, 23-km² sites in southern Minnesota. Probability of detection varied by study site (range $\hat{\bar{\rho}}$ = 0.382 – 0.731) and was negatively associated with disturbance (including initial observer disturbance) and positively associated with crowing frequency and intensity. We also fit a mixture model to the data, which suggested there may be 2 groups of male pheasants with different detection probabilities (e.g., possibly due to different vocalization rates). Mark-recapture methodology may only be practical for intensive studies on relatively small areas, but crowing male indices adjusted for probability of detection could be used as a standard from which to evaluate the more common roadside index.

INTRODUCTION

Ring-necked pheasants are a species for which formal population-estimation methods are not practical. Pheasants are secretive and mobile (i.e., they tend to hide or move away from observers), and difficult to capture (Thomas 1996, Giudice and Ratti 2001, Lancia et al. 2005). Pheasants also do not have the flocking or territorial habits that increase the availability of some species for detection (Brown 1947, Eberhardt and Simmons 1987, Gibbs et al. 1998, Lancia et al. 2005). Because of these difficulties in detecting pheasants, populations are commonly monitored using indices based on crowing calls of males (Brown 1947, Kimball 1949, Kozicky 1952, Rice 2003).

In a crow-count survey, an observer stops at predetermined stations along a route and records the number of pheasant crowing calls heard during a specified time period (usually 2 minutes). The standard index is crows/stop, which is assumed (but not known) to be proportional to the abundance of male pheasants (Kimball 1949, Kozicky 1952, Luukkonen et al. 1997, Rice 2003). The validity of crow-count surveys as an index of male pheasant abundance requires that probabilities of detection are similar among survey periods and locations (i.e., the proportion of males detected is constant across time and space). Fisher et al. (1947), Luukkonen et al. (1997), and Anderson (2001, 2003) have criticized use of such indices because they are highly variable and probabilities of detection are generally unknown.

A variety of factors may affect the probability of detecting crowing male pheasants. Some factors (e.g., calendar period, time of day, weather, observer skill) can be controlled through survey design, but others cannot. Observer presence may cause nearby birds to stop calling at least temporarily (Johnson 2008). Background noise may mask the sound of calling birds (Simons et al. 2007). In addition, the distribution of pheasants in relation to the survey route may affect their availability for detection (Fisher et al. 1947, Hutto and Young 2003). Variability may be controlled by conducting replicated counts (Kimball 1949, Kozicky 1952, Gibbs et al. 1998, Johnson 2008), but the assumption that the expected probability of detection E($\hat{\rho}$) is similar among comparison groups or years remains untested.

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Alldredge et al. (2007) used a time-of-detection method to estimate probability of detection in songbirds. In this method, information about whether a bird is detected during separate time intervals is used to create a detection history. The detection history can be treated like a MR history in a closed population model (Alldredge et al. 2007, Riddle et al. 2008). In this study, we applied time-of-detection methods to replicate surveys of crowing male pheasants from 18 study sites in southern Minnesota to estimate mean detection probability and evaluate the assumption that $E(\hat{\rho})$ was similar among study sites. Our objectives were to: (1) evaluate use of an auditory MR technique to estimate detection probability of crowing male pheasants; and (2) evaluate factors that may influence probability of detection and overall accuracy of crowing male indices.

STUDY AREA

We conducted this study on 18 sites in southern Minnesota as part of a concurrent study on pheasant habitat needs (Haroldson et al. 2007). Nine study sites were selected in each of 2 regions located near Faribault and Windom, Minnesota. Study sites averaged 23 km^2 (9 $miles^2$) in size. The primary land use in all study sites was agriculture, but sites varied in the amount and distribution of grassland habitat, pheasant winter habitat (e.g., emergent wetlands, shrub swamps, and planted shelterbelts), roads, and relative pheasant density (Haroldson et al. 2007). The percent of grassland habitat within the study sites varied from 3.4% to 31.6%, with most grassland habitat idled under the Conservation Reserve Program.

At each study site, we attempted to establish 9 listening stations that were evenly distributed across the site, based on an estimated 0.8 km (0.5 mile) auditory radius, to achieve maximum possible coverage of the study site and minimum overlap among stations (Figure 1). Where possible, we located stations on roads to facilitate convenient access. Where roads were not available, we located stations up to 0.4 km (0.25 mile) from roads. Due to road coverage and landscape obstacles (e.g., lakes), 2 study sites had only 8 stations.

METHODS

Crowing Male Surveys

We conducted 10 replicate surveys at each study site between 20 April and 31 May 2007 during mornings that met standardized weather conditions (wind speeds <16 km/hour and no precipitation at the beginning of the survey). Two trained observers performed surveys on each study site, dividing the 8-9 stations between them (4-5 stations/observer). The starting location for each survey route was selected randomly, and direction of travel was selected to minimize travel time and observer overlap. Observers rotated systematically among 2-4 study sites throughout the survey season.

Crowing male surveys began 45 minutes before sunrise and were completed by sunrise. Observers recorded wind speed, temperature, and amount of dew at the beginning and end of each survey and percent of sky covered by clouds at the end of the survey. Upon arrival at a listening station, observers immediately began to count the number of crowing males and the number of times each male crowed for 2 minutes. Observers attempted to identify each individual crowing male by marking their relative distance and direction from the observer on a map of the listening station. Observers documented potential misidentifications of individual crowing males by classifying the encounter history for each presumed individual as certain or uncertain. Observers also classified disturbance affecting their ability to hear crowing pheasants into 4 categories: none, low (e.g., distant tractor noise), medium (e.g., intermittent traffic), or high (e.g., constant background noise).

Mark-Recapture

We used extended listening intervals at 4 stations/study site to evaluate whether a closed population capture-recapture approach (Huggins 1989) could be used to estimate the mean detection probability of crowing pheasants. We randomly selected the first MR station on each study site, and then systematically selected the remaining 3 MR stations to ensure an even distribution across the study site, facilitate the assignment of 2 MR stations/observer, and ensure the completion of the survey within the 45-minute survey period. The same MR stations were sampled in each survey. Observers at MR stations continued to survey for 2 additional 2-minute intervals immediately following the first listening period. The second and third listening periods identified which birds heard during the first period were heard again, and also birds that had not previously been detected.

Analysis

We used Huggins (1989) closed-capture models to estimate conditional detection probability (ρ) and to evaluate a small set of covariates that could plausibly influence heterogeneity in detectability among study sites, listening stations, and survey days. The beta vector (β) for the most complex model included an intercept, constrained time variation (t = 1 versus t = 2, 3), and covariate effects. The constrained time variation evaluated the potential change in probabilities of detection between the first 2-minute listening interval and the second and third 2-minute listening intervals. The initial group covariates considered were study site (*SITE*) and road type (*RTYPE*) nearest each listening station. Study site was selected as a possible cause of variation because the amount and type of habitat within each site could possibly affect the distribution of pheasants, which in turn could affect the availability of male pheasants for detection. Road type (paved versus gravel) was selected as a surrogate for traffic volume, reasoning that traffic noise may negatively affect perceptibility of crowing calls. Individual covariates included relative disturbance (*DISTURB* [1-3 = none, low, moderate]) and total crows heard in the first listening interval (*CROWS*). Disturbance reflected the ability of observers to hear crowing pheasants over background noise. High disturbance sampling events were excluded from the MR analysis because they would typically be excluded in operational surveys. The variable *CROWS* was viewed as an index of crowing frequency and intensity and, indirectly, local density of male pheasants (at the listening station scale). We postulated that availability and perceptibility (Marsh and Sinclair 1989, Johnson 2008) would be positively correlated with crowing intensity (proportion of roosters that crow) and frequency, respectively, which may inflate the crowing male index if crowing incites other birds to crow more frequently (Gates 1966). We also evaluated a quadratic term for *CROWS* (which modeled a plausible non-linear relationship between detectability and *CROWS*) and an interaction term (*CROWS*×*SITE*) to permit the estimated slope to vary among sites.

Our primary research question was whether $E(\hat{\rho})$ varied substantially among study sites. Consequently, *SITE* was a reasonable covariate to consider in the initial model set. However, we were also interested in understanding what explanatory factors may be correlated with *SITE* effects. Thus, we also evaluated models where *SITE* was replaced with 2 explanatory variables: (1) percent of each study area composed of grassland habitats (*GRASS*); and (2) observer group (*OBSG* = 6 clusters of observers that reflected the restricted assignment of observers to study sites). We postulated that spring pheasant abundance (males and females) on each study site would be positively correlated with *GRASS* (Haroldson et al. 2006), and that bird density and distribution (at the study-site scale) may in turn influence $E(\hat{\rho})$. Finally, we constructed a model that included weather and date effects to evaluate our assumption that survey protocols minimized heterogeneity due to these covariates.

We used the R programming language (R Development Core Team 2008) and the RMark package (Lake and Rexstad 2008) to construct MR models, and the optimization routine in program MARK (White and Burnham 1999) to obtain parameter estimates ($\hat{\beta}$ and $\hat{\Sigma}$). We compared models using Akaike's Information Criterion (AIC_c) and estimated model weights (Burnham and Anderson 2002). Unfortunately, there is no unique way to compute goodness-of-fit for closed-population models with individual covariates (Cooch and White 2008). Therefore, we used estimability and precision of parameter estimates and consistency of covariate effects (observed versus predicted direction) to evaluate model fit and adequacy. We also developed a jackknife assessment of predictability by removing observations from one sampling event (site-station-date), refitting the best-approximating model to the reduced data set, and predicting the conditional probability of detection in the first listening interval (t1) for the held-out event. We repeated this process for each sampling event and then compared observed (proportion of birds in group *i* detected in t1) and predicted detectability. Based on the jackknife comparisons, we constructed an additional post-hoc model that assumed the bimodal distribution in observed detectability was the result of one group or mixture of birds being more detectable than a second group (Pledger 2000, Cooch and White 2008). More specifically, we constructed an additive model where the probability of belonging to a mixture (π) was constant across time and space, which resulted in 2 additional link-function parameters compared to the best-approximating *a priori* model where π was fixed (= 1). We used 2 mixtures, defined the probability of detection in listening period 1 as $\hat{p}_{.1} = \hat{\pi}_1 \hat{p}_{11} + (1-\hat{\pi}_1)\hat{p}_{21}$, and computed $\hat{\mathrm{var}}(\hat{p}_{.1})$ using the delta method (Seber 2002). We also considered more complex mixture models (e.g., where π was modeled as function of covariates), but these models failed to converge, had non-positive-definite variance-covariance matrices, or generated imprecise parameter estimates. Thus, we excluded these models from further consideration.

RESULTS

Although weather varied throughout the survey season, 92% of crowing male surveys were conducted on mornings meeting standardized conditions (i.e., wind <16 km/hour and no precipitation). Fifty percent of surveys were conducted on mornings with ≤ 20% cloud cover, whereas 5% of surveys were conducted under complete cloud cover.

Observers completed 177 crowing male surveys which acquired data at 1586 of 1600 potential sampling events (site-station-date). Pheasants were heard crowing on all 18 study sites. Crowing male indices ranged from 1.2 to 6.4 males/station (Table 1). Mean number of males detected/station declined from 4.5 to 3.0 when disturbance increased from none to high. Likewise, variation in crowing frequency was higher during high-disturbance sampling events (SD = 1.40) than those with moderate to no disturbance (SD ≤ 0.99), although only 4% of sampling events were classified as high disturbance.

The MR dataset consisted of 647 sampling events (site-station-date) and 3,849 encounter histories. The number of males, crowing calls, and calls/male detected in the first listening period varied more among study sites than between MR and non-MR stations within study sites (Figure 2). Total number of crowing male pheasants detected over the 3 listening periods ranged from 0-26 (median = 7) per sampling event. Eighty percent of encounter histories were classified as certain, 17% as uncertain, and 3% were missing quality-control information. The proportion of encounter histories classified as uncertain increased slightly with an increase in total birds detected (Cochran-Armitage Trend Test, Z = -10.087, P < 0.001, Figure 3) especially at densities >10 birds. However, 75% of encounter histories were from site-station-dates with ≤10 total male pheasants detected.

We initially considered a set of 14 MR models to estimate probability of detection of crowing male pheasants. Of these 14 models, only model 13 {π(1) *p* (*TIME* + *SITE* + *CROWS* + *DISTURB* + *CROWS*×*SITE*)} was supported by the data (w_i = 1.0). Post-hoc models where *SITE* was replaced with *GRASS* and *OBSG* were not supported by the data. Model 13 suggested that conditional probability of detection varied among sites, was negatively associated with *DISTURB*, and positively associated with *TIME* (first versus second and third listening periods) and the number of crows (*CROWS*) heard per site (but the effect of *CROWS* varied by site). The amount of variation within sites was generally low with the exception of the site with the lowest pheasant counts (Figure 4, study site 36). The jackknife comparison of observed versus predicted detectability indicated that model 13 fit the data reasonably well (Spearman's *rho* = 0.5; Figure 5). However, detectability approached unity for some sampling events regardless of covariate values, which suggested the presence of some unexplained heterogeneity in ρ. We attempted to explain additional heterogeneity by fitting a finite mixture model (model 17). The mixture model improved model fit compared to model 13 (ΔAIC_c = 210.3) and was clearly the best-approximating model in the final model set (w_i = 1.0). Based on model 17, the estimated proportion of individuals belonging to mixture 1 (group with lower detectability) was 40% (SE = 2.4) and the odds of being detected were 21 times greater for individuals in mixture 2 (Table 2). The signs of the relationships between conditional probability of detection and covariates were consistent with model 13 (Table 2) and relative differences in E($\hat{\rho}$) among study sites were similar between the 2 models (Figure 4). On the other hand, site-specific estimates of E($\hat{\rho}$) were less and estimated variances (of $\hat{\rho}$) were greater in model 17 (Figure 4).

DISCUSSION

Rates of detection are influenced by both the availability of animals to be detected and the perceptibility of observers in detecting animals (Marsh and Sinclair 1989, Johnson 2008). In crowing male surveys, a pheasant becomes available for detection when it crows. In order to be detected, however, the observer must perceive and correctly identify the crowing call. This study considered only detectable animals (i.e., male pheasants that emitted at least 1 crowing call during the survey period). Thus, variability in $\hat{\rho}$ as reported here pertains to perceptibility, which is primarily a function of observer skill and survey conditions (Johnson 2008).

We considered several factors that may influence perceptibility of crowing pheasants and lead to heterogeneous detection probabilities. For example, our survey protocol constrained surveys to mornings with weather conditions favorable to pheasant crowing activity and perceptibility. Thus, not surprisingly, wind was not identified during model selection as an important covariate explaining probability of detection. Similarly, training was provided to observers to correctly identify male pheasant crowing calls and to distinguish individual males. Although uncertainty in distinguishing individual males increased with total number of males detected (especially in the minority of encounter histories when >10 males were detected), observer group was not identified during model selection as an important covariate explaining detection probability. These examples provide indirect evidence that our survey protocols reduced effects of weather and observer skill on perceptibility of crowing pheasants.

Increasing disturbance (e.g., background noise) was associated with reduced probability of detection. Road type may contribute to disturbance (e.g., greater traffic noise on paved roads); however, models containing disturbance were better supported by the data. The negative effect of disturbance may be minimized by selecting listening stations where high disturbance is unlikely. Although some types of disturbance (e.g., birds calling near a wetland) may be difficult to avoid, it may be possible to locate survey routes away from roads that receive a high amount of traffic.

We found that detection probabilities were lower during the first listening period than during the second and third listening periods, which likely reflected temporary disturbance caused by the presence of the observer. This result may indicate the need for a short, 2-3 minute waiting period prior to beginning the survey to allow pheasants to acclimate to the presence of an observer and resume crowing. In addition, these data suggest that a single 2-minute listening period was sufficient to detect an average of only 60% of available males. Extending the listening period to 6 minutes would increase probability of detection, but also reduces the number of stations that could be surveyed.

Some behavioral traits may affect availability of pheasants for detection. Rosenstock et al. (2002) suggested that the presence of non-territorial males may affect calling frequency of some birds. Leonard and Horn (1995) found that dominant domestic roosters (*Gallus gallus domesticus*) crowed more frequently than subordinate roosters. Non-territorial or subordinate males may not crow as often as other males, or may not crow at all, making them unavailable for detection. In this study, model 17 provided evidence of 2 groups of males with dramatically different probabilities of detection, which may reflect different crowing behavior between dominant and subordinate males.

Pheasant density may also affect crowing frequency, which in turn affects probability of detection. Gates (1966) reported significantly lower crowing frequency of individual males at low pheasant densities, presumably because of reduced need to defend territories from competing males. Therefore, a larger proportion of males may be unavailable for detection and crowing male indices may underestimate relative abundance at low pheasant densities. This result may explain the low $E(\hat{\rho})$ and high variance for study site 36, which had the lowest pheasant count.

Rosenstock et al. (2002) indicated that for indices to be reliable, rates of detection must remain constant among sites. We found that $E(\hat{\rho})$ varied among sites, which suggests that reliability of crowing male indices could be improved by adjusting counts for $E(\hat{\rho})$. However, Johnson (2008) cautioned that the cost to adjust counts may outweigh the benefits. We suggest that MR methodology may only be practical for intensive studies on relatively small areas with moderate to high densities of males. For operational surveys, managers may increase probability of detection by locating listening stations away from areas of high disturbance and allowing pheasants to acclimate to observer presence before counting crowing calls. For intensive studies, crowing male indices adjusted for probability of detection could be used as a standard from which to evaluate the more common pheasant population index derived from roadside surveys.

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Table 1. Pheasant population indices from replicate (*n*) crowing surveys on 18 study sites in southern Minnesota during spring 2007.

				Males/station	
Region	Study site	Stations	*n*	Mean	SE
Windom	19	9	10	4.6	0.28
	20	9	10	6.4	0.37
	21	8	10	3.5	0.22
	22	9	10	5.5	0.26
	23	9	9	5.0	0.29
	24	9	9	4.7	0.22
	25	9	10	3.9	0.23
	26	9	10	5.5	0.27
	27	9	9	2.7	0.17
Faribault	28	8	10	3.0	0.21
	29	9	10	3.9	0.25
	30	9	10	2.7	0.15
	31	9	10	4.2	0.22
	32	9	10	3.1	0.18
	33	9	10	3.7	0.21
	34	9	10	3.7	0.17
	35	9	10	3.5	0.24
	36	9	10	1.2	0.10

Table 2. Link-function parameter estimates for study site 19 (reference group) and 33 based on Model 17.

Parameter	MLE	95% CI	Odds
π (mixture1)	-0.4080	-0.6056 – -0.2104	-----
p: intercept (site19)	-2.793	-3.568 – -2.017	-----
p: mixture2	3.041	2.733 – 3.349	20.92
p: time (t2,3)	0.3143	0.2272 – 0.4013	1.369
p: site33	-1.116	-1.938 – -0.2930	0.3277
p: crows	0.1585	0.0935 – 0.2236	1.172
p: site33*crows	0.2492	0.1275 – 0.3709	1.283
p: disturb	-0.1298	-0.2450 – -0.0147	0.8782

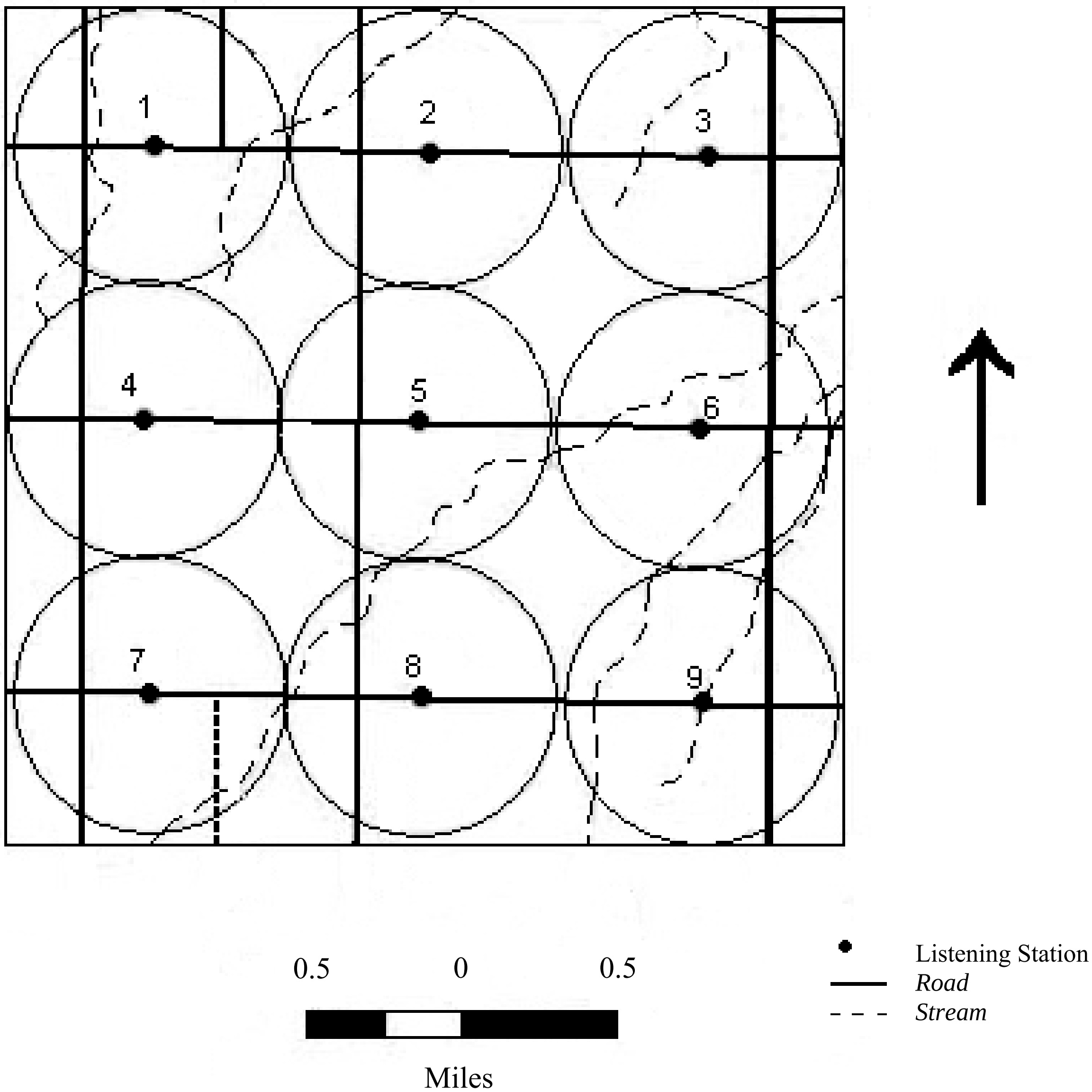

Figure 1. Typical study site showing 9 crow-count listening stations and estimated 0.8 km (0.5 mile) auditory radii, Minnesota, spring 2007.

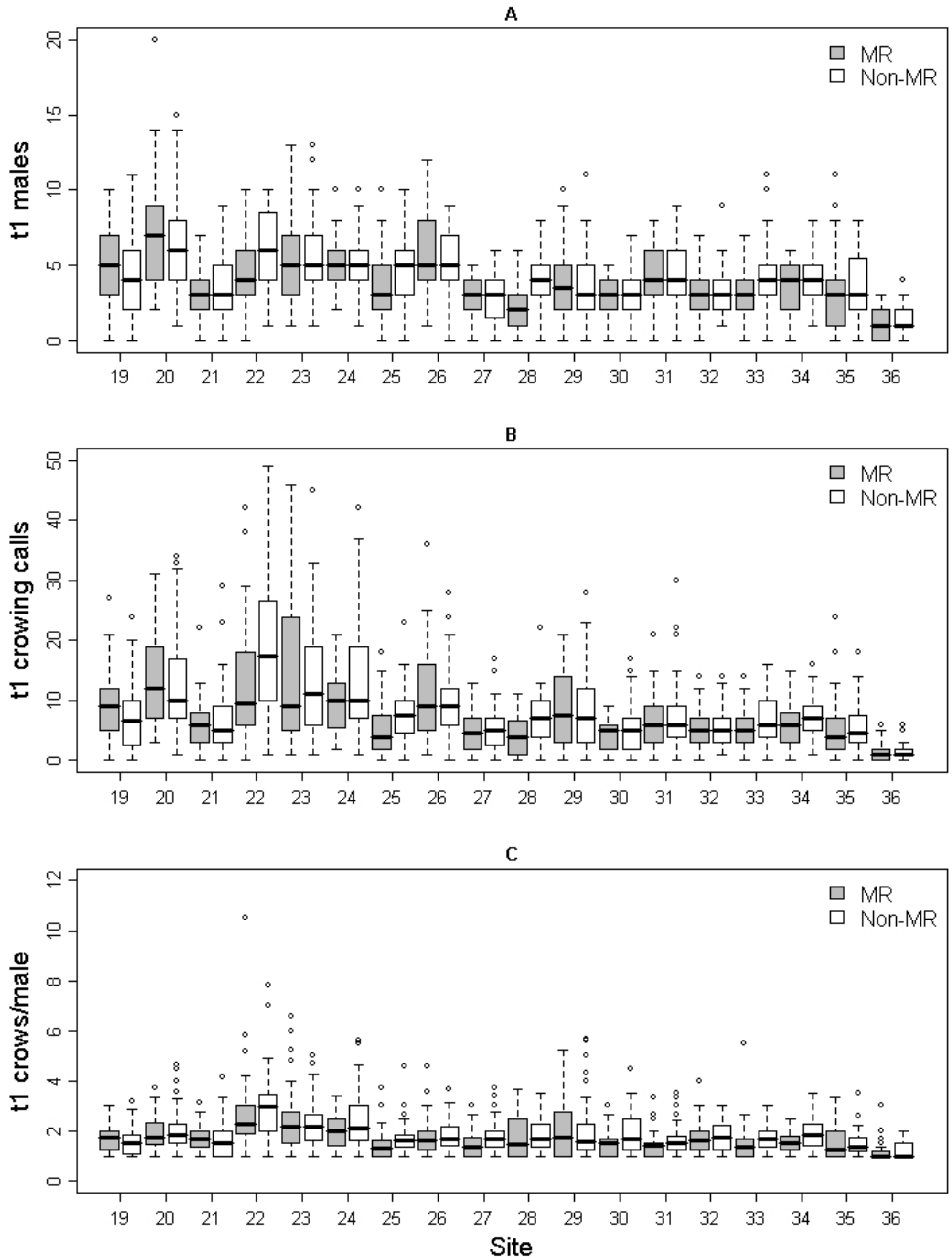

Figure 2. Number of male pheasants (A), crowing calls (B), and crows/male (C) detected in the first listening interval by study site and station type (mark-recapture [MR] versus non-MR). Boxplots do not include 65 sampling events (40 MR, 25 non-MR) with high disturbance.

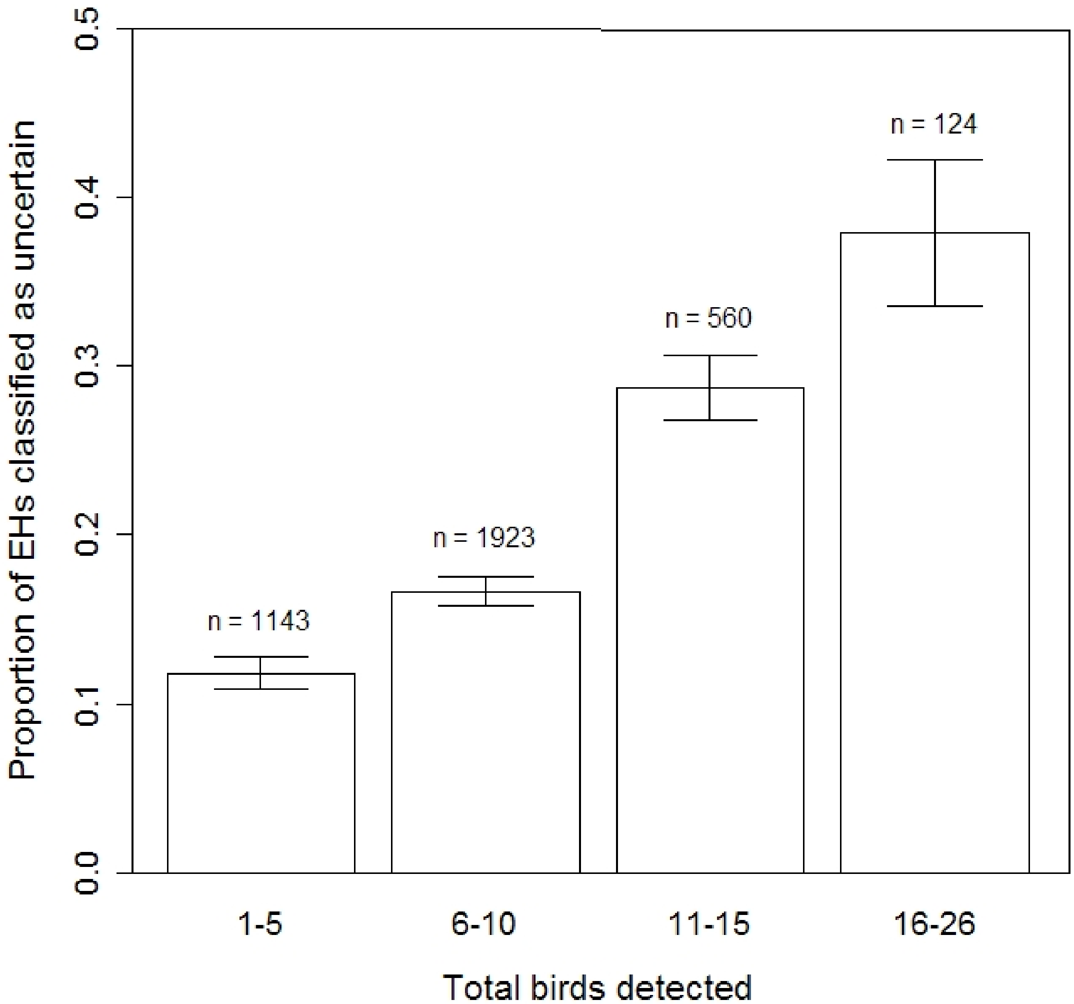

Figure 3. Proportion of encounter histories (EHs) classified as 'uncertain' as a function of total male pheasants detected at each site-station-date.

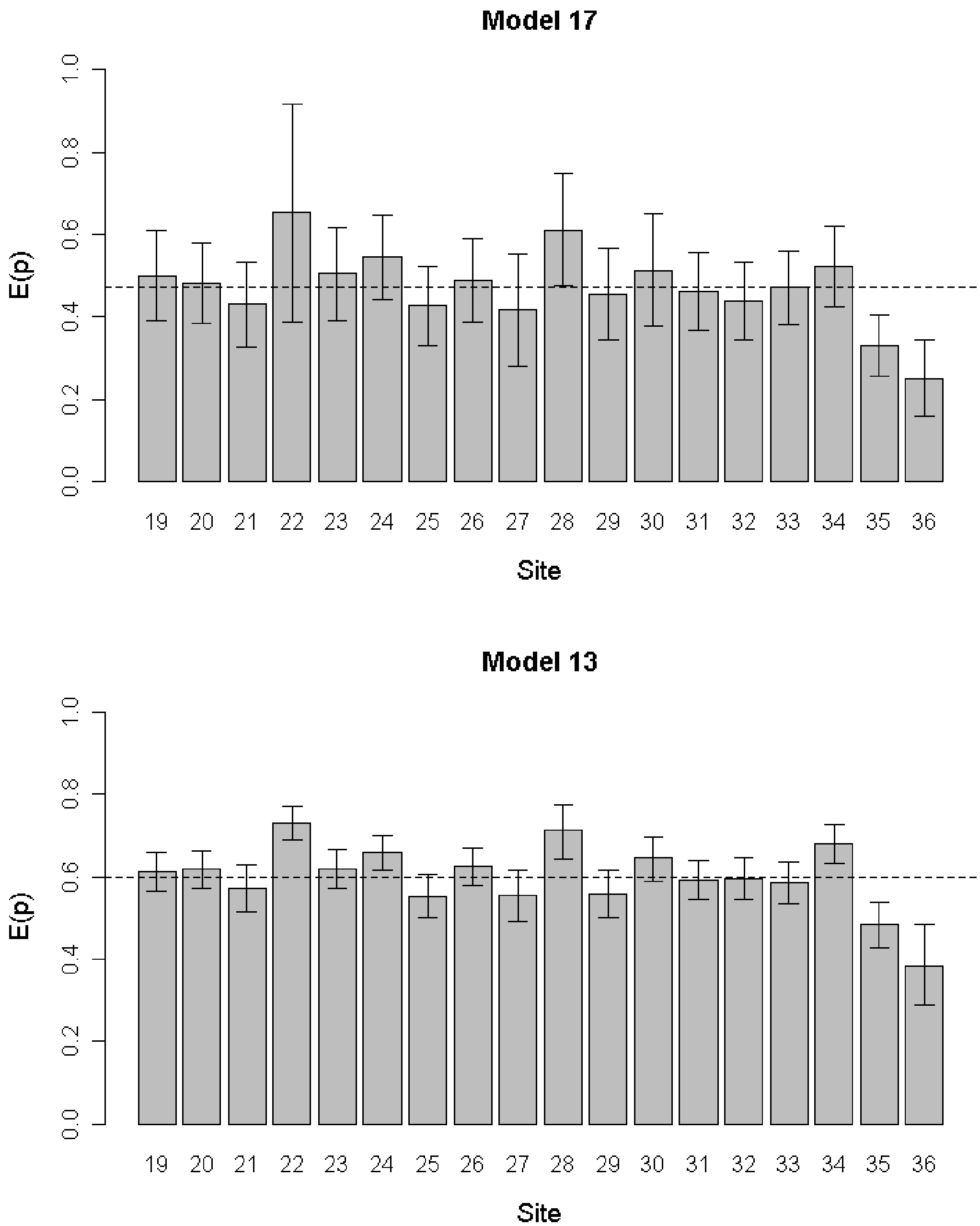

Figure 4. Expected probability of detection $E(\hat{\rho})$ in the first 2-minute listening interval based on model 13 and model 17 (mixture model) given median covariate values (for *DISTURB* and *CROWS*) observed at each site. Dashed lines are the overall mean probability of detection, and error bars denote asymptotic 95% confidence intervals.

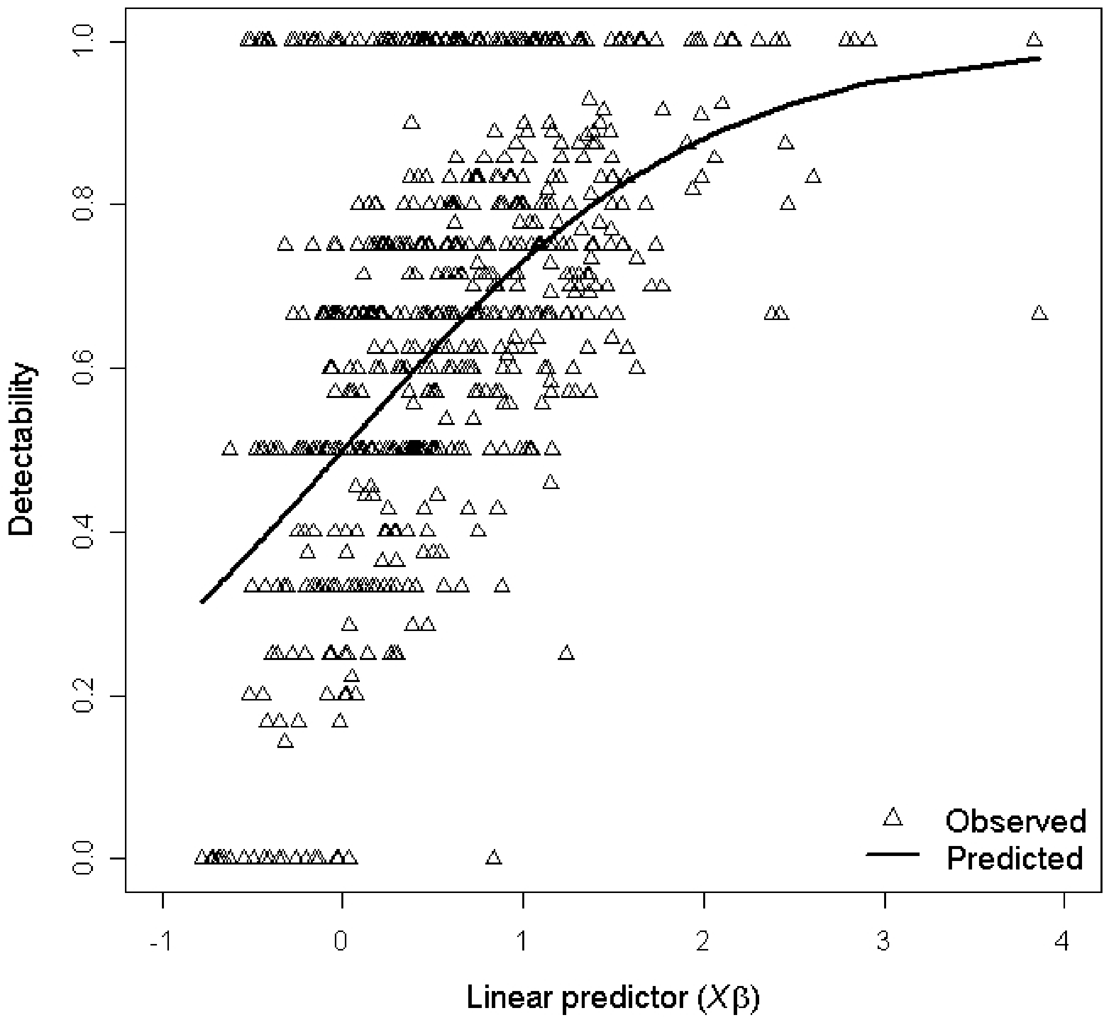

Figure 5. Jackknife estimate of predictability for model 13, where the linear predictor for detection probability contained *TIME*, *SITE*, *DISTURB*, *CROWS*, and *CROWS×SITE* effects. Observed detectability was the proportion of crowing males detected in the first 2-minute listening period of each sampling event (site-station-date). Predicted detectability was the predicted probability of detection for each event (site-station-date) based on fitting model 13 to all other sampling events.

RESEARCH PROPOSAL: ESTABLISHMENT AND MAINTENANCE OF FORBS IN EXISTING GRASS STANDS

Molly Tranel

INTRODUCTION

MNDNR Managers have requested information on establishing and maintaining an abundance and diversity of forbs in grasslands. A diversity of forbs in native grass stands provides better habitat for invertebrates and grassland birds, creates more heterogeneous vegetation structure, and may resist invasion from non-native invasive plants. Interseeding a diverse mix of forbs directly into existing vegetation, followed by management treatments that reduce competition with established grasses may provide a means for forbs to be reintroduced into species-poor grass stands. I developed a study to investigate the effects of 2 mowing and 2 herbicide treatments on: (1) the diversity and persistence of forbs interseeded into established grasslands in the farmland region of Minnesota, and (2) the abundance and community response of insects to interseeding treatments. Forbs will be interseeded in fall 2009 on 15-20 sites located throughout the farmland region that contain >10 acres of uniform, native grasses with few to no forbs present. During summer 2010, 4 management treatments of mowing (once or twice per season) and grass-selective herbicide application (at a high and low rate) will be randomly applied to plots within each site. Beginning in summer 2011, I will evaluate treatment effects by estimating plant species richness based on the presence/absence of 30 planted species and volunteer species. I will also assess vegetation structural characteristics using measures of visual obstruction, litter depth, and percent cover. Insects will be collected prior to vegetation sampling using sweep nets and identified to the lowest practical level. Monitoring will continue through summer 2013, or longer if funding allows.

JUSTIFICATION

Although methods of restoring and maintaining grasslands are fairly well known, 82% of Minnesota Department of Natural Resources (MNDNR) wildlife managers responding to a survey on grassland information needs (Tranel 2008) indicated a lack of information on maintaining plant species diversity in restored grasslands. In particular, managers wanted more information on establishing and maintaining an abundance and diversity of forbs in grasslands. Research on forb establishment could provide managers with the most effective management techniques to improve grassland wildlife habitat.

One important function of forbs in grasslands is to provide habitat for invertebrates, an essential food for grassland birds and their broods (Buchanan et al. 2006). Insect abundance is strongly associated with forb abundance (Jones 1963) and diversity (Haddad et al. 2001), and insect abundance in chick diets has been positively correlated with growth rates and survival in gallinaceous birds such as grouse (Park et al. 2001, Huwer et al. 2008), gray partridge (*Perdix perdix;* Sotherton and Robertson 1990), and pheasants (*Phasianus cholchicus*; Hill 1985). Broods of gallinaceous birds such as the prairie chicken (*Tympanuchus cupido*) move directly from nests to brood habitat (Svedarsky 1979), and habitats with high forb abundance were preferred (Jones 1963, Drobney and Sparrow 1977).

A diversity of forbs also creates more heterogeneous vegetation structure, which some birds require for nesting and brooding (Volkert 1992). Sample and Mossman (1997) reported "several bird species such as dickcissel (*Spiza americana*) and savannah sparrow (*Passerculus sandwichensis*) are most abundant in fields with a strong forb component." In addition to the wildlife benefits, a diverse vegetation structure may better resist invasion of non-native invasive species (Pokorny et al. 2005, Sheley and Half 2006) and increase overall ecosystem health and function.

Regenscheid et al. (1987) found that insect availability and feeding rates of partridge and pheasant chicks were lowest in monotypic switchgrass stands compared to other cover types, and Leathers (2003) reported that mean invertebrate biomass was higher in interseeded Conservation Reserve Program (CRP) fields than controls. A number of MNDNR grasslands that were restored 10-20 years ago were seeded at heavy grass rates, with little or no forb component. In other grass stands, forbs were seeded but are now absent or present in low numbers. As a result, grasslands with little or no forbs are common in the farmland region of Minnesota. Over 60,000 acres of CRP contracts in Minnesota are scheduled to expire in 2009 (U.S. Department of Agriculture 2009). With this potential loss of grassland habitat, restoring the remaining grasslands to their full potential by increasing forb diversity and abundance will become more essential to wildlife.

Managers interested in increasing the diversity and quality of species-poor stands are faced with the costly option of completely eliminating the existing vegetation and planting into bare ground, or attempting to seed forbs directly into that vegetation. The latter technique, known as interseeding, could potentially reduce labor and fuel costs and protect from soil erosion (Packard and Mutel 1997) and was recommended by Rodgers (1999) to improve pheasant habitat. However, MNDNR managers report having limited experience and poor success with a few early interseeding attempts.

Management techniques that reduce competition from established grasses may provide an opportunity for forbs to become established in existing grasslands (Collins et al. 1998). Temporarily suppressing dominant grasses may increase light, moisture, and nutrient availability to seedling forbs, ultimately increasing forb abundance and diversity (Schmitt-McCain 2008). Mowing and application of selective herbicides are familiar techniques for controlling grasses. Williams et al. (2007) found that grasslands mowed frequently in the first growing season increased interseeded forb emergence and reduced forb mortality. Similarly, Hitchmough and Paraskevopoulou (2008) found that forb density, biomass, and richness were greater in meadows where a grass herbicide was used.

In this study, I will investigate the effects of 2 mowing and 2 herbicide treatments on diversity and abundance of forbs interseeded into established grasslands in southern Minnesota. In addition, I will monitor insect abundance in response to interseeding treatments. Finally, I will track the cost of implementing each management technique and conduct a cost-benefit analysis.

STUDY AREA

The study will be conducted on 15-20 sites located throughout the farmland region (Figure 1). I have identified potential sites located on state, federal, and privately owned properties. Only sites with ≥10 uniform acres of native grass with similar soils and no or few forbs will be used. The number of sites will be determined based on a power analysis of pilot study results, conformity of sites to criteria described above, willingness of property managers to fully participate in the study design, and available budget.

Pilot Site

I will assess feasibility of treatments and potential for identifying forb seedlings on 1 pilot site during summer 2009. This site is located on the Wood Lake Wildlife Management Area (WMA) in Redwood County and is a new MNDNR acquisition currently enrolled in the CRP. It was planted to 8 native grass species in 1999, burned in October 2008, and frost seeded with forbs in January 2009 (Figure 2). In addition to the pilot site, I will estimate number of vegetation and insect samples needed by sampling established grasslands with varying forb abundance throughout the farmland region during summer 2009.

METHODS

Prior to interseeding, all sites will be surveyed to determine plant species already present and general condition of each site. Sites will then be prepared in the following manner: mowed in August 2009, burned in October 2009, and interseeded with native forbs during fall 2009 - winter 2010 (dormant season). For the purpose of this study, interseeding is defined as dormant season broadcast seeding directly into existing grass stands without prior tilling or herbiciding. The same seed mix (Table 1), consisting of 29 native forb species and 1 sedge species, will be interseeded at each site.

Treatments

After each site is prepared and seeded, I will select a block with relatively uniform soil type, topography, and other physical characteristics and divide it into 10 plots (Figure 3) of equal size. I will then randomly assign 1 of the 4 treatments and the control to each of the 10 plots. Each of the 4 treatments and control will be replicated twice within each block. The following treatments, which are designed to suppress grass competition, will be applied during the 2010 growing season while the forbs are becoming established:

- Mow to a height of 4-6 inches once when vegetation reaches 10-12 inches in height.
- Mow to a height of 4-6 inches twice when vegetation reaches 10-12 inches in height.
- Apply grass herbicide Clethodim (Select 2 EC) at an 8 oz/acre rate in late May - early June.
- Apply grass herbicide Clethodim (Select 2 EC) at a 16 oz/acre rate in late May - early June.

Vegetation Sampling

I will permanently mark the corners of treatment plots with metal stakes and record their coordinates using a Global Positioning System. I will locate sampling frames along a transect running through the center (excluding edges) of each treatment plot (Figure 3). I will estimate number of transects, transect length, and number of sampling frames needed from the pilot study based on number of plots required to detect >80% of the forb species present in the treatment parcel. I will sample vegetation twice/year, once in the early part of the growing season (June) and once in the late part of the growing season (September), beginning the year after treatment application.

I will estimate species richness by counting the presence/absence of the 30 planted species (Table 1), species present prior to the study, and volunteer species in each sampling frame. I will use Simpson's diversity index (Simpson 1949) to account for species richness and the proportion of each species present. Simpson's diversity index (*D*) is:

$$D = \frac{1}{\sum_{i=1}^{S} p_i^2}$$

D Simpson's diversity index

S total number of species in the community (richness)

p_i proportion of *S* made up of the *i*th species (# species ***i* /** total # of species)

Managers are interested in knowing which planted forb species become established and tend to persist over time. Therefore, I will primarily focus my vegetation sampling on species richness and diversity. However, structural measurements are useful in determining the value of a vegetation community in providing wildlife habitat (Herrick et al. 2005). Sample and Mossman (1997) categorized grassland habitat for birds based on measures of vegetation height-density, litter depth and cover, cover of standing residual vegetation, cover of standing live vegetation, cover of bare ground, and ratio of grass to forb cover. I will assess structural characteristics using measures of visual obstruction (Robel et al. 1970), litter depth, and percent cover of forbs, grass, and bare ground (Daubenmire 1959).

Insect Sampling

If time and budget allow, I will assess the response of the insect community to management treatments. Immediately prior to vegetation sampling, I will collect insects using a 15 inch sweep net along randomly selected segments of the transect through each plot (Leathers 2003). Insects will also be sampled from control plots on each site located in areas that have not been interseeded. Data from the pilot study will be used to determine insect sampling intensity. Collected insects will be frozen, sorted by size, and identified to the lowest taxonomic level.

Timeline (Figure 2)

- **Spring-Summer 2009:** Determine sample size, variance, and sampling techniques from pilot sites. Ground truth study sites and prescribe management treatments.
- **Fall 2009 – Winter 2010:** Initiate full study, collect pre-treatment measurements, mow, burn, and interseed study sites.
- **Summer 2010:** Apply management treatments to study sites.
- **Summer 2011:** Sample vegetation and insects during June and September.
- **Summer 2012:** Sample vegetation and insects during June and September.
- **Summer 2013:** Sample vegetation and insects during June and September

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Table 1. Prairie plant species and seeding rates selected for interseeding into established grasslands in southern Minnesota, 2009. Prices are estimates. Total cost is the total cost of each species per acre. It was determined by multiplying cost per ounce multiplied by the ounces per acre for each species.

Common name	*Scientific name*	Ounce/ acre	Seeds / acre	% of mix[1]	Total cost[2]	Tolerance to transline
Leadplant	*Amorpha canescens*	0.50	8,000	0.79%	$ 6.50	Unknown
Black eyed Susan	*Rudbeckia hirta*	1.20	110,400	10.92%	$ 4.80	Susceptible
Maximilian Sunflower	*Helianthus maximilianii*	0.50	6,500	0.64%	$ 2.50	Susceptible
Yellow Coneflower	*Ratibida pinnata*	1.35	40,500	4.01%	$ 6.75	Susceptible
Golden Alexanders	*Zizia aurea*	1.00	11,000	1.09%	$ 7.00	Tolerant
Sky Blue Aster	*Aster oolentangiensis*	0.85	68,000	6.73%	$ 6.80	Moderate
Canada Milk Vetch	*Astragalus canadensis*	1.75	29,750	2.94%	$ 8.75	Unknown
Prairie Cinquefoil	*Potentilla arguta*	0.85	195,500	19.35%	$ 8.50	Tolerant
White Prairie Clover	*Dalea candida*	1.50	28,500	2.82%	$ 6.00	Moderate-Susceptible
Purple Prairie Clover	*Dalea purpurea*	1.50	22,500	2.23%	$ 6.00	Moderate-Susceptible
False Sunflower	*Heliopsis helianthoides*	1.25	7,875	0.78%	$ 5.00	Moderate
Alumroot	*Heuchera richardsonii*	0.05	35,000	3.46%	$ 3.00	Moderate
Narrow Leaf Purple Coneflower	*Echinacea angustifolia*	0.85	5,950	0.59%	$ 10.20	Unknown
Virginia Mountain Mint	*Pycnanthemum virginianum*	0.20	44,000	4.35%	$ 6.00	Unknown
Common Milkweed	*Asclepias syriaca*	1.00	4,000	0.40%	$ 5.00	Tolerant
Blue Vervain	*Verbena hastate*	0.75	69,750	6.90%	$ 3.75	Tolerant
Rough Blazingstar	*Liatris aspera*	0.15	2,400	0.24%	$ 7.50	Moderate
New England Aster	*Aster novae-angliae*	0.65	42,900	4.25%	$ 5.85	Moderate
Prairie Onion	*Allium stellatum*	0.70	7,700	0.76%	$ 5.60	Tolerant
Hoary Vervain	*Verbena stricta*	0.65	18,200	1.80%	$ 3.90	Tolerant
Heath Aster	*Aster ericoides*	0.15	30,000	2.97%	$ 4.50	Moderate
Stiff Goldenrod	*Oligoneuron rigidum*	0.75	30,750	3.04%	$ 7.50	Moderate
Culver's Root	*Veronicastrum virginicum*	0.10	80,000	7.92%	$ 2.00	Unknown
Showy Tick Trefoil	*Desmodium canadense*	0.85	4,675	0.46%	$ 7.65	Moderate-Susceptible
Wild Bergamot	*Monarda fistulosa*	0.70	49,000	4.85%	$ 7.00	Tolerant
Prairie Coreopsis	*Coreopsis palmate*	0.25	2,500	0.25%	$ 6.25	Unknown
Partridge Pea	*Chamaechrista fasciculate*	2.00	5,400	0.53%	$ 6.00	Unknown
Closed Bottle Gentain	*Gentiana andrewsii*	0.08	22,400	2.22%	$ 4.80	Unknown
Heart Leaf Golden Alexander	*Zizia aptera*	0.20	2,400	0.24%	$ 5.00	Tolerant
Brown Fox sedge	*Carex vulpinoidea*	0.25	25,000	2.47%	$ 1.88	Unknown

[1] % of Mix = (Ounce per Acre * Seeds per Acre) / ∑ Seeds per Acre
[2] Total Cost = Ounce per Acre * Price per Ounce

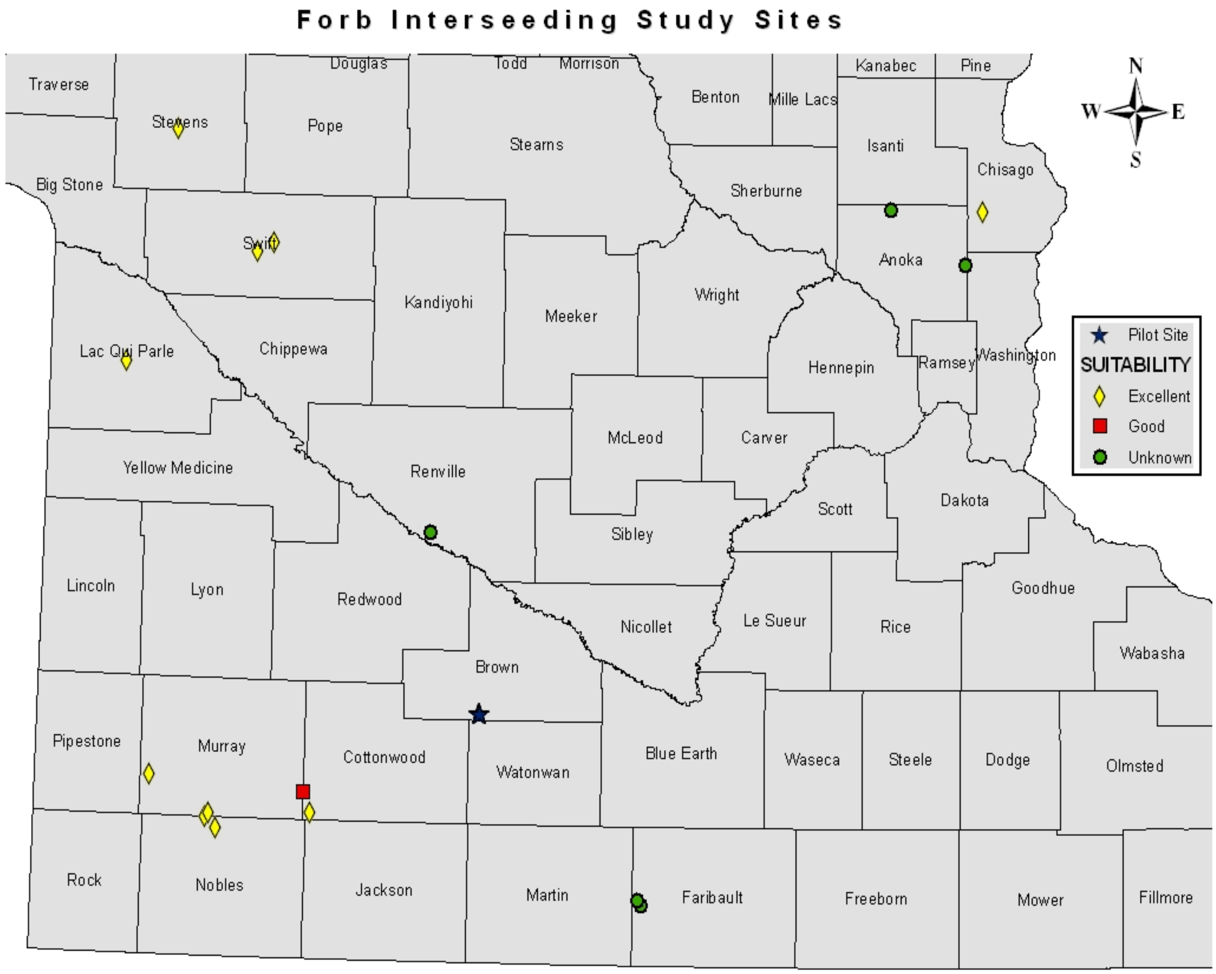

Figure 1. Locations of potential study sites for the forb interseeding study, categorized by suitablility. Sites of excellent suitability were flat, uniform, and contained similar vegetation and soils on >10 contiguous acres. Unknown sites have not been visited yet.

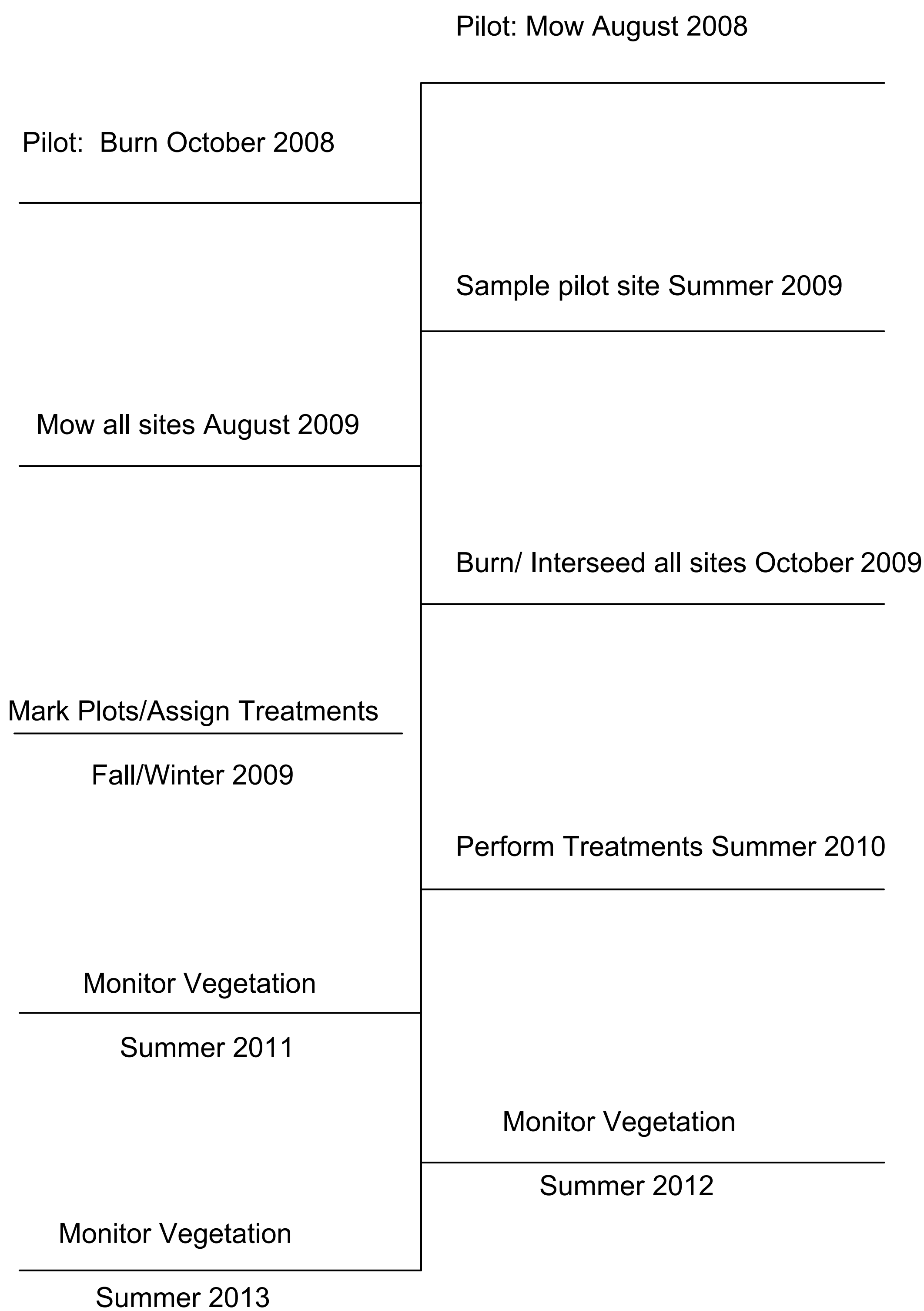

Figure 2. Timeline for forb interseeding study in southern Minnesota, 2009-2013.

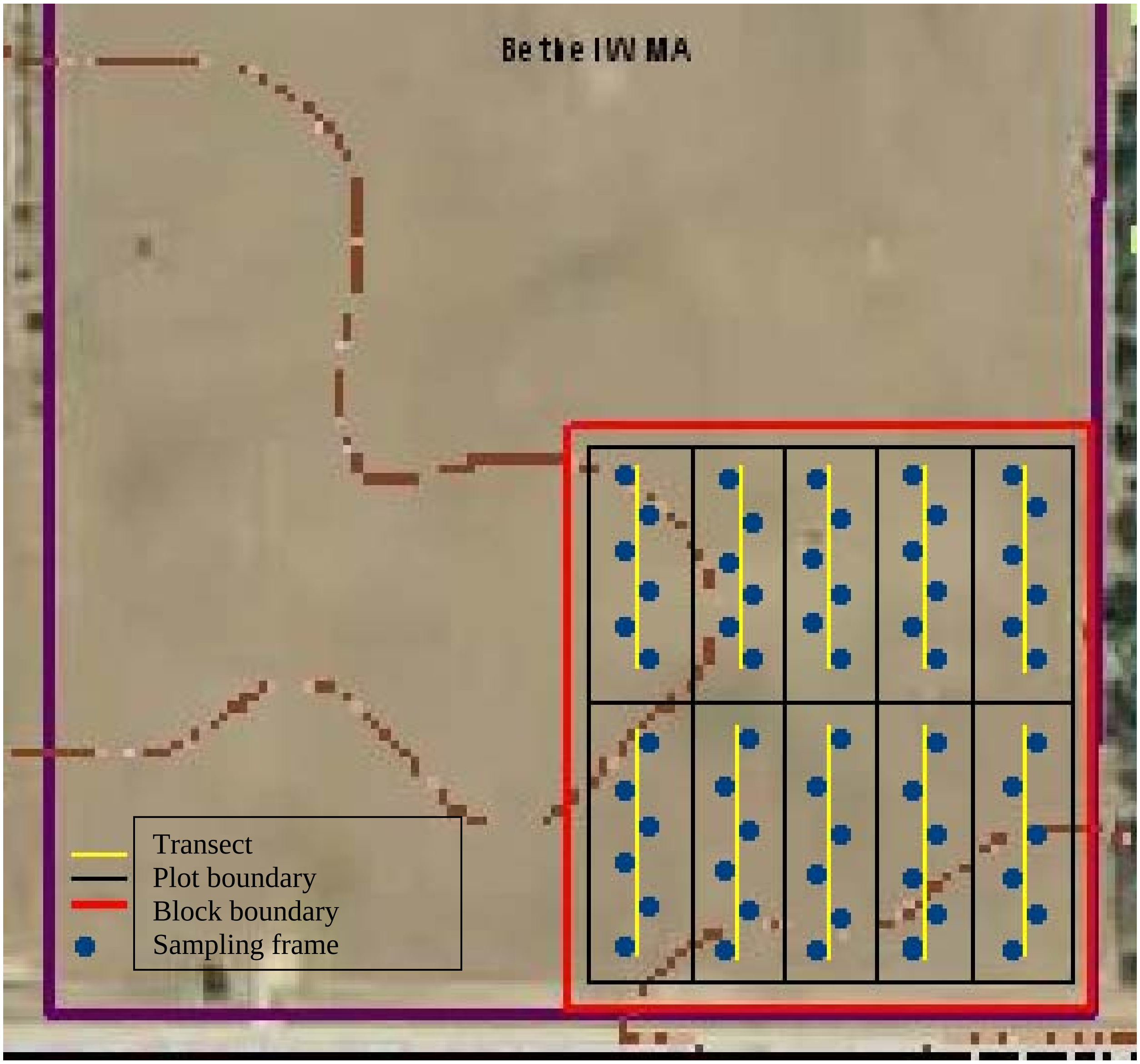

Figure 3. Example of a representative study site on the Bethel WMA. The purple line delineates the entire WMA, and the red line delineates the site (block) that meets study criteria. The black rectangles represent the plots that will receive 1 of 4 management treatments or serve as a control. Each of these treatments and control will be replicated twice within a block. Vegetation will be sampled within frames (blue circles) along transects (yellow lines) running through the center of each plot.

WILD TURKEY FOOD HABITS ON THE NORTHERN FRINGE OF THEIR RANGE IN MINNESOTA

Eric M. Dunton, Joshua T. Ream, John Fieberg, and Kurt J. Haroldson

SUMMARY OF FINDINGS

The purpose of this study was to evaluate diet selection and body condition of eastern wild turkeys (*Meleagris gallopavo silvestris)* in agricultural and forested areas on the northern fringe of their range in Minnesota. During winter 2009, 7 turkeys were collected in forested habitat and 24 in agricultural habitat. Adult females in forested habitat had 32% less body weight, 72% less total body fat, and were assigned to lower body condition classes than adult females from agricultural habitats. Forested turkeys' diets consisted of a mixture of high energy (acorns) and low energy (grass, smooth rose, leaf litter) food items, while diets of turkeys located in agricultural habitats consisted primarily of high energy (corn) food items.

INTRODUCTION

The current range of the eastern wild turkey extends far north of what was identified by Schorger (1966) as their historical range. This northern expansion has been associated with increased availability of food during winter (Wunz 1992, Wunz and Pack 1992, Kubisiak et al. 2001), which was considered limiting prior to settlement by European farmers. Wild turkey range in Minnesota and throughout the northeastern United States and southeastern Canada is currently expanding northward beyond agricultural areas (Kimmel and Krueger 2007). It is unknown how far turkeys will expand outside of mixed forest-agriculture areas into northern forest areas, and what their diet will include. Understanding winter diet selection of turkeys on the northern periphery of their range and the interaction of agriculture, snow conditions, and food habits will provide management tools to enhance turkey survival outside of an agriculturally dominated landscape.

The eastern wild turkey is a food generalist with a winter diet ranging from >20 species (Korschgen 1967) to a restricted diet of only corn (Porter et al. 1980). As wild turkey range expanded north through mixed forest-agricultural habitats, Porter (2007) stated, "Looking back at the field studies of the 1970s, it is clear that they were telling us more than we realized: snow and cold are not the issue, the key is food." Adequate information is available on turkey foraging behavior and survival in northern turkey habitats with access to agricultural foods (Porter et al. 1980, Vander Haegen et al. 1989, Kassube 2005, Kane et al. 2007), but information is lacking on turkey food habits in northern non-agricultural areas.

Our objectives were to: (1) determine winter foods used by wild turkeys on the northern fringe of their range in Minnesota; (2) describe diet as a function of agriculture and snow condition; and (3) compare body condition of wild turkeys with access to high-energy agricultural diets to those without.

STUDY AREA

We conducted this study north of Minnesota's historical wild turkey range (Leopold 1931, Schorger 1966, Snyders 2009) where wild turkey populations were established by translocation during 1990s - 2008. This region is located within the Western Superior Uplands and Northern Minnesota Drift and Lake Plain Ecological Sections of the Laurentian Mixed Forest Ecological Province (MNDNR 2003). The 25,959 km^2 study area is comprised of 35% upland deciduous forest, 31% crop/grass, 16% aquatic environment, 10% shrubland, 4% upland conifer forest, 2% lowland conifer forest, 2% lowland deciduous forest, and 1% non-vegetated (GAP Analysis Program MNDNR 2008).

To aid in locating wild turkeys with and without access to agricultural foods, we stratified the study area using a 500 ha grid and classified each cell to 1 of 3 turkey habitat categories based on reclassified GAP land cover data: agricultural cells contained ≥ 30% cropland and ≥ 20% forested habitat; forested cells contained ≥ 50% forested habitat and 0% cropland; and other cells contained all other combinations of habitats (Figure 1). We used a focal sum analysis to identify forested cells that were surrounded by other forested cells. We focused surveys on clusters of forested cells, but we also surveyed some forested cells that were surrounded by agricultural cells.

METHODS

Using fixed wing aircraft we attempted to locate an equal number of wintering flocks of turkeys in agricultural and forested strata. We used real-time, moving-map software (MNDNR 2005) coupled to a global positioning system receiver and a tablet-style computer to guide transect navigation and record turkey locations and aircraft flight paths directly to a geographic information system (Haroldson 2007). Turkeys were then relocated on the ground within 1-3 days. We then attempted to collect 1-5 turkeys from each flock in late afternoon or early evening, when crops are most likely to be full (Hillerman et al. 1953), by shooting.

At each collection site, we recorded date, snow depth, snow condition (e.g., crusted vs. powder snow), temperature, habitat class (agricultural versus forested), and geographic coordinates. We verified habitat class by plotting collection sites on Farm Service Agency 2008 aerial imagery, and identifying presence or absence of cropland within a 1,545-m radius buffer (based on the 750 ha winter home range of wild turkeys in Minnesota reported by McMahon and Johnson (1980)). Habitat was classified as forested if no cropland was located within the buffer; otherwise habitat was classified agricultural.

We evaluated body condition of wild turkeys collected in forested and agricultural habitats based on relative body weight and 3 estimates of body fat. We estimated total body fat of adult hens using a formula from Pekins (2007). We also assigned turkeys to 1 of 4 body condition classes based on amount and color of visible fat (Carter 1970). Finally, we assigned turkeys to 1 of 3 classes based on the amount of fat visible on the gizzard. We tested for differences in estimated weights and in the distribution of body condition classes among adult females in agricultural and forested habitats using a t-test and Monte Carlo χ^2 test (Hope 1968), respectively. Tests were conducted using the t.test and chi square.test functions in the R programming language (R Development Core Team 2008).

We determined frequency of occurrence and weight of food items present in the crops and gizzards according to the methods of Korschgen (1967). We determined dry matter content of foods by drying to a constant weight at 50°C (Decker et al. 1991). We assigned each food item 1 of 3 classes (high, medium, and low) based on energy content (Decker et al. 1991).

RESULTS

During 2 January – 24 February 2009, we aerially surveyed 122 forested strata and 103 agricultural strata. We located 0 turkeys in forested strata and 1,130 turkeys (mean flock size = 23) in agricultural strata. We collected 31 turkeys; 7 from forested habitats (i.e., no agricultural foods present within 1545 m), and 24 from agricultural habitats. The 7 turkeys collected in forested habitat were all found by MNDNR staff on roads and trails that were plowed for logging, military training, or state park use. Snow plowing exposed snow-covered ground in forested habitats and turkeys were found foraging on the newly plowed roads. Adult females from forested habitats had 32% less body weight ($P < 0.001$), 72% less total body fat, and were assigned to lower body condition classes than adult females from agricultural habitats ($P = 0.0012$) (Table 1). One juvenile female and 1 adult male collected in forested habitats weighed 3% and 30% less, respectively, than their counterparts in agricultural habitats (Table 1). We classified gizzard fat from 6 forest habitat birds as no fat and 1 as fat. In agricultural habitats, we classified 14 turkey gizzards as very fat and 10 as fat.

We classified snow conditions at all forested collection sites as powder, and snow depth averaged 39.2 cm. We classified snow conditions at 10 agricultural sites as powder and 14 as crusted. Snow depth at agricultural sites averaged 27.4 cm across both snow condition classes. Mean collection date of forested birds was 24, January 2009 (range 14 January – 7 February) and 30 January 2009 (range 8 January – 24 February 2009) for agricultural birds. Temperature and monthly snow depth data from the Minnesota Climatology Working Group were not available at the time of this report.

High energy food (i.e., acorn [*Quercus spp.*]) was found in 86% of the crops from forest-habitat turkeys but formed only 47% of the crop contents by weight (Table 2). For agricultural habitat turkeys, high energy foods (e.g., corn [*Zea mays*]) formed 86% of the crop contents by weight (Table 3). Crops from 8% of turkeys collected in agricultural habitats and 14% of turkeys collected in forested habitats were empty or nearly empty.

DISCUSSION

Butler et al. (2007) found that turkey sightability was primarily influenced by flock size and vegetative cover. Our ability to detect birds was also influenced by snow conditions, in addition to type of cover (agricultural versus forested habitat). When snow depth was >25 cm, large wintering flocks of wild turkeys were often congregated near agricultural food sources and were easily detected using fixed wing aircraft. However when snow depth diminished in late February, aerial surveys in agricultural areas became less effective. Aerial surveys in forested strata were ineffective both with and without deep snow even though we searched forested strata that were known to have established populations. We suspect that some turkeys in forested areas moved to areas with supplemental food sources or remained in forested habitats in small groups that were hard to detect by aerial survey.

The forested turkeys we collected were all found along roads and trails where snow plowing occurred. Snow plowing permitted ground foraging on only a small fraction of forested habitats, which may account for the high frequency of occurrence (86%) of high energy food items (acorns) found in crops of turkeys collected in forested habitats, but the lower proportion of total weight (47%). Not surprisingly, turkeys collected in agricultural habitats had been feeding on corn and other high energy foods that were not available in forested habitats (e.g., (Porter et al. 1980, 1983; Vander Haegen et al. 1989, Healy 1992). Although, Hurst (1992) considered crop content analysis the best technique for evaluating wild turkey food habits, it is negatively biased toward succulent foods and soft-bodied invertebrates, which are digested more rapidly than hard and fibrous food items. In this study, bias is assumed minimal because few succulent foods were available during winter

Body weights of adult and juvenile hens collected in forested habitats in this study were below average whereas body weights of adult and juvenile hens collected in agricultural habitats were within the average range reported by Porter (1980) in Minnesota, Vander Haegen et al (1989) in Massachusetts, and Coup and Pekins (1999) in New Hampshire. Pekins (2007) suggested that adult hens weighing < 3.0 kg have minimal body fat and were approaching a critical threshold of malnutrition. Thus, most adult hens collected from forested habitats in this study were showing signs of food deprivation. As supporting evidence, we frequently observed turkeys in forested habitats remaining in their roosts late in the morning. This behavior is generally considered an indication of stress (Hayden and Nelson 1963).

Findings from this study indicate that turkeys in agricultural areas were able to find sufficient food (primarily corn) to maintain energy balance and fat reserves throughout the winter, even when snow depth was >25 cm. In contrast, turkeys using exclusively forested habitats in deep snow were in poor body condition with little to no fat reserves. Powder snow >15-20cm hinders mobility, and >30cm can prevent movement of wild turkeys (Austin and DeGraff 1975, Porter 1977, Healy 1992). Deep persistent snow cover can ultimately result in starvation. Wild turkeys began starving when snow depth was >30cm for >2 weeks in Pennsylvania (Wunz and Hayden 1975), 49 days in Wisconsin (Wright et al. 1996), and 40-59 days in New York (Roberts et al. 1995). Wright et al. (1996) documented starvation when deep snow restricted movements even though food was available within 0.8 km.

Further range expansion of wild turkeys in Minnesota's northern forests may be limited by availability of supplemental food during winter. Wild turkey range may expand during periods with consecutive mild winters and then contract during severe winters. Because opportunities for agriculture are limited in this region, unharvested crops and livestock feeding operations may attract large concentrations of wintering turkeys, resulting in depredation complaints.

ACKNOWLEDGEMENTS

Funding for this project was provided by MNDNR with contributions from Minnesota State Chapter of the National Wild Turkey Federation. R. Wright, MNDR, provided GIS support. MNDNR wildlife, parks, and forestry staff and Camp Ripley staff provided locations of forested turkeys and assisted with data collection. Private landowners in the study area allowed access to their lands. T. Buker, MNDNR enforcement pilot, flew surveys. B. Haroldson provided assistance with survey software. W. Smith, MNDNR Ecological Resources, provided assistance with food item identification. P. Pekins, University of New Hampshire, provided comments on the proposal.

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Table 1. Estimates of body fat for 31 wild turkeys collected in forested versus agricultural habitats on the northern fringe of their range in Minnesota during winter, 2009.

					Visual fat score				Estimated total body fat[e]	
Habitat	Gender	Age	n	Average weight (kg)	Very fat[a]	Fat[b]	Lean[c]	Thin[d]	kg	%
Forest	Female	Adult	5	3.24	0	0	2	3	0.21	5.80
Forest	Female	Juvenile	1	3.81	0	0	1	0	-	-
Forest	Male	Adult	1	6.48	0	0	1	0	-	-
Forest	Male	Juvenile	0	0	0	0	0	0	-	-
Ag	Female	Adult	14	4.75	5	5	4	0	1.02	20.57
Ag	Female	Juvenile	3	3.91	0	2	1	0	-	-
Ag	Male	Adult	3	9.26	2	1	0	0	-	-
Ag	Male	Juvenile	4	6.41	0	4	0	0	-	-

[a] Large deposits of fat on mid-line of breast, thighs, back, around crop, at the posterior of the body cavity, and immediately beneath skin. Fat is bright yellow (Carter 1970).
[b] Large fat deposits on back and thighs and reduced deposits elsewhere. Fat may be orange in color. (Carter 1970).
[c] Fat deposits are completely resorbed. Breast muscle has “normal” contour. Dark orange color in cellular framework of resorbed fat deposits (Carter 1970).
[d] Breast muscle attains wedge-like appearance (“hatchet-breast”). Skin resembles parchment (Carter 1970).
[e] Body fat (g) = 571.3 x (kg body weight) – 1696; $R^2 = 0.59$, $P < 0.05$ (Pekins 2007)

Table 2. Crop contents and energetic value of food items for 7 wild turkeys collected in forested habitats on the northern fringe of their range in Minnesota during winter, 2009.

	Weight			
Food item	Total (g)	% of total	Frequency (%)	Energetic diet class
Acorn (*Quercus spp.)*	158.9	46.6	86	High
Grass (*Poa spp.*)	98.7	28.9	86	Low
Smooth rose (*Rosa blanda*)	48.5	14.2	14	Medium
Leaf litter	30.3	8.9	100	Low
Hazel catkins (*Corylus spp.*)	2.7	0.8	29	Low
Unknown seed	0.8	0.2	14	Medium
Feathers	0.5	0.2	14	Unknown
Sumac (*Rhus spp.*)	0.4	0.1	29	Medium
Poison ivy (Toxicodendron spp.)	0.3	0.1	14	Medium
Beetle (*Coleoptera spp.*)	0.05	TR[a]	14	Unknown
Total	341.2			
Mean forested turkey crop weight	48.7			

[a] Trace (TR) amount of food item present in diet < 0.1

Table 3. Crop contents and energetic value of food items for 24 wild turkeys collected in agricultural habitats on the northern fringe of their range in Minnesota during winter, 2009.

Food item	Weight		Frequency (%)	Energetic diet class
	Total (g)	% of total		
Corn kernels (*Zea mays*)	821.0	79.8	92	High
Corn parts (*Zea mays*)	30.3	3.0	29	Low
Sunflower seed (*Helianthus spp.*)	38.3	3.7	17	High
Grass (*Poa spp.*)	33.3	3.2	42	Low
Millet (*Panicum spp.)*	17.8	1.7	13	High
Cow manure	16.8	1.6	8	Unknown
Club moss (*Lycopodium spp.*)	14.7	1.4	17	Low
Sensitive fern (*Onoclea sensibilis*)	12.2	1.2	17	Low
Unknown seed	10.2	1.0	4	Medium
Oats (*Avena sativa*)	8.5	0.8	13	High
Leaf litter	7.0	0.7	33	Low
Pine needle (*Pinus spp.*)	7.1	0.7	8	Low
Poison ivy (*Toxicodendron spp.*)	3.6	0.3	4	Medium
Unknown Forb seed	3.1	0.3	4	Medium
Acorn (*Quercus spp.)*	2.3	0.2	8	High
Hazel catkins (*Corylus spp.*)	1.2	0.1	13	Low
Thistle seed (*Guizota spp.*)	0.6	0.1	4	High
Aven seed (*Geum spp.*)	0.4	TR[a]	8	High
Unknown tree/shrub bud	0.1	TR[a]	13	Medium
Feather	0.3	TR[a]	4	Unknown
Snail	0.1	TR[a]	4	Unknown
Unknown insect cocoon	0.0	TR[a]	4	Unknown
Beetle (*Coleoptera spp.*)	0.0	TR[a]	4	Unknown
Total	1028.8			
Mean agricultural turkey crop weight	42.9			

[a] Trace (TR) amount of food item present in diet < 0.1

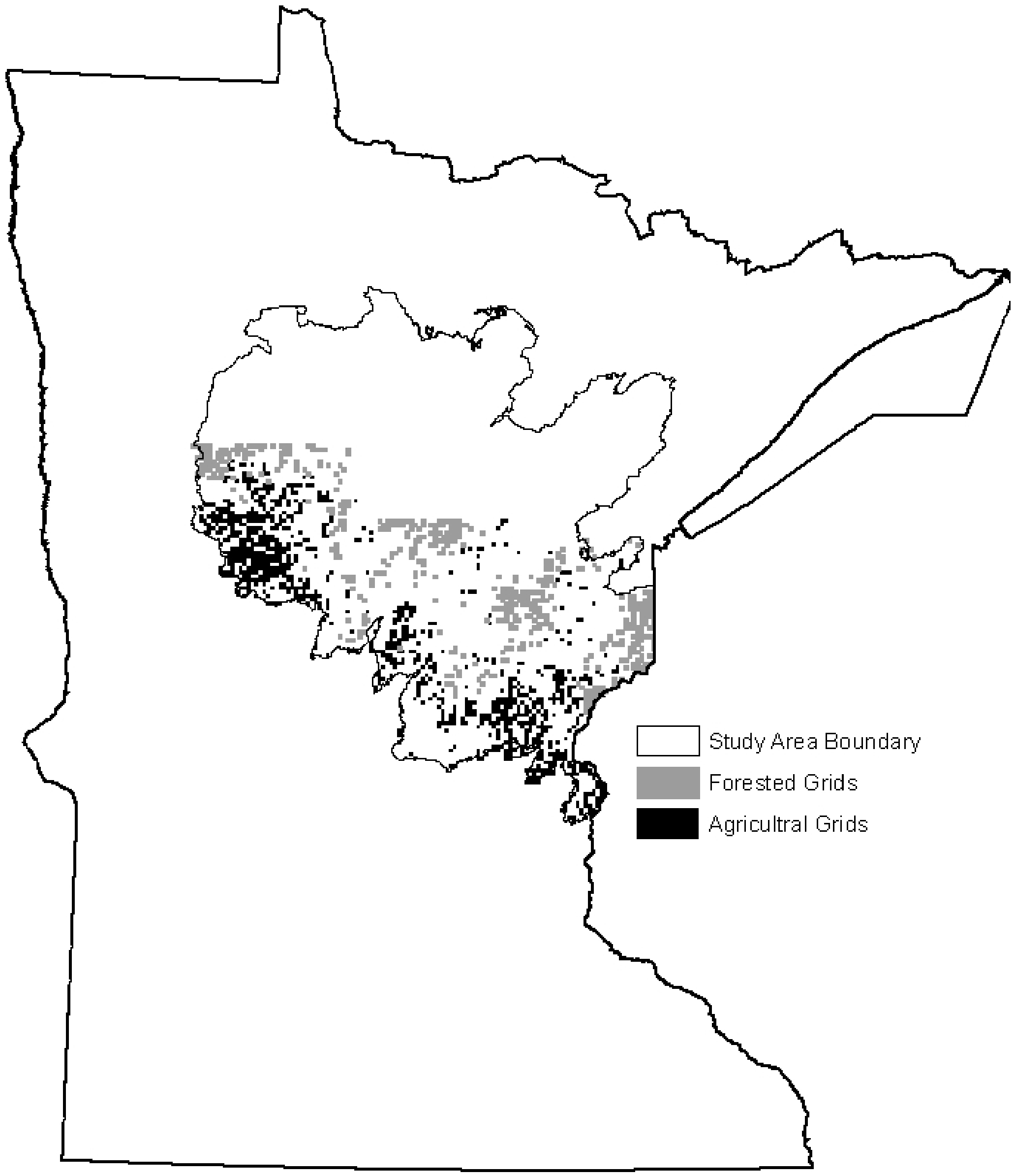

Figure 1. Study area location, forested and agricultural grids surveyed to determine wintering wild turkey flock location for winter food habits project on the northern fringe of wild turkey range in Minnesota, 2009.

FISHER AND MARTEN DEMOGRAPHY AND HABITAT USE IN MINNESOTA

John Erb, Barry Sampson, and Pam Coy

SUMMARY OF FINDINGS

Following evaluation of field methods in 2007-08, we began full-scale fieldwork during winter 2008-09 on a study of fisher (*Martes pennanti*) and marten (*Martes americana*) ecology in northern Minnesota. Including the pilot year of the project, a total of 47 martens and 30 fishers have been radiocollared. Of the 47 marten radiocollared (25M, 22F), 7 individuals (2M, 5F) were able to subsequently slip their collars. In addition, 1 animal has not been relocated since shortly after capture. Of the remaining 39 animals, 26 are currently alive, 6 have died from predation (3 raptor kills, 3 mammalian carnivores), 3 were harvested during the legal trapping season, 2 died from capture or collar-related complications, and 2 are missing. Of the 30 fisher radiocollared (11M, 19F), 9 have shed their collars (2M, 6F), 8 due to insufficient collar design. Of the remaining 21 fisher, 13 are alive, 1 has not been relocated since release, 5 have died from predation (2 raptor kills, 3 mammalian carnivores), 1 additional collar (but no animal remains) was found under an active eagle nest, and 1 animal was trapped outside of the legal harvest season. Age information is not yet available, but 8 of the 13 female fishers monitored this spring produced litters, with an average litter size of 2.9. All natal dens were in tree cavities, primarily in large-diameter aspen trees or snags. Twelve kits from 4 females have subsequently died after the nursing females were killed by other predators. Most fisher kits appear to have been born during the first 10 days of April. Monitoring of marten reproductive success is ongoing, and it appears marten may give birth later than fisher. As of this writing, presence of kits has only been confirmed for 2 of 12 female marten (1 litter in a burrow within a rock/dirt berm, the other in a cedar tree cavity). We suspect other females have kits that we have yet to locate. Opportunistic sampling of rest sites used by marten suggests that during winter, most rest sites are underground, with increasing use of on- or above-ground locations in spring and summer. The majority of rest sites used by fisher, regardless of season, have been in tree cavities, with some use of on- or below-ground sites. Preliminary analysis of temperature sensor data from 4 fishers suggests that during winter they may spend over 75% of their time in den/rest sites. Data from 1 female fisher suggests that females with kits may spend only 25% of their time at the natal den during the first 10 days after kits are born. No temperature sensor data from radiocollared marten has yet been recovered. Visual examination of radiolocation data indicates that marten home ranges may be from 1 – 3 mi^2, while fisher home ranges may vary between 3 – 5 mi^2. The longest confirmed dispersal distance for one of our radiocollared fisher has been approximately 18 miles, while the longest confirmed dispersal distance for one of our radiocollared marten has been approximately 8 miles.

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 counties as far west as Roseau county. Limited information suggests they occurred as far south and west as Crow Wing and Polk counties. As a result of over-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, and also into southeast Minnesota (Swanson et al. 1945, Balser and Longley 1966). The impacts of over-harvest and habitat alteration were equally as detrimental to fisher, with populations substantially reduced by the 1930s.

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

While both species appear to have naturally re-colonized a significant portion of their historic range, Minnesota-specific information on species biology and ecology is limited. Except for carcass data obtained from harvested fisher and marten, we are aware of only 1 published field study in Minnesota. Specifically, Mech and Rogers (1977) opportunistically radiocollared 4 marten and reported survival and home range information for those animals. This information is now nearly 30 years old, and based on a very limited sample size. The low reproductive potential, low density, and comparatively specialized habitat requirements of fisher and marten make them more susceptible to over-harvest and the negative effects of human development and habitat alteration.

The primary objectives of this study are to: (1) estimate survival rates and causes of mortality for fisher and marten in Minnesota; (2) describe and quantify features of natal den sites used by females; (3) directly estimate parturition rates and, if possible, litter size and kit survival; (4) evaluate variability in survival or reproduction as a function of forest attributes, prey abundance and weather conditions; and (5) to evaluate the design of winter track surveys.

After initial evaluation of field methods during the pilot year of the study (Erb et al. 2008), winter 2008-09 marked the beginning of full-scale research activities. Herein we present basic summaries of field methods and preliminary findings for key objectives, particularly radiocollaring activities and survival and reproductive monitoring. More detailed analyses and other objective-specific methods (e.g., stable isotope analysis, home range/habitat analysis, track survey evaluation) will be detailed in future years as results become available.

STUDY AREA

Marten research is focused on 1 study area located in northeastern Minnesota (Figure 1; Area 1). The area (~ 700 km^2) is composed of 69% mixed 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 0.4% deciduous forest. The area 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 summary of sample sizes and known-fates, 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 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, animals were placed on either chemical hand warmers or heating pads connected to a power inverter and 12 volt battery. Portable propane heaters were also used to keep animals warm during processing. We monitored respiration, pulse, and rectal temperature during anesthesia. We weighed and sexed animals and typically removed a first pre-molar for aging. Morphological measurements taken included body length, tail length, hind foot length, and chest, neck, and head circumference. We removed guard hair samples for possible genotyping, and for evaluating the use of stable isotope analysis for deciphering food habits (Ben-David et al. 1997). To determine which females were pregnant in mid-winter, and eventually the percent of those that actually produce a litter in spring, we attempted to draw blood samples to measure serum progesterone levels (Frost et al. 1997). After gaining some experience the first year, we were usually successful at drawing blood from female fisher the second year, but have been largely unsuccessful at drawing blood from female marten. Antibiotics were administered subcutaneously to all animals prior to release. All blood samples were sent to the University of Minnesota Veterinary Diagnostics Lab for progesterone analysis.

During the pilot year, we deployed several radiocollar designs on fisher, including an ATS M1585 zip-tie collar (~ 43 g), an ATS M1930 collar (~ 38 g), and a Lotec SMRC-3 collar (~ 61 g; deployed on adult males only). During 2008-09 collaring efforts, we primarily deployed ATS M1940 collars (~ 43 g) or Sirtrack TVC-162 collars (~ 45 g) on fisher. The majority of martens in both years have been fitted with a Holohil MI-2 collar (~ 31 g).

In an effort to better understand winter activity patterns and frequency of winter rest site use, we evaluated the potential use of miniature temperature loggers (iButton model DS1922-L, Maxim Integrated Products, Sunnyvale, CA) attached to collars. iButtons were epoxied to a sample of 8 fisher collars and 8 marten collars. We programmed them to record temperature at 30 minute intervals. Recovering the data requires recapturing the animal, or recovering the collar if the animal dies or slips the collar. Temperature monitors were also placed in each of 6 cover types in both Study Area 1 and 2 to allow for analysis of the effects of ambient temperature on animal behavior, reproductive success, and survival. Snow depth readings were also recorded throughout winter along a transect in each of the 6 cover types where the temperature monitors were placed.

All radiolocations, except for some taken during the den-monitoring period, will be obtained from fixed-wing aircraft at approximately weekly intervals. During the pilot year, and periodically thereafter, we will test the accuracy of aerial radiolocations by placing transmitters in known locations of varying forest structure, and compute the mean distance between known and estimated locations. Detailed information on radiolocation methods and analysis will be presented in future years.

While data is absent for Minnesota, nearly all reported fisher natal dens have been in elevated tree cavities (Powell et al. 2003). Marten natal 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). We primarily used ground and aerial radiolocations to locate natal 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 (Jones et al. 1997). If a suspected natal den was located in a tree cavity, we used an MVC2120-WP color video camera (Micro Video Products, Bobcaygeon, Ontario) attached to a fiberglass telescoping pole and connected to a laptop computer to confirm and count kits. Underground dens were examined when possible using the same video probe attached to a flexible rod. Dens were only examined when the radiomarked female was not present. After initial den and litter confirmation, we hope to relocate (if moved) kits at 30-day intervals (up to 120 days) to evaluate kit survival and determine which females recruit at least 1 offspring to the fall population.

Prey sampling transects have been established in both study areas. Prey sampling is being conducted primarily to understand within-area trends in prey abundance, and whether trends are correlated with any observed changes in fisher or marten survival and reproductive success. Transects (n ≈ 150 in each study area) consist of 10 sampling locations spaced 20m apart, distributed in various cover types throughout the study area. Transects will generally be oriented perpendicular to roads or trails, with the first plot 30m off the trail. In spring, we will count snowshoe hare (*Lepus americanus*) pellets in a 1-m^2 plot at each sampling station (McCann et al. 2008). During fall, small mammal snap-trapping will occur for 2 consecutive days at the same sampling stations, similar to protocol used on an existing small mammal survey in Minnesota (Olson 2006). 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 (Mattson and Reinhart 1996, Bayne and Hobson 2000), with our small mammal/hare pellet stations as the sampling points, we will simply record the number of unique squirrels observed/heard along each transect while checking pellet plots and small mammal traps. Information on white-tailed deer and ruffed grouse (*Bonasa umbellus*) populations may be available from existing surveys or population models.

RESULTS AND DISCUSSION

Including the pilot year of the study, a total of 47 martens (25M, 22F) and 30 fishers (11M, 19F) have been radiocollared. Tooth aging has not yet been completed, and herein we do not report any formal survival estimates. Instead, we provide a simple overview of the fate of collared animals.

Of the 47 martens collared, 7 individuals (2M, 5F) were able to subsequently slip their collars, 6 within the first 3 months after release, and 1 more than a year after release. In addition, 1 animal has not been relocated since shortly after capture. Of the remaining 39 animals whose status is known, 26 are currently alive, 6 have died from predation (3 raptor kills, 3 mammalian carnivores), 3 were harvested during the legal trapping season, 2 died from capture or collar-related complications, and 2 are missing.

Nine of the 30 fisher have also shed their collars, 8 due to insufficient collar design, and 1 possibly slipped by the animal (with some evidence of other animal or human involvement). Most of these came off within the first 2 months after release. Of the remaining 21 fisher, 14 are alive, 1 has not been relocated since release, 5 have died from predation (2 raptor kills, 3 mammalian carnivores), 1 additional collar (but no animal remains) was found under an active eagle nest, and 1 animal was trapped outside of the harvest season.

This spring represented the first concerted effort to locate natal dens and confirm kit presence. Based on several ‘measures’, we obtained reproductive information for 13 female fishers (Table 1). Of these, 8 produced litters and 5 were not pregnant. Because age information is not yet available for these animals, we cannot determine whether the females that did not produce litters were of pre-reproductive age, or reproductive-age animals that failed to produce a litter. Mean litter size for the 8 females with confirmed litters was 2.9 (Table 1). All natal dens were located in tree cavities, most commonly large-diameter aspen trees or snags (Table 1). We were unable to determine exact dates of birth, but most kits appear to have been born in the first 10 days of April. To date, 4 nursing females have been killed by other predators, indirectly leading to the death of 12 kits.

Reproductive information for female marten is not yet available. We were unsuccessful at drawing blood from marten, so we do not have pregnancy status data (mid-winter progesterone level) at the time females were captured. Examination of the reproductive tract from a dead female marten indicated she may have been 1-5 days post-implantation in early April, and that birth may occur in early to mid-May. We have confirmed that 2 of the 12 have given birth, while the status of the others is unknown at the time of this writing. It appears that the den attendance patterns of marten make it more challenging to confirm natal dens, at least compared to fisher. Intensive monitoring will continue over the next few weeks.

As part of collar/mortality retrieval efforts and ground checks on potentially denned females, we have had opportunity to document and examine various rest/den sites for both species. Throughout winter, most resting sites we have located for marten are on or below ground (or below snow), most commonly in tunnels through the mossy substrate of lowland conifer stands, but also in tunnels in boulder-laden soil, near the base of trees, or near exposed tree roots. During spring and summer, marten rest sites appear more varied, including those used during winter as well as increased use of above-ground features (tree cavities, hollow logs, and 'witches brooms'). While this sparse and opportunistic sample of resting sites is inadequate to draw any strong conclusions, it appears that martens may primarily use on- or below-ground dens in winter, with increasing use of above-ground sites in other seasons.

The majority of fisher rest sites, regardless of season, have been in tree cavities or hollow logs on the ground. However, we have documented use of a slash pile, an abandoned beaver dam, an abandoned muskrat (*Ondatra zibethicus*) or beaver bank burrow, a tree-squirrel leaf nest, and an underground burrow in an upland area. During winter, we confirmed that 1 female rested in a hollow log, without leaving, for 12 days.

Temperature sensor data has been recovered from iButtons attached to 4 fisher collars. While data has not been formally analyzed, herein we illustrate the type of information that may be obtained from the sensors by presenting data obtained from 1 female fisher. Based on preliminary examination, we believe that temperature readings for an animal in a den during winter are typically above 80°F (see Figure 2 (top) as an example), noting that the temperature in this case is a result of the thermal influence of both the den and the animal's body. As an animal leaves a den, ambient temperature has an increasing influence on temperature readings, while the animal's body temperature has a decreasing influence (though always has some influence). For purposes of discussion here, we assumed the animal was using a rest site if temperature readings were above 60°F. Using this criterion, female fisher F09-370 appeared to quickly return to a den site after capture/release, and remained there, with 1 brief exception, for 3 days (Figure 2 - middle). From Jan. 2 – March 31, we estimate that she spent 72% of her time at a den/rest site (81%, 69%, and 66% during January, February, and March, respectively).

While the temperature threshold for distinguishing den/rest site use versus activity will change, or perhaps not be distinguishable, as ambient temperature increases, we believe our preliminary thresholds are still applicable for much of April in northern Minnesota. Examination of data for female fisher F09-370 during the first 18 days of April suggests that her behavior changed significantly around April 9 (Figure 2- bottom). We first confirmed the presence of kits in her natal den on May 4, and suspected them to be approximately 3 weeks old. We believe the temperature sensor data suggests that she gave birth on or around April 10 when her behavior appears to have changed significantly (Figure 2 – bottom). If our above interpretation and preliminary temperature threshold is correct, she spent substantially more time *away* from a den post-birth than during winter (72% of her time in a den/rest site during winter, only 27% of the time in a den/rest site in the 8 days following birth). Data suggests that she returned to a den for brief intervals, and consistently around 3:30 am each morning, presumably to nurse.

Her extended time away from the den during this period may reflect the energetic demands of lactation and the need to hunt for food more often. Fisher are also known to breed in the first week or two following birth. Her increased activity outside a den/rest site may explain the apparent increase in predation mortality of nursing females following birth (no females died during winter, 4 were killed in the first 6 weeks post-birth).

Home range analysis has yet to be conducted. However, visual examination of the data suggests that marten home ranges may be from 1–3 mi^2, while fisher home ranges may be from 3-5 mi^2. The longest distance a fisher has dispersed from its capture location is approximately 18 miles, and several fishers have moved around 8 miles. One marten moved approximately 8 miles from its capture location, spent nearly a year in a new area, and recently returned close to its original location before being killed by a mammalian carnivore. Another marten dispersed approximately 5 miles before its signal was lost.

Because only 1 year of prey survey data has been collected, limited inference can be drawn. Based on the first year (2008) of surveys, hare abundance in both study areas was highest in regenerating forest, followed by lowland conifer and upland conifer. Hares were least abundant in mature upland deciduous forest. Pooling cover types, average hare pellet counts were 2 times higher in the fisher study area (Area 2, Figure 1) than the marten study area (Area 1, Figure 1), though confidence intervals overlap. Conversely, the small mammal capture rate (cover types pooled), dominated in both areas by red-backed voles (*Myodes gapperi*) and *Peromyscus* spp., was nearly 3 times higher in the marten study area (Area 1, Figure 1) than in the fisher study area (Area 2, Figure 1), solely a result of lower red-backed vole abundance in Area 2. More details on prey sampling methods and results will be presented in future years.

FUTURE PLANS

Throughout early summer 2009, we will be continuing to confirm natal or maternal dens of marten, and will attempt to monitor survival of any confirmed marten litters, as well as for litters of fisher kits already confirmed. We will also begin collection of vegetative information from individual fisher and marten home ranges. Sampling will occur in a pre-determined number of 0.04-acre circular plots within each home range. We will collect quantitative data on: (1) tree DBH and height, and ultimately basal area and volume of trees, by species; (2) % canopy cover (deciduous and coniferous); (3) sapling density; (4) understory cover density; (5) density and volume of snags; (6) density, volume, and other characteristics of coarse woody debris; and (7) density and volume of exposed root masses.

Tissue samples from prey species, as well as hair samples from fisher and marten, will be sent to a lab for stable isotope analysis. If species-specific chemical signatures of prey are sufficiently distinct, we will further assess the potential for describing late-summer/fall food habits of fisher and marten based on chemical analysis of their guard hair.

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Table 1. Reproductive data for radiocollared female fisher in Minnesota, 2009.

	Pregnancy		Litter info		
ID	Status	Method	# kits	Method	Natal den
F08-353	Y	Kits	3	Video	Tree cavity; 23” dbh live aspen
F08-375	Y	Kits	3	Video / uterine scars	Tree cavity; 22” dbh w.pine snag
F09-360	Y	Progesterone/Kits	2	Video	Tree cavity; 15” dbh aspen snag
F09-362	N	Behavior	0	--	--
F09-364	N	Progesterone	0	--	--
F09-376	N	Progesterone	0	--	--
F09-380	Y	Progesterone/Kits	3	Video	Tree cavity; 24” dbh aspen snag
F09-390	N	Progesterone	0	--	--
F09-394	Y	Kits	3	Video	Tree cavity; 22” dbh live aspen
F08-077	Y	Nursing	4	Uterine swellings / scars	Died before den located
F08-304	Y	Nursing	2	Uterine swellings / scars	Died before den located
F09-354	N	Behavior	0	--	--
F09-370	Y	Kits	3	Video	Tree cavity; 16” dbh aspen snag

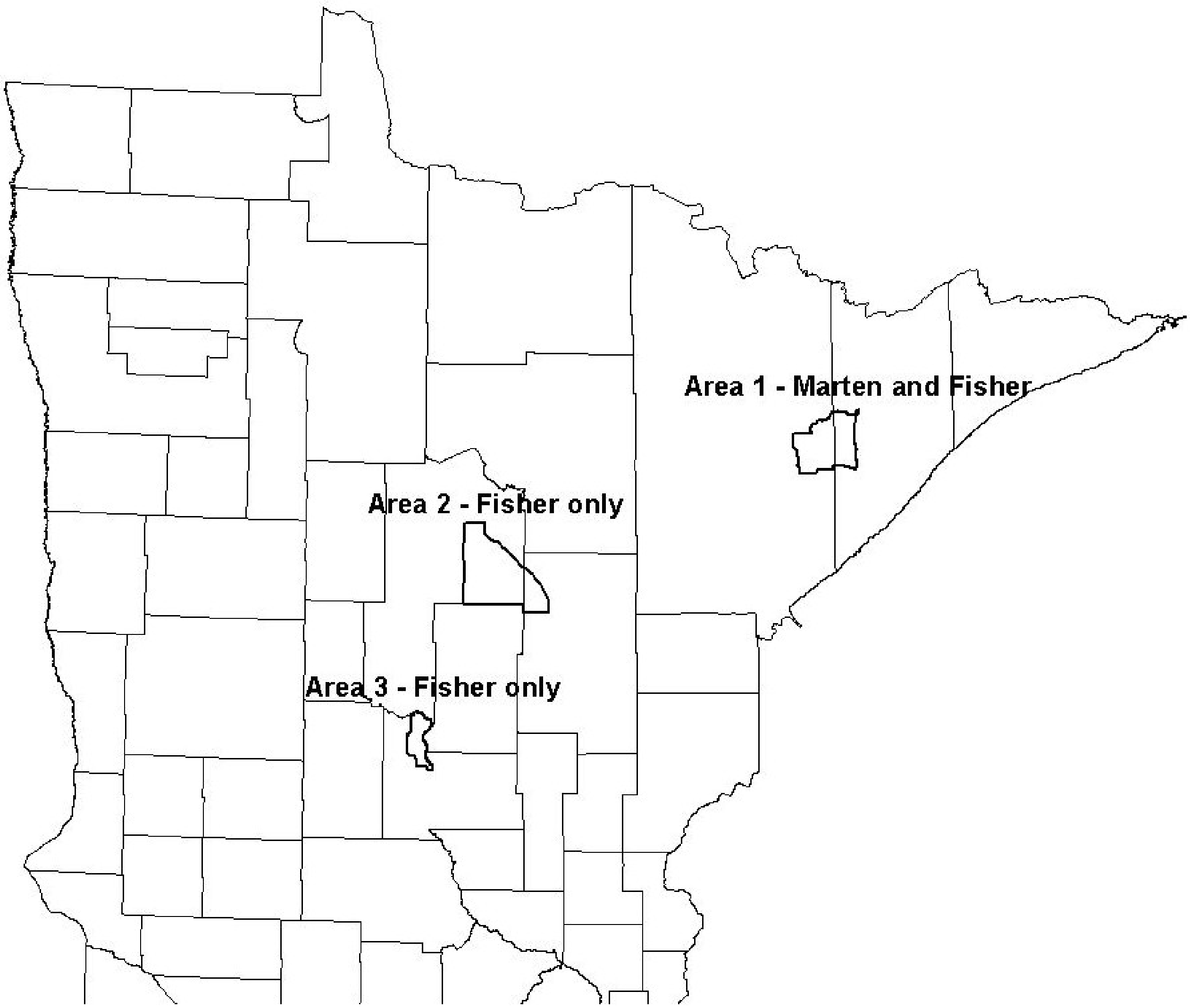

Figure 1. Fisher and marten study areas in Minnesota, 2008-09.

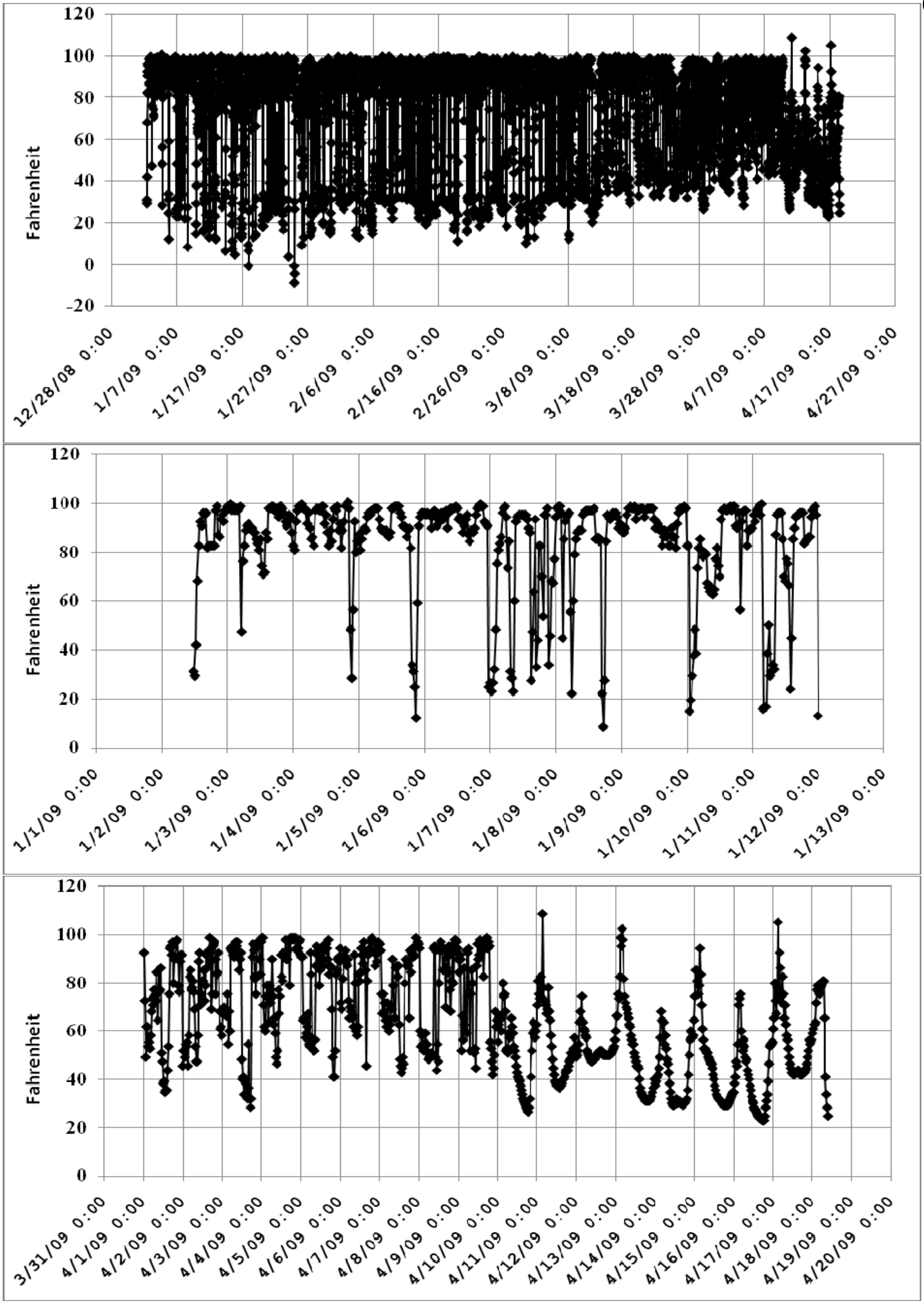

Figure 2. Temperature readings every 30 minutes from the radiocollar of a female fisher in Minnesota (Top: all readings from Jan. 2, 2009 – April 18, 2009; Middle: Readings for the first 10 days following release; Bottom: readings for the first 18 days of April)

DYNAMICS OF A MINNESOTA MOOSE POPULATION IN A WARMING CLIMATE[1]

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

SUMMARY

1. Species on the southern edge of their distribution are especially at risk to climate-induced changes. One such species is the moose (*Alces alces*), whose continental United States distribution is restricted to northern states or northern portions of the Rocky Mountains cordillera. Moose are particularly vulnerable to climate change because of their intolerance to heat. Here, we examine the demographic implications of estimated survival and reproduction schedules for a moose population in northeastern Minnesota, USA, between 2002 and 2007.

2. Estimated age- and year-specific survival rates showed a sinusoidal temporal pattern during the course of the study and were lower for younger and old aged animals. Estimates of annual adult survival (constant across age classes) ranged from 0.74 – 0.85. Annual calf survival averaged 0.40 and the annual ratio of calves born to radiocollared females averaged 0.78.

4. Point estimates for λ from yearly matrices ranged from 0.71 to 0.97 during the 6-year study, indicative of a long-term declining population. Assuming each matrix to be equally likely to occur in the future, we estimated a long-term stochastic growth rate of 0.85. Population growth rate, and its uncertainty, was most sensitive to changes in estimated survival rates.

5. Maximum daily temperature at Ely, Minnesota, USA increased between 1960 and 2007 and displayed several short-term fluctuations. The telemetry data for this study were collected during a period with the highest maximum temperature values.

6. If heat stress is responsible for current levels of survival and temperatures continue to increase, survival rates are likely to decline even further and ultimately the southern edge of moose distribution will shift northward into Ontario.

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TEMPERATURE RELATIONSHIPS IN MOOSE HABITAT: A PILOT

Mark S. Lenarz

SUMMARY OF FINDINGS

I deployed Hobo data loggers at 10 weather stations in northeastern Minnesota to determine the magnitude of variation in temperature metrics in various cover types used by moose (*Alces alces*). Maximum ambient temperature and dew point temperature varied little among stations. Maximum black globe temperature displayed more variation, presumably in response to differences in canopy closure at the weather stations. Soil moisture apparently affected variability in soil temperatures. Heat stress indices calculated from these metrics displayed little variation among stations.

INTRODUCTION

Heat stress as experienced by moose is dependent on their immediate environment. Because of their dark color, moose are likely to absorb more solar radiation than light colored animals and more likely to benefit from using habitat as a thermal refuge. Ambient air temperature as used by Lenarz et al. (2009) as a measure of heat stress, explained variation in seasonal and annual moose survival but solar radiation and humidity may be even more important to the level of stress experienced by moose. Research into the effects of heat stress on humans and domestic cattle have utilized 2 different indices for heat stress. The Wet Bulb Globe Temperature (WBGT) is a composite of ambient air temperature, solar radiation, and humidity that was originally developed to determine appropriate exposure levels for humans (U. S. Army 2003). A similar measure, the Temperature Humidity Index (THI) was developed to determine the effect of summer conditions on human discomfort (AMS 2000).

The primary objective of this research was to determine the magnitude of variation in temperature metrics used to calculate WBGT and THI across a sample of habitats known to be used by moose. I also wanted to evaluate the limitations of a variety of data logging equipment that would be needed in any future research evaluating the thermal value of moose habitat.

METHODS

To calculate WBGT , 3 metrics are required: the ambient air temperature in °C (T_a), the dew point temperature in °C (T_d), and the black globe temperature in °C (T_g).

WBGT is derived from the formula: $WBGT = 0.7T_d+0.2T_g+0.1T_a$.

THI is derived from the formula: $THI = 0.55T_a+0.2T_d+5.3$.

At each station, I used a Hobo data logger (Onset, U23-001, Bourne, MA) enclosed in a thermometer shelter (Ben Meadows, Janesville, WI) to record T_a and T_d. The shelter was located approximately 1 m above the ground facing north. I used a Hobo (U23-003) enclosed within the shelter with 2 external probes, one of which was inserted into a 6” copper globe (Naugatuck Manufacturing, Waterbury, CT) painted flat black to measure T_g. The globe was located approximately 1 m from the shelter and 0.5 m above the ground. The second probe of this data logger was buried approximately 15 cm in the ground to measure soil temperature (T_s). I also monitored light intensity (Lux/m^2) as a covariate using a pendant Hobo UA-002064 that was hung on a hook on the north side of the shelter. Measurements were recorded every 30 minutes.

RESULTS AND DISCUSSION

I deployed 6 weather stations on 7 Aug 2008 and 4 stations on 22 Aug 2008 and removed all stations on 12 Sep 2008. Weather stations were located in LandSat-Based Land Use Land Cover (Manitoba Remote Sensing Centre) cover types designated as Conifer Forest, Wetland Bog, Regenerated Young Forest, and Mixed Forest. I attempted to get a range in canopy closure across the locations. The most distant stations were separated by approximately 13 km.

The Hobo data loggers within the thermometer shelters recorded T_a and T_d without any problems. At 2 stations, however, snowshoe hare (*Lepus americanus*) or deer (*Odocoileus virginianus*) chewed through the cables connected to the external probes and all data for T_g and T_s were limited. In addition, the external probe fell out of 1 globe and all subsequent data on T_g were lost. Analyses involving T_s and T_g were restricted to the 7 functioning stations. All pendant data loggers were effective at recording light intensity. Any future studies using external probes should protect the cables (e.g. copper tubing) to prevent loss of data.

Maximum ambient temperature (T_a) and dew point (T_d) varied little among the 10 stations. Restricting data to the period 22 Aug to 12 Sep (when all stations were operational), the mean maximum daily T_a ranged from 18.2 – 20.7 °C ($\bar{x}$ = 19.4, SE = 0.3, n = 10) and T_d ranged from 12.3 – 13.1 °C ($\bar{x}$ = 12.7, SE = 0.1, n = 10). In contrast, mean maximum T_g ranged from 19.5 – 29.1 °C ($\bar{x}$ = 25.3, SE = 1.4, n = 7) which likely reflects differences in canopy closure among stations. Soil temperature (T_s) was bimodal. Several stations were consistently warmer and tended to be correlated to T_a (e.g. station 1, r^2 = 0.816). Other stations were 3 to 4ºC cooler with little daily variation and were not correlated with T_a (e.g. station 4, r^2 = 0.293). I suspect that the latter stations had water-saturated soils. Temperatures at the more variable stations declined and stabilized after 2 significant rain events. The amount of shading (total lux/m^2/day) varied considerably among stations with an 11 fold difference between station 1 (29,600 lux/m^2/day) and station 3 (2,600 lux/m^2/day).

Despite substantial differences in the degree of shading, there was little variation in WBGT among stations. Mean maximum daily WBGT ranged from 13.6 – 15.9 ($\bar{x}$ = 15.0, SE = 0.3, n = 7). Daily maximum values of WBGT ranged as high as 24.3 during the period 7 Aug to 12 Sep 2009. Considering that T_g represents only 20% of WBGT, the lack of variation among stations is not surprising.

The Temperature Humidity Index (THI) incorporates only T_a and T_d, neither of which varied substantially among stations. Maximum daily THI ranged from 23.8 to 25.3 ($\bar{x}$ = 24.5, SE = 0.2, n = 10).

The difference between black globe temperature (T_g) and ambient air temperature (T_a) is a measure of the increased energy that a moose would absorb as a result of radiant energy (mostly solar). The maximum daily value for this variable was a linear function of the total amount of light at the site (Figure 1). This relationship, however, varied according to the site. At 3 stations (1, 8, and 9) where the total daily light (lux/m^2/day) averaged 12,887 (1,637-29,600), the slope of the regression line was considerably higher (6.7E-4, r^2 = 0.868) while at 2 stations (2 and 4) where total daily light averaged 76,800 (5,100-137,900) the slope was 1.0E-4 (r^2 = 0.935). Most likely, the increased shrub layer at stations 1, 8, and 9 was radiating additional energy as measured in T_g.

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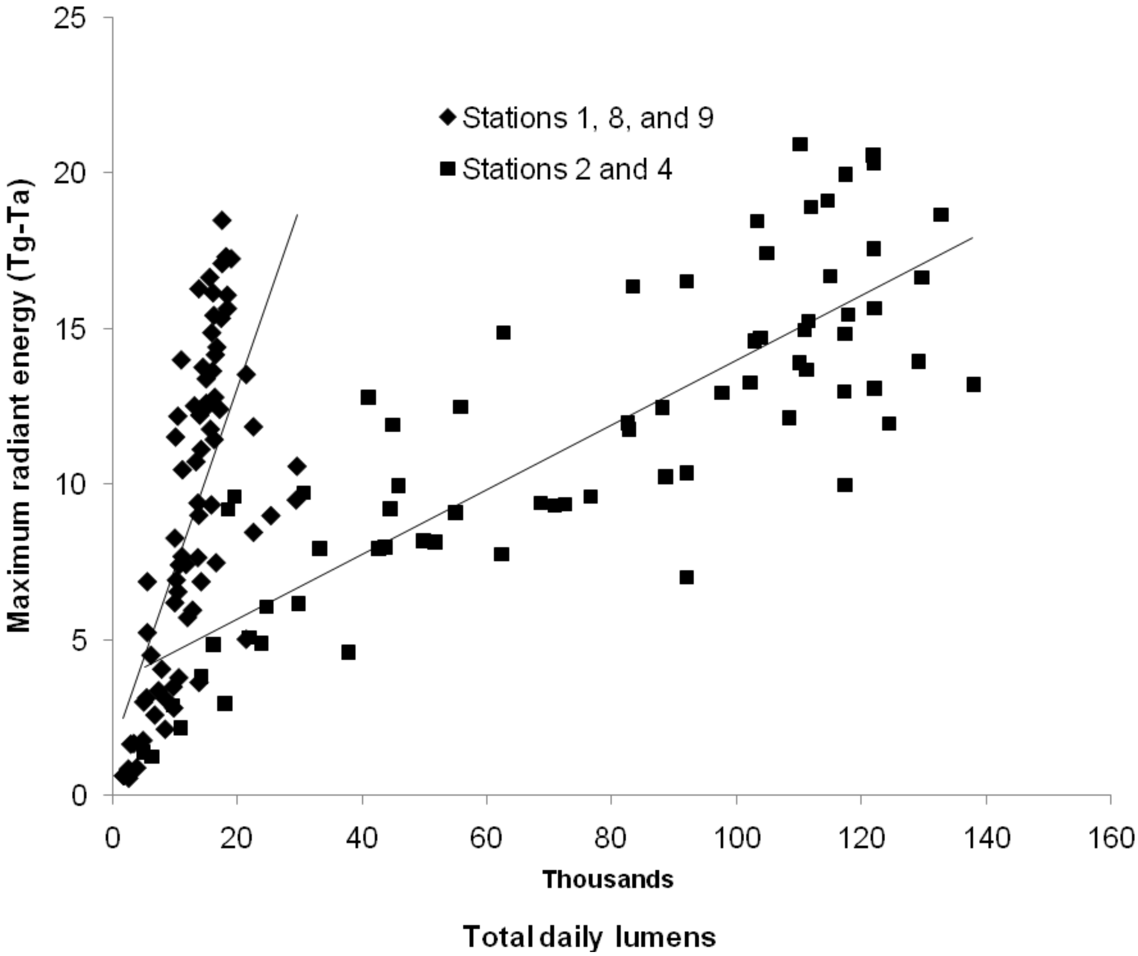

Figure 1. Maximum radiant energy as a function of light as measured at 5 weather stations in northeastern Minnesota, USA. At 3 stations (1, 8, and 9) the slope of the regression line was considerably higher (6.7E-4, $r^2 = 0.868$) than at the other 2 stations (2 and 4) where the slope was 1.0E-4 ($r^2 = 0.935$). Most likely, the increased shrub layer at stations 1, 8, and 9 was radiating additional energy as measured in T_g.

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

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

ABSTRACT

Because winter nutrition of moose (*Alces alces*) and other northern ungulates has been strongly associated with mortality and reproduction, assessments of the condition of individuals may be particularly informative to understanding the dynamics of populations and other influential factors. During February–March 2003 to 2005, we assessed the nutritional condition of 79 moose (39 females, 40 males) in a declining population in northeastern Minnesota by ultrasonographic measurement of rump fat (Maxfat) and 2 body condition scoring (BCS) systems (whole body and rump-only). Our objective was to compare the 2 BCS techniques, relate them to a more quantitative measure of condition, and determine if condition was a contributing factor to non-anthropogenic mortality documented in a companion study and to pregnancy status of these moose. Scores of the 2 BCS systems were correlated ($r = 0.81$; $P < 0.0001$), and each was related to Maxfat ($r^2 = 0.34–0.35$) and ingesta-free body fat (IFBF; $r^2 = 0.37–0.41$), estimated from Maxfat. Body condition scores of males were less ($P \leq 0.009$) than those of females, and there was a significant ($P = 0.021$) sex × capture-year effect on Maxfat (and IFBF) with no effect of age. Mean estimated IFBF was 9.9% (± 0.5 [SE], range = 2.5–15.0%) for females and 8.8% (± 0.3, range = 6.2–11.4%) for males. During winter 2003, when the pregnancy rate was 55%, mean IFBF of females was 24–25% less than during 2004 and 2005 when all females were pregnant. During winters 2004 and 2005, mean Maxfat values, indicative of 7.8–11.5% IFBF, were consistent with winter and spring survival rates (≥95%). However, over all 3 winters, IFBF (< 5%) of 15–21% of these moose was indicative of a more compromised probability of subsequent survival. We did not observe a direct relationship between winter condition and non-anthropogenic mortality of these moose, but collective evidence suggests that heat stress, implicated in reported relations between January, spring, and "warm-season" temperatures and non-winter and annual survival of these moose, may have imposed an additive and cumulative adverse effect on their condition not detected by our sampling. Specific relations among nutritional limits, seasonal heat stress, and the use of thermal refuges by moose require more comprehensive and in-depth investigation.

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MOOSE POPULATION DYNAMICS IN NORTHEASTERN MINNESOTA

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SUMMARY OF FINDINGS

We captured and radiocollared a total of 150 adult moose (*Alces alces*; 55 adult males and 95 adult females) between 2002 and 2008. As of 1 April 2009, 100 collared moose (46 adult males and 54 adult females) have died. Annual mortality rates varied among years, and generally were higher than found elsewhere in North America. Estimates of fertility for this population were also low compared with other North American moose populations. Data analyses from this research are progressing and 1 manuscript was published and 2 other manuscripts were submitted for publication.

INTRODUCTION

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

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

METHODS

We captured a total of 116 moose in southern Lake County and southwestern Cook County between 2002 and 2005, attached radiocollars, and collected blood, hair, fecal and tooth samples. See Lenarz et al. (2009) for greater detail on the study area and research methods. In February 2008, we captured and sampled an additional 34 moose and attached radiocollars. We monitored a sample of up to 78 radiocollared moose weekly to determine when mortality occurred. We calculated annual non-hunting mortality rates (1 – survival) using the Kaplan-Meier procedure (Kaplan and Meier 1958) modified for a staggered-entry design (Pollock et al. 1989) and censored all moose killed by hunters, those that died from capture mortality, moose that had emigrated from the study area, and apparent transmitter failure. We used a Cox Proportional Hazard (CPH) model (Cox 1972, SAS PROC PHREG, SAS Institute 2008) to test for a difference in annual survival between sexes. Beginning in 2004, we used helicopter surveys in late May – early June (MJ) to estimate fertility of radiocollared females and additional surveys in late April – early May (AM) to estimate survival of calves born the previous spring.

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RESULTS AND DISCUSSION

As of 1 April 2009, 100 of the 150-radiocollared moose (46 adult males and 54 adult females) have died. In addition, 1 moose slipped its collar, 1 moose moved out of the study area, and we lost contact (apparent transmitter failure) with 4 moose. Moose that died within 2 weeks of capture (6) were designated as capture mortality. Hunters killed 16 moose, 2 were poached, and 11 were killed in collisions with vehicles (cars, trucks, or trains). The remaining mortality (65) was considered to be non-anthropogenic and causes included wolf predation (6), bacterial meningitis (1), or unknown (58).

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

Annual non-hunting mortality rates (1 June to 31 May) for adult moose averaged 21% for males (7 to 40%, SE = 5, *n* = 6) and 21% for females (5 to 30%, SE = 4, *n* = 6; Table 1). Sex did not contribute to the prediction of survival ($\chi^2 = 0.001$, $P = 0.98$), which implies that there was no difference in survival rates (non-hunting) between adult male and female moose. Non-hunting mortality was substantially higher than documented for populations outside of Minnesota (generally 8 to 12%; Ballard, 1991, Bangs 1989, Bertram and Vivion 2002, Kufeld and Bowden 1996, Larsen et al. 1989, Mytton and Keith 1981, Peterson 1977) and similar to that observed for adult moose in northwestern Minnesota (Murray et al. 2006).

Serum samples from 91 radiocollared adult female moose were collected and analyzed using radioimmunoassay for levels of serum progesterone between 2002 and 2008. Using a pregnancy threshold of 2.0 ng/ml progesterone, annual pregnancy rate varied from 55 to 100% ($\bar{x}$ = 80%, SE = 8, *n* = 5). Boer (1992), in his review of moose reproduction in North America found that adult pregnancy rate across North America averaged 84%. Although pregnancy rate of yearling moose is reduced (Schwartz 1968), our sample included only 1 yearling moose. Our estimates may be biased low because 4 cows that tested negative in 2003 (55% pregnancy rate) were subsequently observed with a calf.

Between 2004 and 2008, 161 radiocollared adult females gave birth to a minimum of 131 calves (76 singles, 26 twins, and 1 set of triplets; Schrage unpublished). The annual ratio of calves: radiocollared females ranged from 0.53 to 0.96 ($\bar{x}$ = 0.79, SE = 0.07, *n* = 5). These estimates were biased low because in 3 of 4 years, radiocollared females not accompanied by calves during the MJ survey were subsequently observed to be accompanied by a single calf (4 in 2004, 2 in 2005, 1 in 2007). It is also possible that post natal mortality occurred prior to the MJ survey. Nonetheless, these estimates are low compared with other locations in North America. Boer (1992), for example, reported estimates ranging from 0.88 to 1.24 calves/adult female, in moose populations above and below carrying capacity, respectively.

During the past year, 3 manuscripts discussing the results of this research have been prepared for publication. The first, entitled “Temperature meditated moose survival in northeastern Minnesota” was published in the May 2009 issue of the Journal of Wildlife Management. A second manuscript, entitled “Assessing winter body condition of moose (*Alces alces*) in a declining population in northeastern Minnesota” has been submitted to the Journal of Mammalogy. The final manuscript, entitled “Dynamics of a Minnesota moose population in a warming climate” has been submitted to the Journal of Animal Ecology. At least 2 additional manuscripts are planned from the data collected during the first phase of this research.

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

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

[1] Period: 1 June 2002 – 31 May 2003.
[2] Sample size as of 31 May.

A REVIEW OF THE ECOLOGY OF *PARELAPHOSTRONGYLUS TENUIS* IN RELATION TO DEER AND MOOSE IN NORTH AMERICA

Mark S. Lenarz

SUMMARY OF FINDINGS

It is well established that white-tailed deer (*Odocoileus virginianus*) are the normal host for *P. tenuis* and that this parasite either kills moose (*Alces alces*) directly or predisposes them to other causes of mortality. Despite the historical record of moose dying from this parasite, there is little evidence, that *P. tenuis* is a major cause of mortality in moose or that it was responsible for historic declines in moose populations. When white-tailed deer expanded their range northward into moose range following logging, they undoubtedly introduced *P. tenuis* to moose. While it may seem intuitive that higher deer numbers should translate into higher moose mortality, research has not corroborated this relationship. Rather, it has discovered that the transmission of *P. tenuis* between deer and moose is a complex relationship and after almost 45 years, this relationship is still poorly understood. Based on our current knowledge, reductions in deer density on moose range will likely have little effect on the population status of moose in Minnesota.

INTRODUCTION

As early as 1912, a “moose sickness” was identified in Minnesota moose (Fenstermacher and Olson 1942). The disease was characterized by apparent blindness, lack of fear, aimless wandering, and ataxia (Karns 1967), which either killed moose directly or predisposed them to other causes of mortality. Histopathological analysis of diseased animals revealed irreversible damage to the central nervous system (Kurtz et al. 1966). Although moose sickness was associated with the presence of white-tailed deer as early as the late 1950s (Benson 1958), it was the experimental work by Anderson (1964), who demonstrated that the nematode lungworm (*Parelaphostrongylus tenuis*) caused moose sickness. Since this discovery, several hypotheses have been proposed regarding the relationship between deer, moose, and *P. tenuis*. It is the objective of this review to examine these hypotheses and subsequent research that either corroborate or refute them.

LITERATURE REVIEW

The life cycle of *P. tenuis* normally incorporates a definitive host, white-tailed deer, and an intermediate host, which includes several species of gastropods (slugs and snails). Once a deer is infected, *P. tenuis* larvae develop into adults and live in association with nervous tissue in the spinal cord and in the subdural spaces and venous sinuses of the cranium (Lankester and Samuel 1998). After a complex journey through the deer’s body, first stage larvae are shed in the mucosal coating on feces and may survive as long as 10 months outside the host (Lankester and Anderson 1968). Gastropods that live in the litter on the forest floor crawl over the deer feces and become infected with the first stage larvae. Within the gastropod, the larvae ultimately molt into 3^{rd} stage larvae that are infective to cervids if accidentally ingested (Lankester and Samuel 1998). White-tailed deer apparently do not succumb to the neurologic disorders caused by *P. tenuis* in other cervids (Alibasogulu et al. 1961, Anderson 1963).

Shortly after Anderson identified *P. tenuis* as the cause of moose sickness, several authors hypothesized that *P. tenuis* was a major cause of mortality in moose and responsible for historic declines in moose populations (Karns 1967, Telfer 1967). By the mid-1970s, there was general agreement on this supposition, especially in areas of high deer densities in eastern North America (Gilbert 1974, Prescott 1974, Kearney and Gilbert 1976, Lankester 1987). In the 1980s, however, some scientists began to question this conjecture based on the low number of moose deaths attributed to “moose sickness” at a time when both moose and deer populations were increasing on shared range (Brown 1983, Upshall et al. 1987, Lenarz and Kerr unpubl.). Subsequently, Nudds (1990) questioned the hypothesis that *P. tenuis* was a major cause of mortality in moose and suggested that circular logic was used in making this inference. After reviewing all available data from Maine, Minnesota, New Brunswick, and Nova Scotia, Whitlaw and Lankester (1994a) indicated that the historical information available did not corroborate the hypothesis that *P. tenuis* had caused declines in moose populations.

Recent research in Minnesota also suggests that while present, *P. tenuis* is not a major cause of mortality. In northwestern Minnesota, for example, mortality of only 5% of radiocollared moose was attributed to *P. tenuis* (Murray et al. 2006). In northeastern Minnesota, 17% (18/108) of moose had positive titers for *P. tenuis* (Lenarz et al., unpublished data). Assuming that this parasite was responsible for the subsequent death of moose testing positive (except for 1 capture mortality and 1 hunter kill), annual cause specific mortality from *P. tenuis* averaged 4% (0 to 10%) and represented an average of 19% (0 to 32%) of the total mortality the population experienced each year (Lenarz et al., unpublished data). Considering the relatively low proportion of moose mortality attributed to *P. tenuis* in both northeastern and northwestern Minnesota, it is questionable whether this parasite represents a major threat to the moose populations.

Early researchers also suggested that the infection rate in moose increased as a direct response to increasing deer density (Anderson 1965, Karns 1967, Kelsall and Prescott 1971, Gilbert 1974). These early researchers reasoned that as deer density increased, more deer were infected, and more larvae would be shed. As a consequence of the increased number of larvae, more gastropods would be infected, and the probability that moose would consume an infected gastropod and die would increase. Subsequent research, however, has indicated that the relationships between deer density, *P. tenuis* infection rates, and moose mortality are complex and poorly understood (Anderson and Prestwood 1981, Whitlaw and Lankester 1994a, b; Lankester and Samuel 1998).

The hypothesis that more larvae are shed as deer density increases assumes that the prevalence of *P. tenuis* in deer is constant or increases as deer numbers increase. Based on meager evidence, Karns (1967) and Behrend and Witter (1968) suggested that the prevalence of *P. tenuis* increased as deer numbers increased. Gilbert (1973), however, found a lower prevalence at higher deer density after comparing 2 areas in Maine. Thomas and Dodds (1988) found no relationship between deer infection rates and deer density (2 levels) or moose density (3 levels) in Nova Scotia. Based on deer sampled from 17 Deer Management Districts, Bogaczyk et al. (1993) found that neither prevalence nor intensity of infection in white-tailed deer was associated with deer density over a range of 1.4 to 5.8 deer/km^2. Hence, there are few data to suggest a relationship between prevalence and deer density. It is logical to assume, however, that more deer will deposit more feces on the landscape and unless prevalence declines in response to deer density, there will be more infected feces.

Even if the density of infected feces is high, it doesn't imply that higher numbers of gastropods are infected. Lankester and Peterson (1996) found a prevalence rate of only 0.16% (7 out of 4,401) in a deeryard that seasonally supported 50 deer/km^2. Other surveys in Minnesota and Ontario have generally found a prevalence rate less than 1% (Lankester 1967, Kearney and Gilbert 1978, Pitt and Jordan 1995, Lankester and Peterson 1996). Research that reported both infection rates of gastropods and local deer density is limited (Lankester and Anderson 1968, Maze and Johnstone 1986, Platt 1989, Lankester and Peterson 1989, Pitt and Jordan 1994). Pooling these data, there was no correlation (r = 0.09, *P* = 0.86) between prevalence and deer density. The infection rate of gastropods is likely dependent on the density, residence time, and defecation rates of infected deer; the survivorship of first-stage larvae on the feces or in the soil; and the abundance of and mobility of suitable gastropods (Lankester and Peterson 1996). Even in a situation with 120 deer living year around on a 1.3 km^2 island (240 deer/mi^2), the prevalence of *P. tenuis* in gastropods was only 4.2% (Lankester and Anderson 1968).

Considering the extremely low infection rate of gastropods, it is unclear how large numbers of deer become infected. Based on the prevalence documented by Lankester and Peterson (1996) in northeastern Minnesota and assuming that a deer or moose could be infected by consuming a single infected gastropod, each deer or moose on summer range would need to consume 2,500 gastropods to become infected. Anderson and Prestwood (1981) proposed that infected gastropods might live in small concentrations that are not adequately sampled by researchers but encountered by foraging cervids. In a study of gastropod climbing behavior, McCoy and Nudds (1997) found that species were highly variable in the degree that they climbed; some species climbed infrequently while other were primarily arboreal. They suggested that data from studies which restricted sampling to the use of damp cardboard (e.g. Gleich and Gilbert 1976, Kearney and Gilbert 1978, Upshall et al. 1986, Lankester and Peterson 1996), resulted in estimates biased to the less arboreal species, which are more likely to encounter *P. tenuis* larvae. If correct, prevalence rates in gastropods would be even lower than currently estimated.

Finally, if moose mortality was a simple function of deer density, there should be an inverse correlation between changes in deer and moose density. In the 1980s and 1990s, several authors documented simultaneous increases in sympatric moose and deer populations in some eastern states and provinces (Clark and Boyer 1986, Upshall et al. 1987, Thomas and Dodds 1988, Boer 1992, Bogaczyk et al. 1993). Several hypotheses were proposed to explain this conundrum. Working in Maine, Clark and Bowyer (1986) found a high prevalence of *P. tenuis* larvae in moose feces and suggested that co-evolution favoring a reduction in the debilitating effect of *P. tenuis* upon moose may have occurred. McCullough and Pollard (1993), however, suggested that faulty lab procedures might have been responsible for the high prevalence of *P. tenuis* in moose feces found by Clark and Bowyer (1986). Upshall et al. (1987) found no larvae in New Brunswick moose and suggested that moose were feeding in different areas than deer, an argument first proposed by Telfer (1967) and subsequently challenged by Nudds (1990).

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ANDEAN BEAR DENSITY AND ABUNDANCE ESTIMATES — HOW RELIABLE AND USEFUL ARE THEY? [1]

David L. Garshelis

ABSTRACT

Estimates of bear abundance and density are difficult to obtain, for a number of reasons. Reliable estimates have been obtained for several populations of American black bears (*Ursus americanus*), brown bears (*U. arctos*), and polar bears (*U. maritimus*), and one population of sloth bears (*Melursus ursinus*). Efforts have been made to obtain estimates of the other bear species, but often with unresolved problems that are disregarded. Very few attempts have been made to estimate numbers and densities of Andean bears (*Tremarctos ornatus*). One very crude rangewide estimate, from 10 years ago, was made by applying the average density of American black bears (obtained from a compilation of many studies, mostly in hunted populations) to the range area of Andean bears. The resulting estimate of ~20,000 bears has been widely cited, without understanding how the value was derived. A more recent estimate (2003), based on the extent of genetic heterozygosity, and assuming that Andean bear numbers have remained stable for thousands of years, yielded a wide span of values (24,000–90,000); nevertheless, the closeness of the low value to the former estimate based on black bears seemed to strengthen the view that there are at least 20,000 Andean bears across the range. Three estimates in smaller study sites produced widely varying and contradictory estimates. At one site in Bolivia, a density estimate based on photographs of 3 bears at remote camera traps (4.4–6 bears/100km^2) was one-half to one-third that of an earlier study in the same site, based on home ranges of 2 radiocollared bears (~12 bears/100 km^2). Neither is reliable because of the small sample sizes, but both suggest that the original rangewide density, derived from black bears (25 bears/100 km^2) could be an overestimate. Likewise, a DNA mark-recapture study in Ecuador also obtained a low density estimate (3–7 bears /100 km^2). In this case, sample size was adequate, but there were strong indications that closure was violated, especially for males (twice as many males as females were detected), indicating that the density could be even lower. These issues are not simply incorporated into confidence intervals: CIs only include error due to sampling, not study design flaws; CIs around biased estimates may not include the true population number. I suggest that more work of a rigorous nature needs to be conducted if density estimates are to be useful in conservation and management.

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ECOLOGY AND POPULATION DYNAMICS OF BLACK BEARS IN MINNESOTA

David L. Garshelis and Karen V. Noyce

SUMMARY OF FINDINGS

During April 2008–March 2009, we monitored 40 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), and a new site at the northwestern (NW) edge of the range. Hunting was the primary source (78%) of mortality in all areas, even though hunters were asked not to shoot radiocollared bears and bears cannot be legally hunted in 2 of the areas (but can be hunted when they wander outside). Reproduction was highest at the fringe of the bear range, at Camp Ripley (near the southern edge) and the NW site (western edge), due largely to an abundance of oaks and hazelnut in these areas, as well as agricultural crops consumed by NW and some Camp Ripley bears in the late summer–fall. Data from GPS-radiocollars indicated that males had home ranges of ~1000 km^2 (~400 mi^2), 18x larger than those of females, and the largest yet measured for black bears across North America. In this highly-fragmented landscape, males travelled large distances, particularly in late summer to locate oaks and agricultural crops; they spent a third of their time in cropfields during September. In contrast, only 2 of 7 GPS-collared females visited cropfields. Although 15 bears had GPS-radiocollars in 2008, 7 of these failed, so both the bears and data were lost. A different brand of GPS-collar will be used in the coming year.

INTRODUCTION

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

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

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

OBJECTIVES

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

METHODS

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

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

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

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

We plotted GPS locations downloaded from collars on bears in the NW study site. We calculated home range areas as 100% minimum convex polygons (MCP) and 100% fixed kernels. We recognize that there is a recent trend toward using 95% MCPs or kernels, but in examining the data, we found that these estimators excluded a large number of points, which in this fragmented landscape accounted for a sizeable travel area.

We used a GIS overlay (Minnesota Land Cover – NLCD 2001) to categorize the covertypes of GPS locations, and then grouped these into 6 broad categories. We calculated percent use of these types by month for each bear, then obtained monthly averages among bears. We examined temporal trends in use of forests (deciduous, evergreen, and mixed types combined), wetlands (woody wetlands and herbaceous wetlands combined), and agricultural areas. Bears rarely used the other 3 categories, so they are not examined further here.

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

RESULTS AND DISCUSSION

Radio collaring and Monitoring

Since 1981 we have handled >800 individual bears and radiocollared >500. As of April 2008, the start of the current year's work, we were monitoring 28 collared bears: 5 in the CNF, 10 at Camp Ripley, 4 in VNP, and 9 in the new NW study site. Two bears at Camp Ripley and 7 in the NW had GPS-collars. The small sample sizes in the CNF and VNP were due to high harvests of radiocollared bears during the past several years.

We captured more bears in the NW study site during the month of June, concentrating in the far northwestern corner of the state. We captured 12 new bears (8 males, 4 females), and collared them, 6 with VHF radiocollars and 6 with GPS-collars.

During fall telemetry flights in the NW, 7 GPS-collars (all manufactured by ATS, Isanti, MN) could not be located; we assume that they failed.

Mortality

Legal hunting has been the predominant cause of mortality among radiocollared bears from all study sites; 78% of mortalities that we observed were due, or likely due to hunting (Table 1). In earlier years of this study, hunters were encouraged to treat collared bears as they would any other bear so that the mortality rate of collared bears would be representative of the population at large. With fewer collared bears left in the study, and the focus now primarily on reproduction and habitat use rather than mortality, we sought to protect the remaining sample of bears. We asked hunters not to shoot radiocollared bears, and we fitted these bears with bright orange collars so hunters could more easily see them. However, the mortality rate on collared bears has remained high because many hunters indicated that they could not see the collars in dim light conditions. Some hunters, though, saw collars on bears photographed by camera traps at their bait sites, and thereby avoided killing the bear.

This year (September–October 2008), hunters killed 1 of 5 CNF bears (unfortunately, a 23-year old individual whom we had monitored through 10 reproductive cycles and had hoped to track into her reproductive senescence) and 1 of 4 VNP bears (killed outside the park), but no bears at Camp Ripley and only 1 of 21 in the NW (another was killed after it lost its collar). The lower rate of loss in the NW site may have been due to more widespread publicity of this project in that area. It is also possible that some of the GPS-collared bears that could not be located during aerial searches of this area were killed by hunters, although normally hunters have returned collars, and in the rare cases where they have not, we have been able to locate collars left in the woods.

One other mortality occurred this year, a Camp Ripley bear that was hit by a vehicle outside the Camp. Camp Ripley has had the highest rate of vehicle-caused bear mortality, with 8 of 27 deaths (30%) attributable to this cause; in CNF, <5% of deaths were due to collisions with vehicles (Table 1). Statewide, only 27 car-killed bears were reported in 2008, but this is certainly an underestimate of the number actually killed.

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

Natural mortality is a relatively minor cause of death among Minnesota bears >1 year old. Natural mortalities were most common in VNP (Table 1). Now, however, with a small remaining sample of bears in this area, natural mortalities will be harder to detect. However, all of the remaining bears are unusually small females (2 4-year olds weighed only 69 and 99 pounds in March, 2009), so may be susceptible to undernutrition or cannibalism.

Reproduction

Of 13 mature female bears checked in dens during late winter, 2009, 5 (38%) had cubs and 8 had yearlings. However, 1 mother with a yearling in the NW remained awake much of the winter, and separated from her yearling during January or February. Additionally, a 35-year-old has been post-senescent since 1999, when she was 25 years old.

Bears at Camp Ripley grow faster and thus have an earlier age of first reproduction than at CNF and VNP. Two years of data from the NW study site suggest that the bears there are also large, with early maturity (e.g., 1 male yearling weighed 131 lbs in the den in early March, the heaviest of this study). Both Camp Ripley and the NW site are at the fringe of Minnesota's bear range, where acorns, hazelnuts (*Corylus americana, C. rostrata*), and agricultural crops are plentiful, accounting for the high growth rates of these bears.

At Camp Ripley, 6 of 7 female bears produced their first litter at age 3; however, only half of these 6 litters survived the first year. Among 4–6 year-old females, the reproductive rate (cubs born/female) was nearly twice as high at Camp Ripley as in VNP (where no bears produced cubs at 4 years old, including 2 monitored this year); the reproductive rate of 4–6 year-olds was intermediate at CNF (Table 2). This gradient was also apparent in the reproductive rates of older bears, due to fewer missed reproductive opportunities in Camp Ripley versus more whole-litter losses and skipped litters at VNP (Table 2). If no bears skipped litters, all would be on a 2-year reproductive cycle, and thus 50% of females would have cubs, on average, per year. In both Camp Ripley and the NW, the proportion of adult females with cubs exceeded 50% due to an artifact of sampling (Table 2). This proportion was lowest in VNP and intermediate in CNF. Reproductive rates were also most variable, year-to-year, in VNP, and least so at Camp Ripley (Figure 1).

Mean litter size was not appreciably different among sites (2.3–2.6 cubs/litter, Tables 3–5). Data from the NW site produced a higher average litter size, but the sample size is still too small to make a reliable estimate in this area.

Average sex ratio of cubs shortly after birth was slightly, but consistently male-biased (pooled average across all areas = 52% male, n = 594 cubs examined). Observed year-to-year variation in cub sex ratios (Tables 3–6) was likely attributable to sampling error, although it is possible that some real year-to-year variation may occur as a result of varying food conditions.

Cub mortality averaged 18–20% in CNF and Camp Ripley, and somewhat higher in VNP (Tables 3–5). Assessment of cub mortality in the NW was hampered by the failure of several radiocollars on adult females with cubs. Across all areas, the mortality rate of male cubs was significantly higher than (nearly 2x) that of females ($\chi^2 = 10.3$, $P = 0.001$), however, the predominant cause of cub mortality in Minnesota is not known.

Cub production (Figure 1) and cub mortality did not show either an upward or downward trend during our 28 years of monitoring. However, statewide bear harvests have contained an increasing proportion of young bears (Figure 2), suggesting a changing statewide age structure, likely due to increased hunting pressure.

Movements and Habitat Use of NW Bears

Data from GPS-collars in the NW study site indicated that home ranges were especially large. The average male home range in this study was 900–1100 km^2 (350–420 mi^2; depending on method of estimation, Table 7). We compiled results from 32 other studies of black bears across North America, which reported male home ranges of 5–465 km^2 (2–180 mi^2; median = 108 km^2 = 42 mi^2). Female home ranges in this study averaged 60 km^2 (23 mi^2) mi^2) based on MCPs. These were well above average from across North America (median = 25 km^2 = 10 mi^2), but were within the range observed in other studies (2–295 km^2 = 1–114 mi^2). The ratio of M:F home range sizes in our study (18:1 for MCP) is the largest yet reported.

These large home ranges, especially for males, are related to the fragmented habitat. Bears in this area must travel more because of the gaps between patches of suitable habitat. We purposefully included the gaps of cultivated land between forested patches as part of the home range area, as they are clearly relevant to the space use of these bears. Moreover, although kernel estimators are now more commonly used than MCP, we feel that the MCP better represented the scope and scale of the land area necessary to support these bears, if not the area that they actually used. The normal concept of a home range (the area routinely used) does not apply very well in a fragmented, patchy landscape like this, because although large areas between the main habitat patches are rarely used, they stretch out the size of the home range. If only the small patches of actual use were summed (as tends to occur with kernel estimators), it would appear as though bears existed within much smaller areas (although 100% kernels were larger than MCPs for females, Table 7).

Monthly home ranges of males varied seasonally, and were not in accordance with temporal trends previously reported for bears in continuous forested habitats, including our other 3 study sites. The smallest monthly ranges were in June and July, during the breeding season, when males appeared to home in on several estrus females. Their home ranges during these months were 4–7x larger than those of females (Figure 3). Normally, in continuous habitat, males enlarged their ranges during breeding, to encompass more females. Afterward, in the late summer, males in this study tended to go on long excursions, attracted to scattered natural oak stands as well as agricultural crops; thus, their August and September home ranges were greatly expanded. Females generally did not expand their home ranges in late summer (Figure 3). The ratio of M:F monthly home range size peaked in September at 33:1 (excluding November, when most females had denned).

Male habitat use reflected the seasonal expansion of their home ranges. In September, 34% of their locations, on average, were in agricultural lands, presumably corn, oats, or sunflowers (Figure 4). All of the GPS-collared males used some agricultural crops, and they often travelled significant distances to find a cropfield to settle into (Figure 5). Conversely, only 2 of 4 solitary females and 0 of 3 females with cubs used agricultural cropfields in late summer and fall, and those that did so left these fields earlier than the males (Figure 4). It appeared that females only used cropfields that were adjacent to their home range. Most of the females had some agricultural lands adjacent to their home range, but the use of these lands was minimal (Figure 6), so we presume that they did not contain edible crops (we cannot ascertain this from the GIS overlays, but will determine what was grown there from county records and follow-up interviews with farmers).

Both males and females showed a trend of decreasing use of the “forest” category of land cover, and increasing use of “wetlands”, from May–November (Figure 4). We cannot yet speculate on the cause of this trend. We plan to investigate these wetland areas more thoroughly to see what sorts of habitats they really include, and from this we hope to be able to explain their increasing use through the year. This trend was opposite what we expected; in our other study sites, bears used wetlands primarily during spring.

Fruit Sampling

From July 15 to August 20, we sampled 65 stands for soft mast and hazelnut production in the NW study site, including: 36 stands that were predominantly aspen (but varying in age, silvacultural treatment, and ground moisture), 18 oak, 5 balsam poplar (*P. balsamifera)*, and 6 lowland hardwood. We quantified abundance of fruit-producing plants and fruit production of these plants. We also picked representative fruit samples to convert fruit production ratings to biomass estimates. These data have not yet been analyzed, however, it was clear that fruit production in 2008 was much less than in 2007, when juneberry (*Amelanchier* sp.), chokecherry (*Prunus virginiana*), and both species of hazel were especially productive.

Denning

Bears in the NW study site tended to den during November or even December, later by >1 month than CNF bears. Some individuals moved dens, either on their own or due to our disturbance.

Two denning bears (1 large male, 1 female with yearlings) were found by local people in the NW study site, so we collared them. The female had a large wound on her back, probably a nonfatal injury from a bullet or arrow. The male had apparently been flooded in its den during a rare, heavy rain in mid-February, so had moved and denned on the ground in an open field (Figure 7). He remained there for ~6 weeks.

We did not actively monitor when collared bears arose from hibernation, but we checked some in NW and found that by April 1, 7 (4 males, 3 adult females) of 8 bears had vacated their dens. The ground was saturated and underground dens were filled with water (ice). The only collared bear still in its den at that time was a young female whose underground den was not flooded. We did not check the den of the only collared female in the NW with newborn cubs, but she had denned underground in a very low, wet area, so we suspect that she too would have been flooded out. In the CNF during the 1980s, bears typically stayed in dens until at least the first week of April, and those with cubs did not emerge until mid-April.

FUTURE DIRECTION

We plan to continue monitoring bears on these 4 study sites, although sample sizes have been greatly diminished by the exceedingly high harvest of collared bears in the past few years. Our main emphasis in the next few years will be at the new study site in Northwestern Minnesota, where we replaced GPS-collars with those of a different manufacturer. Our goal there is to assess the factors that may limit range expansion, including highly fragmented forested habitat, availability of agricultural crops that bears can eat, and human-related mortality.

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Table 1. Causes of mortality of radiocollared black bears ≥1 year old from the Chippewa National Forest (CNF), Camp Ripley, Voyageurs National Park (VNP), and northwestern (NW) Minnesota, 1981–2008. Bears did not necessarily die in the area where they usually lived (e.g., hunting was not permitted within Camp Ripley or VNP, but bears were killed by hunters when they traveled outside these areas).

	CNF	Camp Ripley	VNP	NW	All combined
Shot by hunter	221	11	15	4	251
Likely shot by hunter[a]	8	1	0	0	9
Shot as nuisance	22	2	1	0	25
Vehicle collision	12	8	1	1	22
Other human-caused death	9	0	0	0	9
Natural mortality	7	3	4	0	14
Died from unknown causes	3	2	0	0	5
Total deaths	**282**	**27**	**21**	**5**	**335**

[a] Lost track of during the hunting season.

Table 2. Reproductive rates (cubs/female), mean litter size, and proportion of females with cubs (in all cases, counting only litters in which at least 1 cub survived 1 year) in winter dens (March) in 4 study sites (ordered from lowest to highest reproductive output): VNP (1997–2009), CNF (1981–2009), Camp Ripley (1991–2009), NW (2008-2009) (*n* = 4+ year-old female-years of observation). Data from the new study site in the northwest are still too sparse to separate by age categories, and the overall values presented here are biased high by the loss of some bears due to radiocollar failure (see text).

	VNP (*n* = 59)			**CNF** (*n* = 407)			**Camp Ripley** (*n* = 51)			**NW** (*n* = 11)		
Age of female	Repro rate	Litter size	Prop w/ cubs	Repro rate	Litter size	Prop w/ cubs	Repro rate	Litter size	Prop w/ cubs	Repro rate	Litte r size	Prop w/ cubs
4–6 yrs	0.53	2.0	0.26	0.84	2.3	0.37	1.04	2.2	0.48			
7–25 yrs	1.15	2.7	0.44	1.33	2.8	0.48	1.58	2.7	0.58			
4–25 yrs	0.93	2.5	0.37	1.15	2.6	0.43	1.29	2.4	0.53	1.75	3.0	0.58

Table 3. Black bear cubs examined in dens of radiocollared mothers in or near the Chippewa National Forest during March, 1982–2009. High hunting mortality of radiocollared bears has reduced the sample size in recent years to the extent that the data are no longer suitable for monitoring.

Year	Litters checked	No. of cubs	Mean cubs/litter	% Male cubs	Mortality after 1 yr[a]
1982	4	12	3.0	67%	25%
1983	7	17	2.4	65%	15%
1984	6	16	2.7	80%	0%
1985	9	22	2.4	38%	31%
1986	11	27	2.5	48%	17%
1987	5	15	3.0	40%	8%
1988	15	37	2.5	65%	10%
1989	9	22	2.4	59%	0%
1990	10	23	2.3	52%	20%
1991	8	20	2.5	45%	25%
1992	10	25	2.5	48%	25%
1993	9	23	2.6	57%	19%
1994	7	17	2.4	41%	29%
1995	13	38	2.9	47%	14%
1996	5	12	2.4	25%	25%
1997	9	27	3.0	48%	23%[b]
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%	
Overall	**174**	**455**	**2.6**	**52%**	**18%**

[a] Cubs that were absent from their mother's den as yearlings were considered dead. Blanks indicate no cubs were born to collared females.
[b] Excluding 1 cub that was killed by a hunter after being translocated away from its mother.

Table 4. Black bear cubs examined in dens of radiocollared mothers in Camp Ripley Military Reserve during March, 1992–2009.

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%	
Overall	**28**	**66**	**2.4**	**50%**	**20%**

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

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

Year	Litters checked	No. of cubs	Mean cubs/litter	% Male cubs	Mortality after 1 yr[a]
1999	5	8	1.6	63%	20%
2000	2	5	2.5	60%	80%
2001	3	4	1.3	50%	75%
2002	0	0	—	—	—
2003	5	13	2.6	54%	8%
2004	0	0	—	—	—
2005	5	13	2.6	46%	20%
2006	1	2	2.0	50%	0%
2007	3	9	3.0	44%	—
Overall	**24**	**54**	**2.3**	**52%**	**28%**

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

Table 6. Black bear cubs examined in dens of radiocollared mothers in Northwestern Minnesota during March, 2007–2009. Loss of 1 large litter in 2007 skews the sample, so overall cub mortality is not calculated (value for 2008 may be more typical).

Year	Litters checked	No. of cubs	Mean cubs/litter	% Male cubs	Mortality after 1 yr[a]
2007	2	6	3.0	33%	100%[b]
2008	5	15	3.0	67%	22%
2009	1	3	3.0	33%	
Overall	**8**	**24**	**3.0**	**52%**	—

[a] Cubs that were absent from their mother's den as yearlings were considered dead.
[b] Only one 5-cub litter was monitored, and all the cubs died (mother produced a litter of 4 cubs the next year).

Table 7. Estimates of annual home ranges of GPS-collared bears in Northwestern Minnesota, 2007 and 2008, based on 100% minimum convex polygons (MCP) and fixed kernels.

		MCP (km^2)			Fixed kernel (km^2)		
Sex	*N*	Mean	Min	Max	Mean	Min	Max
Males	6	1,081	100	2,976	893	160	1,837
Females	7	60	35	107	151	114	208

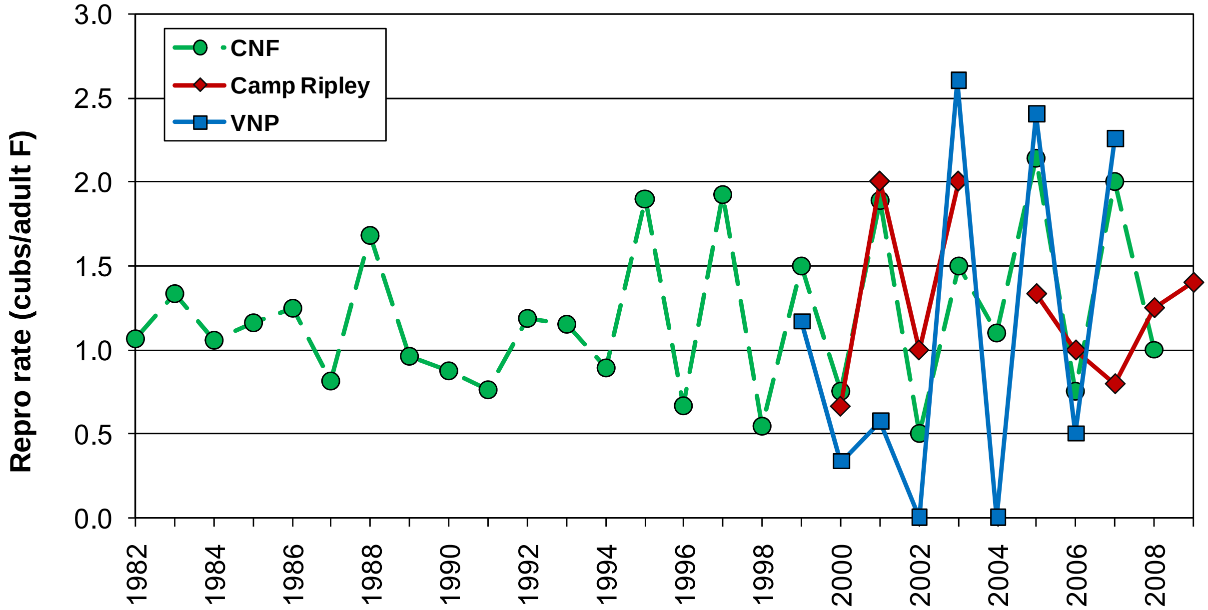

Figure 1. Reproductive rate (measured as [total cubs produced]/[no. of 4+-year-old females monitored]) in each of 3 study sites in Minnesota, 1982–2009. Gaps in data indicate inadequate sample size.

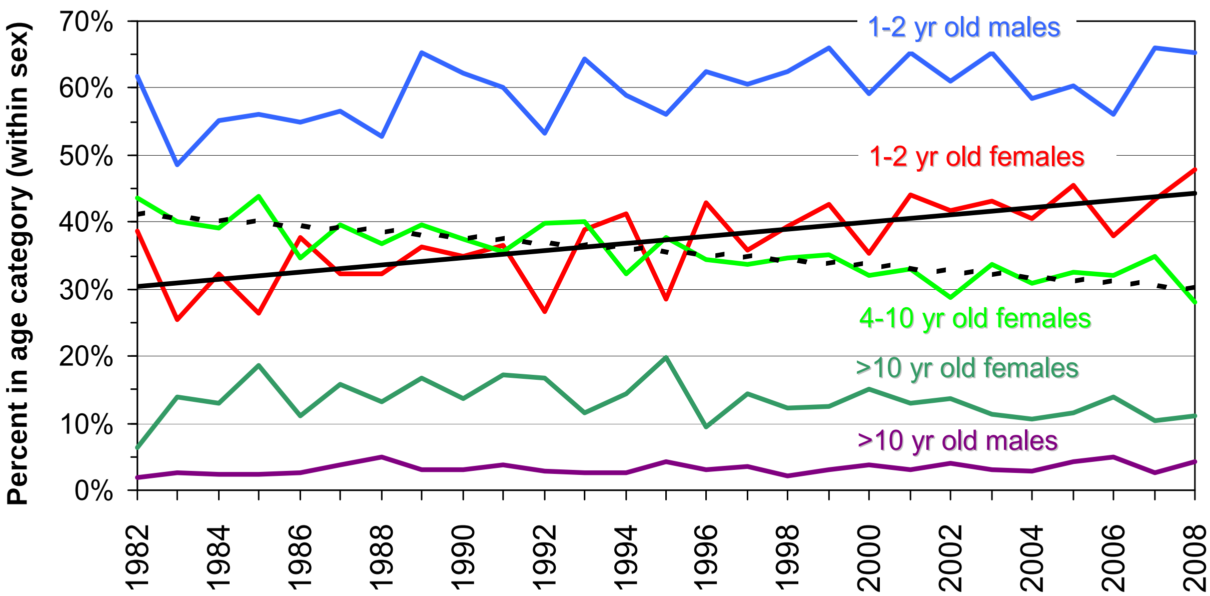

Figure 2. Proportion of harvested bears of each sex in indicated age groupings (3-year-olds excluded for clarity). Increasing trends for 1–2 year-old males and females, and declining trend for 4–10 year-old females are statistically significant (r^2 = 0.26, P = 0.006; r^2 = 0.48, P = 0.0001; r^2 = 0.66, P < 0.0001; respectively).

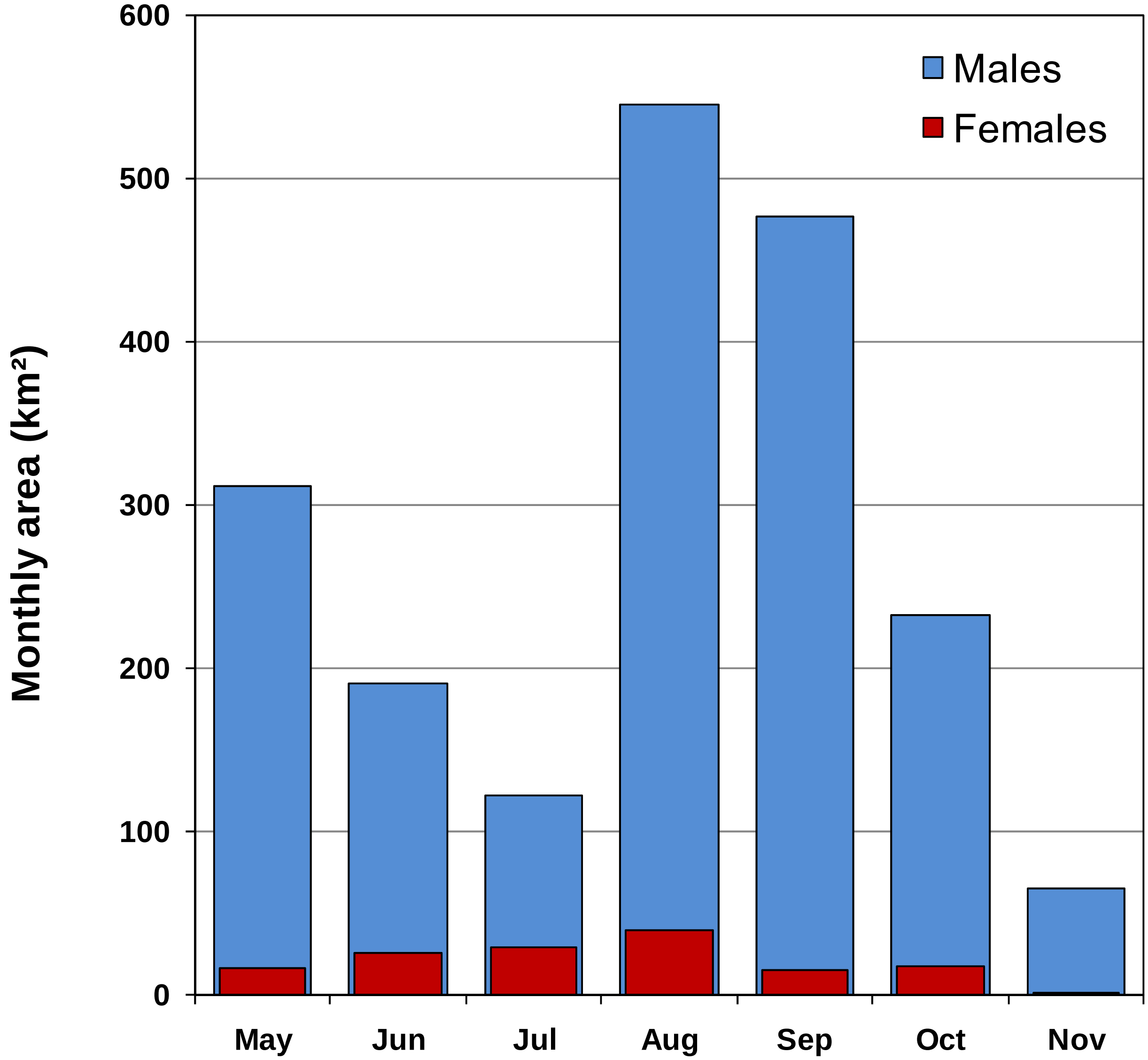

Figure 3. Mean monthly home ranges (100% MCP) of male and female black bears in Northwestern Minnesota, 2007 and 2008. Sample sizes were 4–6 males and 3–5 females each month. Females denned in November and males denned in November or December.

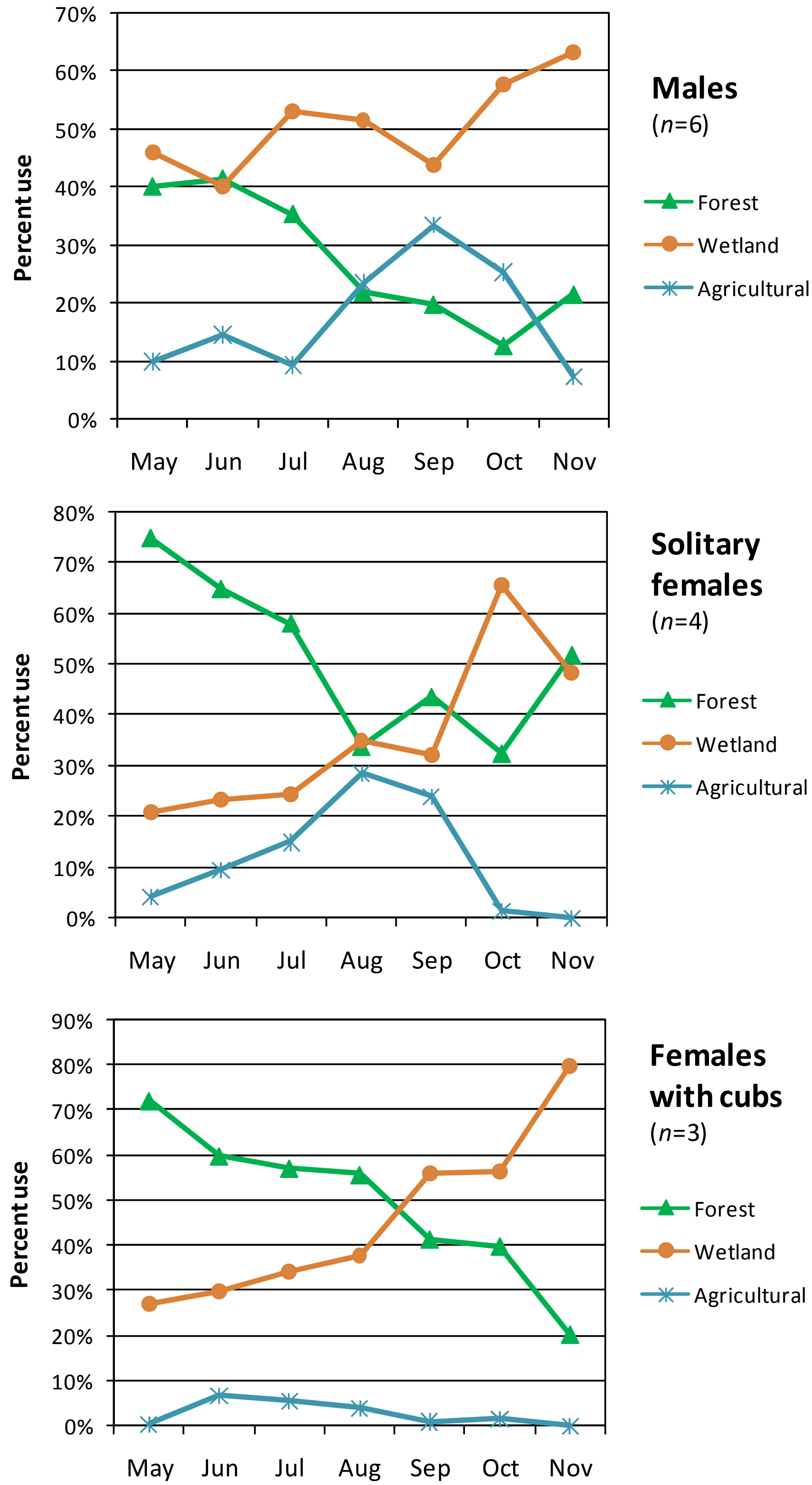

Figure 4. Trends in habitat use of 3 classes of black bears in Northwestern Minnesota, based on locations from GPS-radiocollars. Wetland areas include woody (shrub or forested) wetlands.

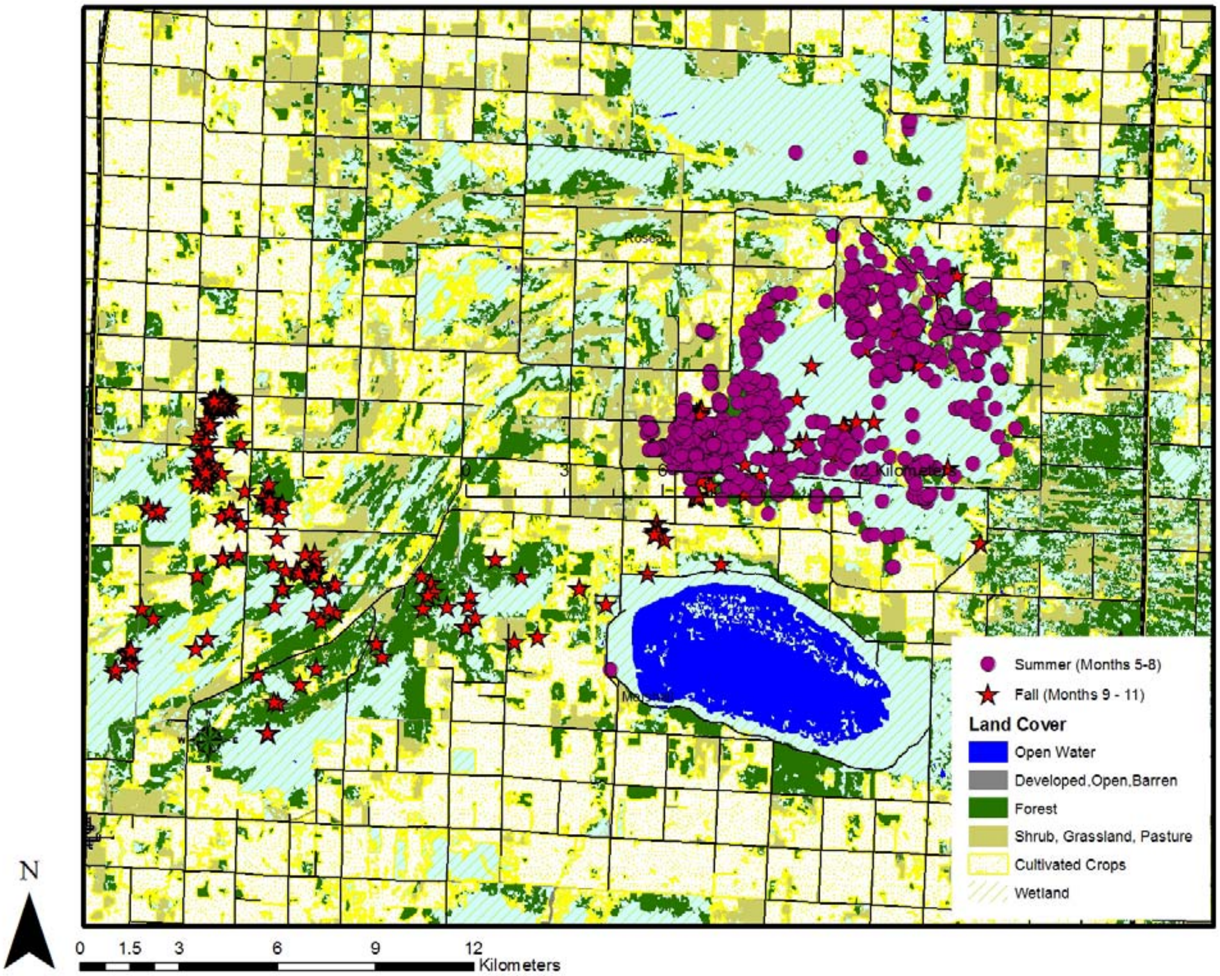

Figure 5. Home range of a GPS-collared male black bear, showing a distinct movement from the summer to fall range, 2007. The area that this bear moved to in the fall overlapped the area used by the female bear depicted in Figure 6. However, the male extensively used the agricultural fields to the north of the female's range. The northwestern-most cluster of stars represent locations during September and October, when it routinely visited a few cropfields.

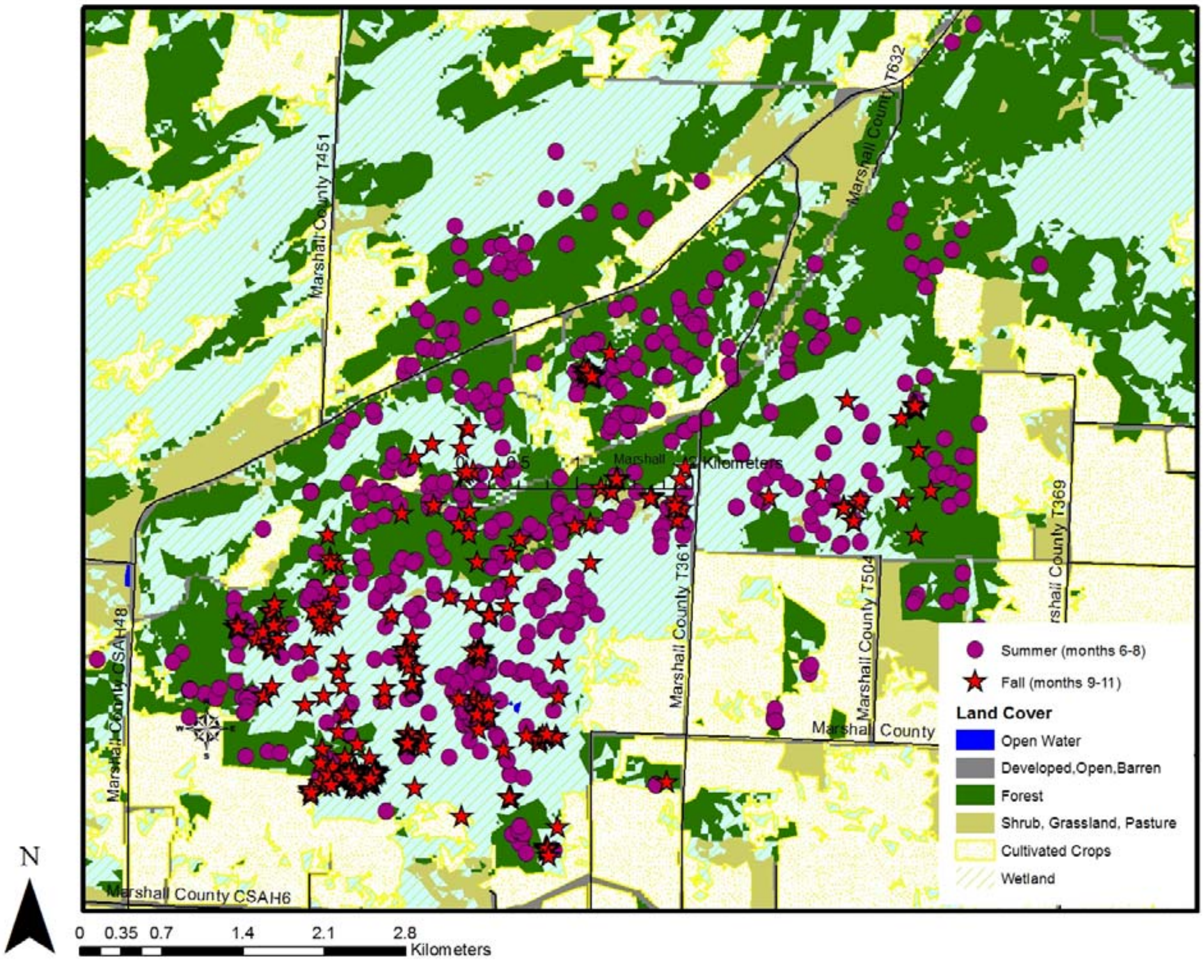

Figure 6. Home range of a GPS-collared female black bear on Thief Lake Wildlife Management Area, Northwestern Minnesota, showing areas of use during summer and fall months, 2007. This bear did not travel during fall, and although agricultural lands were adjacent to its range, it did not use them (presumably because they did not contain edible crops).

Figure 7. Bear denned in a CRP (Conservation Reserve Program) field in Northwestern Minnesota, 2009. Apparently this bear (weighing 347 lbs in early March) was flooded from its den by a heavy February rain. It remained denned in the field until March 27.

PRELIMINARY RESULTS OF TETRACYCLINE MARK-RECAPTURE ESTIMATION OF MINNESOTA'S BLACK BEAR POPULATION, 2008

Karen V. Noyce and David L. Garshelis

SUMMARY OF FINDINGS

During summer, 2008, we conducted tetracycline mark-recapture sampling of black bears (*Ursus americanus*) throughout their range in Minnesota. This was the 4th such survey in 17 years, providing independent estimates of Minnesota's bear population (bears ≥1 year old). These estimates are used to track population trends and aid in formulating harvest quotas. Bears were marked when they ingested baits of tetracycline-laced bacon, placed at approximately 4.8-km intervals across the bear range. During June–July, >3500 baits were set, marking an estimated 473–489 bears. This was lower than expected, in part due to use of a box to shield the bait from competing animals, high natural food abundance, and reduced bear abundance. During September–October, successful hunters submitted rib and tooth samples from harvested bears. These were screened microscopically for tetracycline, which fluoresces yellow under ultraviolet (UV) light. Of 1,498 samples examined, 57 were positive for tetracycline. We used a Lincoln-Petersen estimator for sampling without replacement to derive an initial population estimate of 12,400 ± 3000 bears. Because estimates based on recapture samples collected in the same year as the marking have consistently been biased low (particularly in years like 2008, when natural foods were abundant), we expect the current estimate to significantly increase after samples from the 2009 harvest are added. Despite this, the estimate indicates that the bear population has declined since 2002.

INTRODUCTION

Mark-recapture population estimates derived in 1991 and 1997 indicated that Minnesota's black bear population grew rapidly during the 1990s from about 15,500 (±1500 95%CI) bears in 1991 to around 24,000 (±3000) by 1997. In response to the perceived rapid growth rate, bear hunting permits were increased by 50% in 1998 and remained high through 2003, in an effort to curtail further population growth and avoid exceeding population levels that humans would tolerate, particularly in years when natural foods failed. At the time, some population modeling runs suggested that the bear population could "escape", reaching levels that would not be controllable through hunting. However, a third tetracycline survey in 2002 yielded a population estimate of 25,000 (±4500) bears, indicating that high hunting pressure had stemmed population growth and bear numbers had leveled off. Following the 2002 estimate, hunting permits were progressively reduced in an attempt to keep the population from declining.

After 2002, however, several indicators suggested that the population had begun to decline: (1) the number of bears killed by hunters was consistently lower than expected, based on food availability and hunter numbers (regression model); (2) population reconstruction, based on harvest age structure, suggested a recent downturn in bear numbers; (3) the proportion of the harvest composed of reproductive females continued to increase; (4) for several years, radiocollared bears experienced unsustainably high hunting mortality, and (5) wildlife managers consistently reported low levels of nuisance activity and a paucity of bear sign. The persistence of these observations indicative of population decline, despite reductions in hunting permits, prompted a 4th tetracycline mark-recapture survey in 2008.

A growing problem in previous tetracycline surveys had been the increased incidence of non-target species, specifically raccoons (*Procyon lotor*), fishers (*Martes pennanti*), and martens (*Martes martes*) taking baits. This reduced both the precision and accuracy of the estimates. By 2002, about a third of tetracycline baits were taken by small carnivores, making them unavailable to bears. In anticipation of a similar problem, Alaskan researchers (Peacock 2004) enclosed baits in wooden boxes that bears, but not small carnivores (martens in this case), could dismantle. A similar box design was successfully employed in Wisconsin to also preclude fishers and raccoons from taking tetracycline baits. We adopted this approach for the 2008 tetracycline survey in Minnesota.

OBJECTIVES

1. Obtain an updated estimate of the number of black bears (≥1 year old) in Minnesota;
2. Increase the precision of the population estimate by reducing the proportion of baits normally taken by non-target species;
3. Test the effectiveness of adding beaver (*Castor canadensis*)meat to baits to increase their attractiveness and thereby mark more bears.

METHODS

Marking

During June and July, 2008, baits with tetracycline were set across the bear range in Minnesota (Figure 1A). Bait sites were located within a grid approximately 4.8 km apart; adjacent baits were not set along the same road or trail. Where it was not possible to achieve optimal spacing, baits were set at alternative sites, but not closer than 3.2 km apart. Baits were checked 3–4 weeks after setting.

Each bait consisted of 0.4–0.5 kg of bacon, wrapped tightly around 9 500-mg capsules of tetracycline. The bait was completely enclosed within a wooden box. The box was stapled to a backboard that was then nailed 2.5-m high in a soft-barked tree. Prior tests indicated that bears, but not raccoons or fishers, could pull the box off the backboard and access the bait. Four 1.2-cm holes were drilled into the sides of the box to allow scent to emanate (but small enough to keep out squirrels). As an extra attractant, a 0.2-kg patty of ground whole beaver was added to 2/3 of the bait boxes. Again, prior tests suggested that ground beaver was more attractive to bears than bacon. After boxes were secured to tree trunks, 2 tablespoons of semi-solid grease (waste grease from a restaurant “rib-cooker”) was smeared on the outside of the box to enhance the scent at bait sites. There was not sufficient grease available to supply all bait sites, so some baits were set without added attractant and some were enhanced with other types of attractants supplied by survey participants.

Because of unexpectedly low visitation rates by bears at tetracycline baits during June and early July, 299 supplemental baits were set in late July at selected sites (Figure 1B). One objective was to try to mark more bears, although this is normally not an opportune time to bait because of the ripening of natural fruits. A second, more important objective was to assess the *cause* of the low visitation to the initial baiting. We posed 3, non-mutually-exclusive hypotheses: (1) lack of attractiveness of the baits due to poor scent emanation from inside the box; (2) lack of attractiveness of baits due to rotting of the beaver meat; or (3) low bear population. We tested these using the following 4 bait configurations, set at bait sites where bears had not taken the bait during June or early July:

1. Bacon packed in a plastic mesh bag and wired to a tree trunk; this was the method used in 1991, 1997, and 2002, so served as the control.
2. Bacon packed in a wooden box, with grease smeared on the outside, as in the 2008 survey.
3. Bacon packed in a wooden box, with beaver meat hung in a plastic mesh bag on a nearby tree.
4. Bacon packed in a wooden box, with bacon fat and molasses smeared onto the tree and loose bacon placed atop the box as extra attractants.

Recapture and Population Estimates

We obtained “recapture” samples from hunters. During the fall bear hunting season, hunters were required to submit 2 teeth (first upper premolars) and a 5-cm sample of rib bone from the bear they shot. Rib bones, cut in cross-section, were screened for tetracycline under a microscope illuminated with ultraviolet (UV) light; tetracycline incorporated into the bone tissue fluoresced yellow under UV illumination. Tetracycline reliably appears in rib bones of any bears that ingested a 4500-mg dose of tetracycline (Garshelis and Visser 1997). The mark persists until internal remodeling of bone tissue replaces older bone material, which, in young bears, can occur within 3 years. However, the amount of remodeling varies, and we could not reliably distinguish, using rib bones, marks caused by ingestion of other environmental sources of tetracycline in other years. In contrast, tetracycline marks in teeth are permanent, and their time of deposition can be determined by matching the position of the mark to cementum annuli. Thus, for all marked ribs, we examined a corresponding tooth. We also used teeth to distinguish cubs (bears <1 year old, which although not legally harvested, were sometimes shot inadvertently); these were excluded from the recapture sample.

We derived population estimates using a Lincoln-Petersen (L-P) estimator for sampling without replacement, as all sampled bears were dead and thus permanently removed from the population. Population estimates refer to the time of marking (July 2008), and exclude cubs. In marking years prior to 2008, we used a cumulative sample of recaptures from 2–3 years of harvesting, but these estimates still pertain to the year of marking.

We divided the data by Bear Management Unit (BMU) and made separate estimates for each. However, we had to group some units together either because sample sizes were too small or because there was strong evidence that bears moved between BMUs from the time of marking to the time of recapture (this is a problem in all cases, but more so for some units).

RESULTS

Marking

Wildlife managers set >3500 baits in 2008, more than in any previous survey (Table 1). New areas baited included the Red Lake and Bois Forte Indian Reservations, as well as several areas at the fringe of the Minnesota bear range. Bears visited only 17% of bait sites, considerably less than in past surveys (Table 1); visits were also surprisingly clumped, with large areas having no bear visits (Figure 2A). Also surprising, 18% of bears that visited baits either did not break into boxes or broke into boxes but did not consume the bait. In past surveys, <6% of baits visited by bears (ascertained from their claw marks on the tree) were not eaten. Consequently, fewer baits were eaten by bears in 2008 than in any previous marking year.

Both the addition of beaver meat to baits and the use of external attractants influenced the attractiveness (or detectability) of bait sites to bears. During initial baiting, bears investigated 19% of baits containing bacon plus beaver but only 14% of those with just bacon (Table 2; $\chi^2 = 11.5$, $P = 0.0007$). Also, baits with beaver meat were more likely to be consumed (85%), if investigated, than those without (78%; $\chi^2 = 3.8$, $P = 0.05$). The net effect of including beaver meat in baits was an extra 110 marked bears. Boxes smeared with the grease we supplied attracted more bears than boxes with no attractant, but boxes enhanced with attractants supplied by survey participants attracted the most bears ($\chi^2 = 25.4$, $P < 0.0001$). The effects of beaver meat and these other attractants appeared to be independent and additive (Figure 3).

The wooden boxes prevented most non-target animals from accessing baits and may have reduced their ability to detect baits. Visits by animals other than bears, primarily small carnivores, was down during initial baiting from 38% in 2002 to only 6% in 2008, and of those baits visited, only 17% were consumed (Figure 2B). During supplementary test baiting in late summer, small carnivores visited 12% of bagged baits, but only 9% of those in boxes. They ate 58% of the bagged baits they investigated, but failed to gain access into any of the boxes (Table 2).

However, boxes also may have interfered with bears' detection and consumption of baits, though this was not conclusive due to small sample sizes (Table 2). Bears visited baits in mesh bags nearly twice as often as baits in boxes with "rib-cooker" grease or beaver meat added to the site external to the bait box. Molasses and bacon grease, though, seemed to be more effective attractants; those baits were visited as frequently (or more) than bagged baits (Table 2). Boxes clearly interfered with consumption of baits; whereas bears ate 93% of the bagged baits that they investigated (n=14), they consumed only 59% of the baits in boxes (n=22; χ^2 = 4.9, P = 0.03). Notably, even bagged baits were visited far less frequently than in previous years, when this was the method of tetracycline delivery (Tables 1, 2).

Recapture and Population Estimates

The 2008 bear harvest was lower than in any of the previous 5 years, due in part to fewer hunters and very good fall foods. Harvest was similar to 2002, a year with excellent fall foods, and coincidentally the last time that bears were marked with tetracycline baits. Hunters in 2002 and 2008 submitted a similar number of usable teeth and rib bones and the number of samples that were positive for tetracycline was nearly identical (Table 3). However, because the total number of bears marked in 2008 was lower than in 2002, the number of marks recovered by hunters represented a larger proportion of the total in the population, yielding a considerably lower estimate of population size.

Samples obtained from the 2008 fall hunting season produced a population estimate of 12,400 (95% CI: 9400–15,600) bears. This was lower than any of the previous 3 estimates, based on the first year of recapture samples (i.e., from the 1991, 1997, and 2002 hunting seasons, Table 4). Population declines appeared to have occurred throughout the north-central, northeastern, and southern portions of the bear range, but not in the northwestern BMUs (Figure 4).

DISCUSSION

Tetracycline marking of bears during summer 2008 was less successful than we had hoped and was particularly disappointing in light of the extra effort made to increase the number of bears marked by increasing the number of baits, increasing their attractiveness (with beaver meat and extra lure), and preventing non-target animals from obtaining them. We had hoped to increase the number of bears marked in order to narrow the confidence intervals bracketing a new population estimate. Confidence intervals had increased with each successive tetracycline survey to date (Table 4), due to non-target animals taking baits; this had hampered our ability to interpret population trends.

Several factors contributed to the relatively poor success at marking bears in 2008. First, wild summer bear foods were better than average in all parts of the bear range and more abundant than in any previous tetracycline-marking year (Garshelis and Noyce 2009). Wild sarsaparilla (*Aralia nudicaulis*), chokecherry (*Prunus pennsylvanicus*), Juneberry (*Amelanchier* spp.), and blueberry (*Vaccinium* spp) were particularly abundant and no species surveyed experienced crop failure. A number of berry species were somewhat delayed in fruiting due to early summer conditions, but, with abundant crops and good summer moisture, the fruit-bearing season lasted longer than usual. The widespread and prolonged availability of a variety of fruits during July and August (Noyce and Garshelis 1997) likely reduced the propensity for bears to investigate and consume tetracycline baits (Table 2). Second, housing baits in boxes probably reduced bears' detection of them and certainly interfered with their consumption. Anecdotal reports of signs of bears near baits that were not taken suggest that bears did not detect them from the range normally expected. There is no obvious explanation for at least 5 cases where bears visited baits and opened the box but did not consume the bacon. Addition of beaver to baits demonstrably increased both visitation and consumption rates, thus mitigating in part the poor detection and consumption rates attributable to the boxes. Results of test baiting suggest that finding the right attractants to use at bait sites could fully compensate for the negative effects of enclosing bacon in boxes. Finally, visitation at baits was lower in 2008 because bear numbers appear to have declined significantly.

The tooth and rib “recapture” sample was also disappointingly small in 2008. The number of bears shot by hunters was 40% lower than the previous 10-year mean (Garshelis and Noyce 2009). This was attributable to better-than-average fall foods, reduced numbers of hunters, and reduced numbers of bears. Unlike 2002, the number of bears shot, after correcting for the number of hunters, was lower by about 25% than predicted based on fall food availability, continuing a trend that began in 2003 (Garshelis and Noyce 2009). Further exacerbating the relatively low number of recapture samples was a continued decline in the proportion of successful hunters that submitted samples (71% in 2008, compared to 91% in 1991; Garshelis and Noyce 2009).

The low sample sizes in both the mark and recapture phases of the 2008 tetracycline survey compromised both the accuracy and precision of the initial population estimate. Nevertheless, we believe there is ample justification for viewing this estimate as strong confirmation that Minnesota’s bear population is no longer increasing and has likely declined since 2002. The actual estimate, though, is most certainly biased low. Estimates based solely on samples collected during the fall immediately following marking have been consistently biased low (Noyce et al. 2001, Garshelis and Noyce 2006). This is because bears that consumed tetracycline baits during the summer are more likely than other bears to be shot over hunters’ baits the following fall. However, the linkage between bears consuming tetracycline baits and visiting hunters’ baits is much weaker a year later. Thus, the addition of another year’s samples (2009 hunting season) is likely to yield a higher and more accurate estimate. The amount by which the population estimate will increase is uncertain, but has ranged from 7% for the 1991 estimate to 45% for the 2002 estimate (Table 4). We assume, given the similarity of food conditions in 2008 and 2002, that the estimate could increase by an amount similar to 2002 (i.e., >45%) after another year of sampling. This would put the estimate at ~18,000 (±~3500), which is markedly fewer bears than there were in 2002 (Table 4).

BMU-specific estimates should be viewed with caution, as they violate a basic assumption of the L-P mark-recapture procedure, namely geographic closure. We know that bears routinely move among BMUs during the interval between marking and recapture, and that the extent of these movements vary year-to-year with varying food resources. There is no way to account for such movements, and an argument could be made that BMU-specific estimates are therefore unreliable. We grouped units where it was evident that such movements occurred, and present the results with some diffidence. Nevertheless, they seem to confirm a widespread population decline, but indicate that this decline did not occur in the northwestern part of the state (despite some recent record high harvests in parts of this area). A better assessment of BMU-specific changes will be made after inclusion of samples from the 2009 harvest.

FUTURE DIRECTION

We will continue to collect rib samples and teeth from dead bears at least through the 2009 bear hunting season. Depending on sampling success and the condition of fall bear foods in 2009, we will determine whether to extend sampling through a third hunting season in 2010 before calculating a final population estimate for 2008. Despite the limitations of the preliminary 2008 population estimate, we believe current evidence for recent declines in Minnesota’s black bear population is strong and that imposing annual reductions in bear hunting permits, as implemented since 2002, has been and continues to be a prudent course of action.

Contrary to popular belief, black bears in Minnesota have a relatively high reproductive rate and populations can rebound quickly when hunting pressure is reduced. Adult females are long-lived and have low natural mortality. Cub mortality in Minnesota is also low (18–20%), as is non-human-related mortality of juveniles. Control of hunter numbers constitutes a targeted means of effecting a quick turnaround in bear recruitment rates and population growth (λ).

We recommend continued periodic statewide tetracycline surveys to track population trend, unless other, more suitable methods are discovered. Based on results of 2008 sampling, we recommend continued use of wooden boxes to contain baits, but with several modifications: (1) weaken the connection between the box and backboard to make it easier for bears to dismantle boxes; (2) include more attractant external to the bait boxes; and (3) conduct further experimentation with different attractants and box ventilation to determine ways of enhancing detection, attractiveness, and consumption of baits.

ACKNOWLEDGMENTS

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Table 1. Tetracycline-marking data: 1991, 1997, 2002, and 2008.

	1991	1997	2002	2008
Baits set and checked				
Baits set	2905	2989	3122	3539
Baits not found	9	20	16	11
Baits checked	2896	2969	3106	3528
Baits visited and eaten				
Baits visited by other mammal or bird [a]	507	747	1181	218
Percent visited by other mammal or bird	18%	25%	38%	6%
Baits taken by a person	0	6	9	0
Baits taken by animal, not a bear	---	---	1015	37
Baits taken by ambiguous (possibly bear) [b]	2	64	30	16
Baits available for bears [c]	2701	2580	2572	3509
Baits visited by bears	1009	1214	755	594
Percent of available baits visited by bears	37%	47%	29%	17%
Baits eaten by bears	998	1213	707	490
Percent of visited baits eaten by bears	99%	100%	94%	82%
Percent of available baits eaten by bears	37%	47%	27%	14%

[a] Includes all baits visited by small mammals and/or birds. Some of these were not consumed; others were also visited by bears, in which cases they were recorded as taken by bears.
[b] These ambiguous cases were considered first as non-bears, then as bears in population estimates.
[c] Baits taken by small mammals or birds were considered to be available for bears half the time (½ bait).

Table 2. Visitation rates of black bears and other carnivores at baits of several configurations deployed across Minnesota during June–mid-July (primary baiting period) and late July (supplemental test baiting), 2008. Percentages reflect the portion of the baits in the given configuration (*N*).

		Bears				Other carnivores			
		Baits visited		Baits eaten		Baits visited		Baits eaten	
	N	*n*	Percent of total	*n*	Percent of visited	*n*	Percent of total	*n*	Percent of visited
Primary baiting period									
Bacon only	1051	148	14%	115	78%	52	5%	4	7%
Bacon and beaver	2118	401	19%	340	85%	139	7%	21	15%
Total [a]	3169	549	17%	455	83%	190	6%	25	13%
Supplemental test baiting									
Bacon in bag	95	14	15%	13	93%	12	12%	7	58%
Bacon in box w. grease	67	5	7%	4	80%	5	8%	0	0%
Bacon in box w. beaver in bag	64	6	9%	2	33%	3	5%	0	0%
Bacon in box w. molasses and bacon fat	64	11	17%	7	64%	9	13%	0	0%
Total	290[b]	36	12%	26	72%	27	9%	7	26%

[a] Totals for the primary baiting period are less than actual totals (see Table 1) because type of bait (with or without beaver) was not recorded at 62 original bait sites.
[b] Of 299 test baits deployed, 2 were not found again and 7 were miscoded as to type of bait deployed.

Table 3. Tetracycline recapture data: 1991, 1997, 2002, and 2008.

	1991	1997	2002	2008
Recapture				
Harvest	2143	3212	1916	2135
Ribs/teeth collected from harvest	1958[a]	2594	1417	1488
Percent of harvest sampled	91%	81%	74%	71%
Ribs/teeth collected from nuisance or car killed bears	0	17	12	10
Cub samples excluded		13	16	23
Total samples checked for tetracycline	1958	2611	1429	1498
Positive samples				
Tetracycline-marked samples	122	149	56	57
Percent of samples marked	6.2%	5.7%	3.9%	3.8%
Double-marked samples	11	10	2	2
Percent of samples double marked	9.0%	6.7%	3.6%	3.5%

[a] Excluding cubs, which are not counted in population estimates.

Table 4. Tetracycline-based population estimates, 1991, 1997, 2002, and 2008. Final estimate for 2008 will not be available until after a second recapture sample (2009 harvest).

	1991	1997	2002	2008
Estimated no. marked bears[a]				
Excluding ambiguous cases	916	1134	680	473
Including ambiguous cases		1193	709	489
Population estimate from recaptures in year of marking (Yr 1)				
Mean of estimates with and without ambiguous cases	14,600	20,300	17,500	12,400
Lower 95% CI	12,300	17,000	13,000	9,400
Upper 95% CI	16,900	24,000	22,200	15,600
Population estimate from recaptures in year after marking (Yr 2)				
Mean of estimates with and without ambiguous cases	15,800	25,600	27,900	
Lower 95% CI	13,400	20,300	20,160	
Upper 95% CI	18,200	31,100	35,860	
Population estimate from recaptures in Yr 1 + Yr 2				
Mean of estimates with and without ambiguous cases	15,300	22,400	22,700	
Lower 95% CI	13,700	19,400	18,400	
Upper 95% CI	16,800	25,400	27,100	
% increase from first-year estimate	4.8%	10.3%	29.7%[b]	
Final population estimate[c]	15,600	24,000	25,300	
% increase from first-year estimate	6.8%	18.2%	44.6%[b]	

[a] Baits consumed by bears (Table 1) divided by estimated rate of double marking (Table 2).
[b] Abundant fall foods and low hunter success rate in 2002 created a much larger first-year under-estimate in 2002 than in 1991 or 1997. A similar situation existed in 2008.
[C] Final estimate is the average of Yr2 and (Yr1 + Yr2) estimates. Modeling suggests this to be least biased.

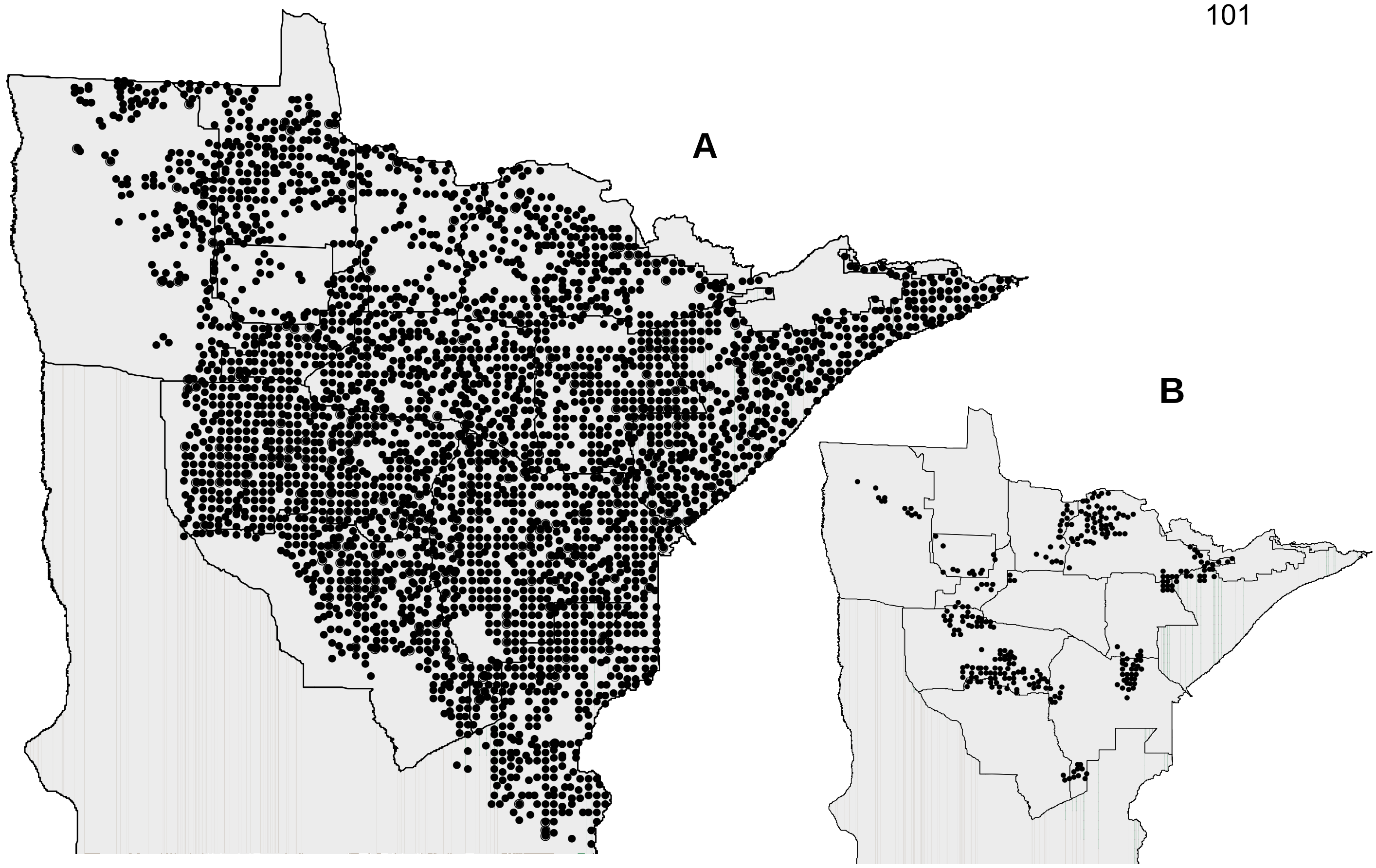

Figure 1. Locations of tetracycline baits across Minnesota's black bear range, summer 2008: (A) initial baiting, June and July, and (B) supplementary test baiting, late July.

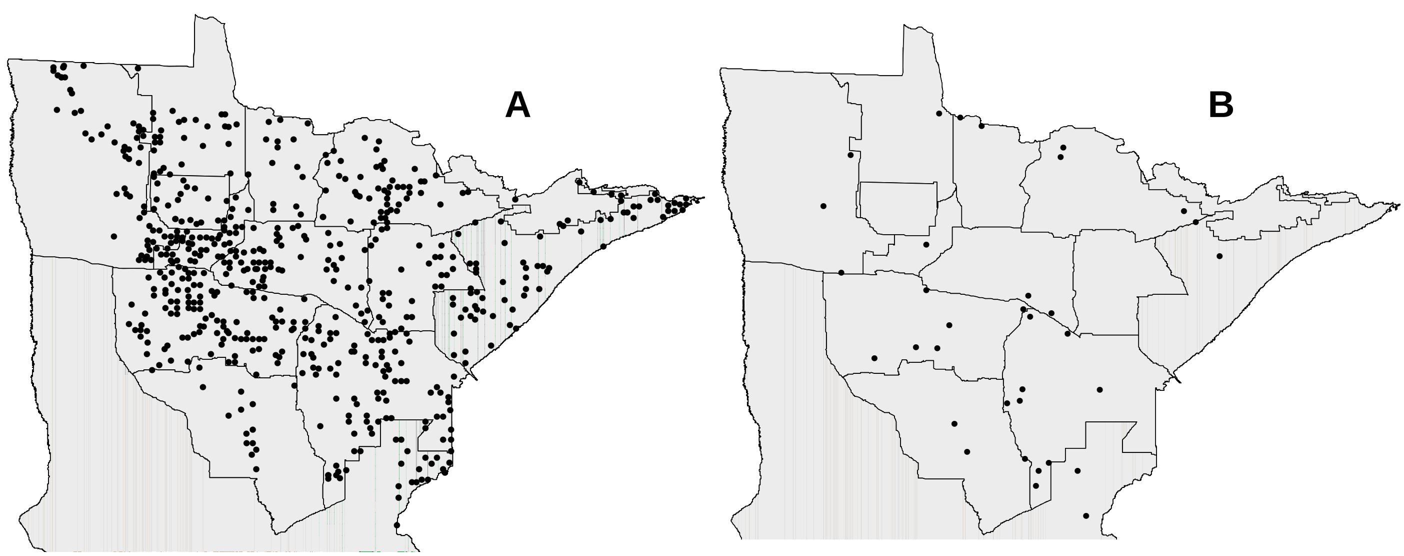

Figure 2. Locations of tetracycline baits consumed by: (A) bears and (B) small carnivores (fisher, marten, and raccoon) across Minnesota's bear range, summer, 2008.

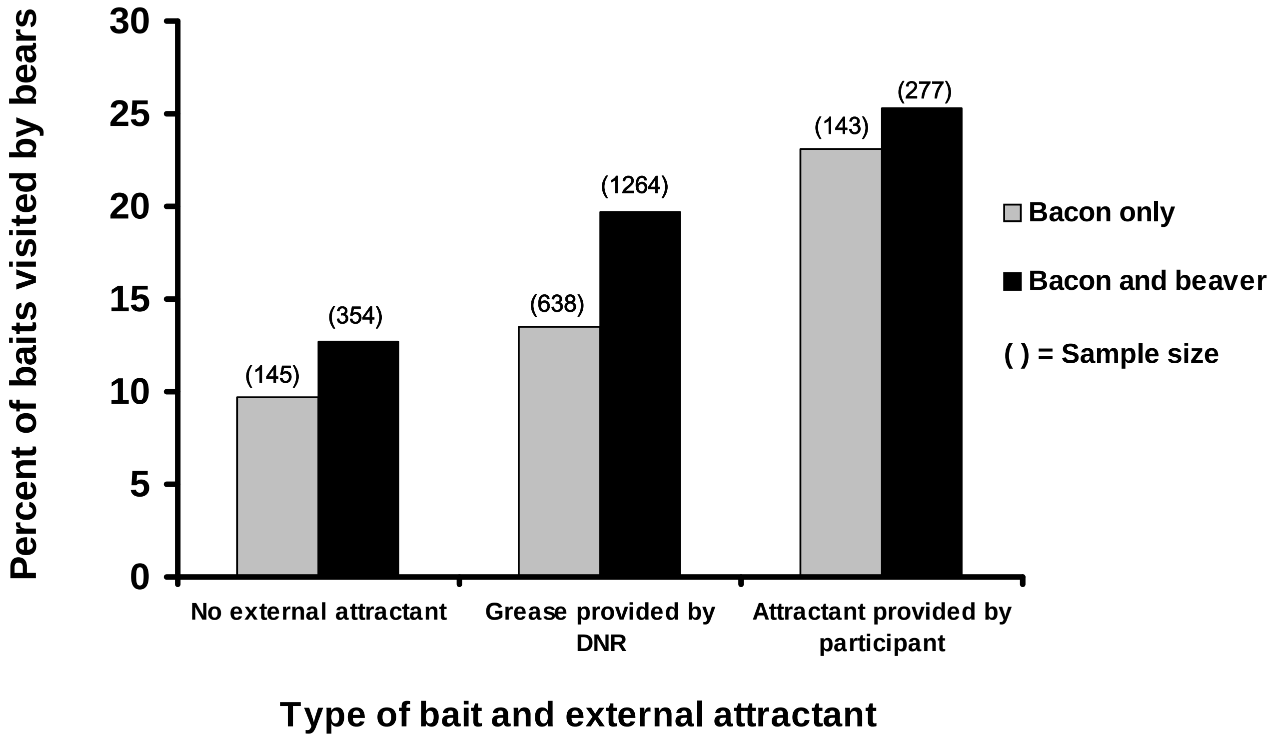

Figure 3. Influence of beaver meat and external attractants on the visitation rates of bears at tetracycline baits in Minnesota, June – July 2008.

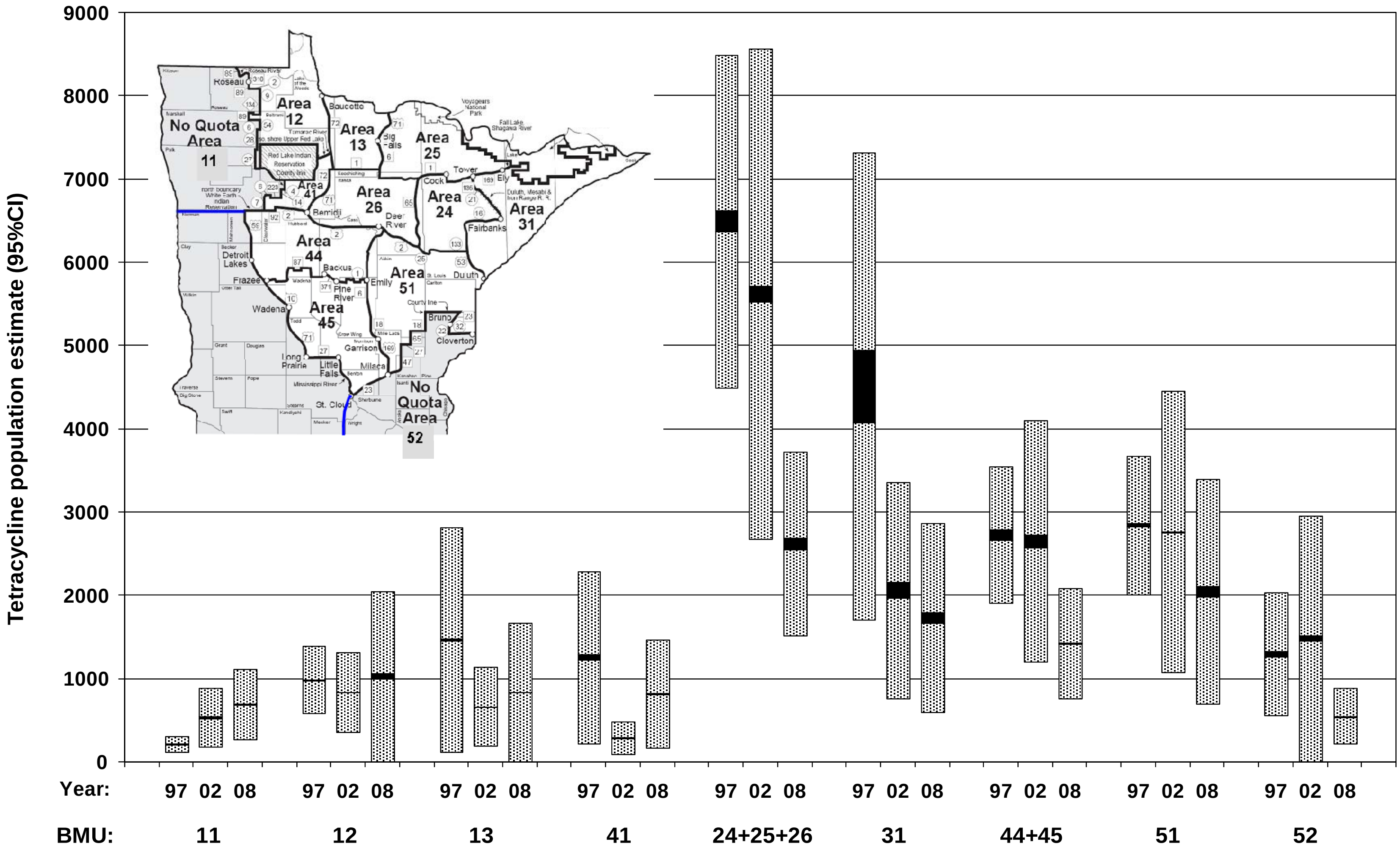

Figure 4. Population estimates of Minnesota black bears, by Bear Management Unit (BMU; boundaries shown on inset map), derived from tetracycline marking, based on sample recoveries only in the year of marking, 1997, 2002, and 2008. Black band in middle of each bar represents range of estimates; stippled bar shows span of 95% CI. Estimates based on recoveries from the year of marking tend to be biased low, by variable amounts, so across-year comparisons should be made with caution.

HABITAT SELECTION BY MALE RUFFED GROUSE AT MULTIPLE SPATIAL SCALES

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

SUMMARY OF FINDINGS

No results are available yet for this study. Field work began during spring 2009.

INTRODUCTION

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

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

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

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

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

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conifers in aspen stands, the presence of aspen clones in conifer stands, the relative importance of different age classes of aspen, and variation in the density of woody stems regenerating after harvesting aspen.

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

OBJECTIVES

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

STUDY AREA

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

Six of the 9 potential study sites had >7 times as much area in the aspen cover type than in conifer cover types. The other 3 potential study sites had the most conifer cover (10–24%, ratios of conifer:aspen area = 0.46–1.33). The site with the most conifer cover was adjacent to 1 of the 6 aspen sites, so we selected them to comprise our study area. These sites are in portions of Red Lake Wildlife Management Area and adjacent Beltrami Island State Forest.

METHODS

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

Data Collection

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

Each of the 30 selected transects will be surveyed on foot beginning 0.5 hours before sunrise during 8 different mornings during an 8-week period ending on the Friday nearest 31 May. When drumming grouse are detected during a survey, the exact location of each one will be determined by approaching it and identifying the log or other platform on which it was standing to drum, often indicated by the presence of fresh droppings. Drumming locations will be estimated with a hand-held GPS unit and can be confirmed by approaching again during subsequent surveys.

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

We will measure characteristics of ruffed grouse habitat at 3 spatial scales. The smallest scale will be the area immediately surrounding drumming locations identified during surveys. Characteristics at this scale will be measured in the field. The same variables will be measured at an unused but potential drumming platform (e.g., log or stump with no signs of use by grouse) nearest a randomly selected point within 85 m of each identified drumming location. A circle with a radius of 85 m represents the "core area" (2.3 ha) of a male's home range during the 2-month "drumming season" (6.7 ha, Archibald 1975). An 85-m radius will ensure that selected unused locations will be within the home range, whereas the 146-m radius of the home range would not.

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

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

Data Analysis

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

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

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HABITAT SELECTION, NEST SUCCESS, AND SURVIVAL OF SHARP-TAILED GROUSE IN RELATION TO MANAGEMENT OF OPEN LANDSCAPES IN MINNESOTA

J. Wesley Bailey and Michael A. Larson

SUMMARY OF FINDINGS

The Forest Wildlife and Populations Research Group of the Minnesota Department of Natural Resources (MNDNR) initiated research to examine habitat selection, nest success, and survival of sharp-tailed grouse (*Tympanuchus phasianellus*) in relation to management of open landscapes in east-central Minnesota. The 2009 breeding season served as the pilot period of a planned long-term study. During spring 2009 we trapped and radiocollared 17 sharp-tailed grouse; 10 hens and 7 cocks from 4 dancing grounds. Because of late trapping efforts, we only caught hens at 2 of the 4 dancing grounds. As of July 6, 2009, unknown predators killed 3 hens and 1 cock. We found 8 nests all within a variety of habitats ranging in succession from wooded bog edge to willow brush-grass, to treeless residual grass within a burned meadow. The mean distance from nearest dancing ground to nest locations was 973.97 m (min. 504.17, max. 1899.74). The mean distance between neighboring nest locations for all samples was 1546.80 m (min. 528.25, max. 3074.80).Of the 8 nests, 1 was excluded from survival analysis because the hen died while foraging away from the nest; all 13 eggs remained in the nest. Of the 7 nests considered for analyses, predators destroyed 4 nests during late incubation and 3 nests successfully hatched at least 1 sharp-tailed grouse chick. For all nests, mean clutch size was 11.0 eggs (7 minimum, 13 maximum). To date, data collection is ongoing; therefore, complete results are not available at this time. We do, however, anticipate results from this study will help managers in east-central Minnesota focus their brushland management efforts, and provide the context to develop refined management objectives for managing open landscapes for sharp-tailed grouse consistent with MNDNR's Strategic Conservation Agenda.

INTRODUCTION

Brush and open-landscape management efforts comprise a considerable portion of MNDNR's Section of Wildlife budget. Despite the time and resources allocated to open-brushland management, it is uncertain how these efforts have influenced sharp-tailed grouse populations. Annual dancing ground surveys may provide some insight into sharp-tailed grouse response to management. Information on sharp-tailed grouse habitat selection and survival is lacking and would better guide management efforts in Minnesota. This research will evaluate habitat selection of sharp-tailed grouse hens during the breeding season, determine habitat characteristics that best predict nest-sites and influence nest survival, and finally, determine factors contributing to hen mortality.

Sharp-tailed grouse have the most extensive range of all prairie grouse (Silvy and Hagen 2004), but for several decades have suffered marked population declines (Connelly et al. 1998) because of poor management or habitat loss (Riley 2004). In Minnesota, sharp-tailed grouse populations have declined sharply because open brushland habitat they inhabit is vulnerable to destruction, senescence, and conversion to incompatible cover (Berg 1997). Annual spring surveys of dancing grounds in Minnesota indicate the mean number of sharp-tailed grouse per dancing ground has fluctuated since 1980 but the overall trend has been negative, although a positive trend occurred within the past 5 years (Larson 2008).

Because of the population decline, and subsequent lower harvest, MNDNR set a long-term goal to nearly quadruple the annual sharp-tailed grouse harvest from a 5-year average (2000-2004) of 11,400 to 40,000 (MNDNR 2007). An increase in harvest may indicate a positive response to management. To attain this goal, forest planning efforts have identified priority open landscapes for brushland management (MNDNR 2007) with an objective of increasing the quantity and quality of sharp-tailed grouse habitat throughout its range in Minnesota. However, brushland management is expensive and this management activity comprises a considerable portion of MNDNR Section of Wildlife budget. Openland/brushland management ranked first in average annual expenditures of all habitat management activities within the Forest Habitat Program for FY06-FY08 with mean expenses averaging $800,000 (MNDNR, unpublished data).

Despite the amount of resources allocated to brushland management, significant information gaps about sharp-tailed grouse habitat relationships exist in Minnesota. Managers are unsure whether their actions benefit the species. Managers suspect their efforts are at least maintaining sharp-tailed grouse populations, but do not have current information to bolster their assumptions. A cursory examination of annual dancing ground counts compared to management in Aitkin County revealed no statistical correlation between the mean number of sharp-tailed grouse counted at dancing grounds and acres managed (Figure 1). Although annual dancing ground surveys may provide some insight to a population level response to management (Connelly et al. 1998), linking survival and productivity to habitat features would better direct management resources (Martin 1992).

Information on sharp-tailed grouse habitat in Minnesota stems from studies conducted in northwestern MN (Artmann 1971, Schiller 1973, Wells 1981) and in east-central MN (Niemi and Hanowski 1992, Hanowski et al. 2000). Based on results from these studies, MNDNR and Minnesota Natural Resources Conservation Service (MN NRCS) provide management guidance on recommended habitat characteristics for sharp-tailed grouse. However, without basic information on nesting habitat and factors that limit adult and nest survival, managers make habitat management decisions with limited information, which results in management of unknown efficacy. Results from this study would help managers make informed management decisions by linking on-the-ground efforts to positive responses by sharp-tailed grouse populations. MNDNR would achieve its goals for the sharp-tailed grouse more quickly and efficiently by understanding the impact of management activities.

OBJECTIVES

1. To determine which habitat features most influence nest-site selection;
2. To determine which habitat characteristics and time-specific factors most influence nest survival;
3. Determine if sharp-tailed grouse hen mortality is influenced by time-specific factors, habitat characteristics at multiple scales, and site management history; and
4. Determine if site management history is correlated with habitat use of hen sharp-tailed grouse.

STUDY AREA

During the 2009 pilot period, we conducted field work during late April through July on Willowsippi Wildlife Management Area and two private parcels in Aitkin County near Palisade, Minnesota (Figure 2). Each site has a history of open-brushland management and supports at least one sharp-tailed grouse dancing ground.

METHODS

Capture, Marking, and Monitoring Sharp-Tailed Grouse

We utilized radio-telemetry to determine sharp-tailed grouse habitat use and selection, nest success, and hen survival. We captured sharp-tailed grouse hens at active dancing grounds in late April and early May using walk-in funnel traps (Toepfer et al. 1987). To efficiently trap hens from several dancing grounds during any given trapping period (i.e., 2-4 days), we did not trap at dancing grounds more than 10-15 miles apart. We deployed and operated 4 traps, 1 per dancing ground, from 05:00-09:00 am and from 17:00-20:30 pm from 28 April through 07 May 2009. We fit sharp-tailed grouse with 14-15 g necklace radiocollars

(Advanced Telemetry Systems, Model A3960 and Holohil Systems, Model RI-2BM) with a lifespan of 18-24 months. Each transmitter included a 12 hour mortality sensor. We located each bird using a null-peak vehicle mounted telemetry system (Brinkman et al. 2002) and with a portable receiver and 2-element H-style antenna. To locate individual birds, we triangulated locations by obtaining 3 directional azimuths to the radiocollared bird's position (White and Garrott 1990). We used homing to find nests and locate dead birds (White and Garrott 1990).

Sampling Design for Habitat Selection During Breeding Season

The most important reproductive decisions a hen must make is selecting where to nest and selecting sites where broods can grow and be protected from predators (Bergerud and Gratson 1988). Furthermore, nest success and survival of sharp-tailed grouse hens and broods during the breeding season may be the most important vital rates affecting population viability (Schroeder and Baydack 2001, Manzer and Hannon 2007). Therefore, our objective is to determine habitat selection during the breeding season with emphasis on obtaining locations pre- and post-hatch. To determine habitat selection, we will triangulate each hen's location daily to once every 5 days, depending on if hens are actively nesting or brood-rearing, until the end of the breeding season.

We will attempt to collect ≥30 locations per 3-month spring-summer field season for each sharp-tailed grouse hen (Seaman et al. 1999, Leban et al. 2001). Before hatch, hen movements will likely be limited to foraging bouts near the nest (Connelly et al. 1998, Roersma 2001). Therefore, locations may be more concentrated in space, in which case, fewer observations may be needed to characterize habitat use during laying and incubation.

We will evaluate habitat selection by determining the number of locations per hen within defined cover types and disturbance categories during the breeding season. Cover types are defined as those amenable to management (e.g., alder/willow swamp). For each hen, we will use a Geographic Information System (GIS) to delineate cover types of interest on digital aerial photographs covering breeding home ranges as defined by the minimum convex polygon of telemetry locations from all hens. If cover types cannot be distinguished from digital imagery, we may ground-truth these as needed. To determine cover type categories, we will use Minnesota Gap Analysis Project GAP cover types and cover types within the modified Minnesota land cover classification system. Because management histories at study sites will likely include more than 1 management type (e.g., burn, shear, herbicide), we will evaluate time since disturbance rather than management technique. We will determine time since disturbance for lands with a history of management during 1999-2008 and categorize time since disturbance into 2-year intervals (e.g., 0-1, 2-3, etc.). We will include 2009 management information as data become available.

Sampling for Nest-Site Selection

We will collect nest-site habitat data immediately after nest fate is determined. Breeding home ranges define available habitat during the breeding season; however, nesting habitat requirements may differ from brood-rearing habitat (Connelly et al. 1998). For each nest, we will identify 2 non-nest points by selecting a random bearing and distance from the nest to measure available habitat. Selection of nest-sites occurs after hens select nesting areas (Johnson 1980; third-order selection). Because hens will nest near locations from the previous season (minimum distance 0.2 km; Roersma 2001) we will constrain sampling of random non-nest points to within 55-120 m of the nest to ensure we are sampling habitat available to each hen. We will ensure that non-nest points are in suitable habitat by selecting another random point and distance should a non-nest point occur in unusable habitat (e.g., forest). We will search non-nest points to be sure a nest is not present. At each non-nest point we will measure the same habitat variables used at the actual nest.

We will evaluate nest-site selection by measuring used habitat variables from a variety of spatial scales and compare these to available habitat. The scales of interest include habitat features directly at the nest, within 30 m, 100 m, and 3200 m of the nest. We chose a 30 m radius because this scale represents the minimum mapping unit for MN GAP data and approximates the smallest scale at which management may appreciably affect sharp-tailed grouse habitat. We chose a 100 m radius because at this scale available habitat should include many potential nest sites, and the 3200 m represents the maximum distance sharp-tailed grouse are known to nest away from a dancing ground (Connelly et al. 1998).

Because we are interested in structural features of sharp-tailed grouse habitat amenable to management, we will focus on type of cover selected for nesting, vegetation height, and density at and surrounding the nest. We will estimate overhead cover by placing a black 20-cm diameter disk with 9 25-mm^2 equally spaced white squares in the nest bowl (or random point), and count the number of white squares ≥50% blocked by vegetation measured from 1-m away directly overhead (Roersma 2001, Manzer 2004). We will measure the height of woody and herbaceous vegetation at the nest and random points to the nearest 0.1 m.

Because sharp-tailed grouse often nest under or near brush (Connelly et al. 1998), our measures focus on quantifying the characteristics of the brush patch associated with the nest. We will use the area of an ellipse to calculate the area of the brush patch. We will then calculate mean vegetation height by measuring the height of the tallest stem in each cardinal direction from the center of the brush patch using a telescoping measuring pole. We will calculate brush patch volume by multiplying patch area by mean patch height and will use volume in analyses. To measure low vegetation cover we will use a density board (Nudds 1977, Noon 1981) 2 m high and 0.3 m wide and divided into 0.25-m sections each containing 25 5x5 cm squares. We will place the board at the nest and determine the percent cover of woody and herbaceous vegetation in each 0.25-m section from 5 m away in each cardinal direction by subtracting the number of squares <50% covered from 100. For analysis, we will use the average value of each 0.25 m section. If during the pilot season we determine sharp-tailed grouse are nesting in grassland more than brush, we may instead use Robel pole readings, which are a standard measure for grassland bird nests (Robel et al. 1970). Within 30 m of the nest, we will count the number of stems ≥1.50 m in height because these may provide perches for avian predators (Manzer and Hannon 2005).

We will classify cover types on digital aerial photography based on MN GAP cover types and cover types within the modified Minnesota land cover classification system, and convert vector data to raster to facilitate analyses. We will use GIS to measure spatial pattern (i.e., configuration) and characteristics of woody and herbaceous cover. We will use FRAGSTATS (McGarigal and Marks 1995) to measure the following characteristics: area (ha) of cover types, cover type patch size (ha), mean cover type patch size (ha), number of cover type patches, cover type density (number/100 ha), edge density (m/ha) between cover types, core area metrics (e.g., total core area), and nearest neighbor metrics (e.g., average proximity of brush patches to determine if sharp-tailed grouse select brush cover on average closer to other brush patches). We will record the distances from nests to “hostile” features, both natural and anthropogenic, (e.g., shelterbelts, buildings, improved roads, unimproved roads, transmission lines, hard forest edges) to determine whether the features are related to nest location (Pitman et al. 2005).

Sampling for Nest Survival

After capture and allowing for an adjustment to the transmitter to lapse (1 week), we will triangulate hens daily to locate early nesting attempts. Once a hen has initiated a sedentary behavior (i.e., is found repeatedly at same location), we will home in on the hen and visually confirm the nest location, mark it with GPS, photograph the nest location and surrounding vegetation, and record the number of eggs. We will remotely monitor hens every 1-2 days during the nesting period. We will obtain a final clutch-size count when the hen is absent from the nest. On subsequent visits, we will assess if nests are still active by triangulating locations every 1-2 days. To avoid flushing the hen, we will only visit nests if monitoring indicates incubation has ceased (i.e., multiple absences) or if the hen substantially changes movements (e.g., long-distance away from the nest or more frequent movements). We will visually inspect the nest at that time to determine nest stage, nest fate, and number of hatched eggs. We will consider all nests hatching ≥1 egg successful. If a nest is depredated, we will attempt to locate the hen and repeat daily sampling until a new nest location is determined; afterwards, we will repeat our nest monitoring protocol of triangulating hens daily to once every 2 days until a fate is determined.

During the course of nest monitoring and post-hatch monitoring, we will check for mortality signals to the end of the breeding season. Although our focus is on survival during the breeding season, and because transmitters will have a 24-month battery life, we will remotely check for mortality signals approximately 2 times a month throughout the year. Dead hens will be recovered as quickly as possible and categorized by the probable cause of mortality into 4 classes: avian predation, mammalian predation, unknown cause, and exposure. We will classify predation type while in the field by assessing any damage to the transmitter and carcass, and the location of the recovery site. Exposure is assigned if a severe weather event occurred immediately before mortality is indicated. If no damage is found on the transmitter and/or carcass, we will assign this as unknown.

Data Analysis

Before collecting radio-telemetry observations, we will evaluate telemetry error to estimate precision of directional azimuths (White and Garrott 1990, Withey et al. 2000). To test error for actual field conditions (Withey et al. 2000), and to emulate transmitters on birds, we will place 5 transmitters 10 cm off the ground attaching them to saline filled bottles (J. Giudice, personal communication). Locations will be unknown to the observer in habitats representative of our study area. We will obtain 60 directional bearings for each transmitter from 4 known locations (Lesmeister et al. 2009). We will report linear error of each estimate, the standard deviation of bearing errors, and estimated size (ha) of each error polygon (confidence ellipse associated with location estimate). We will use a regression model to relate linear error to observer, distance of transmitter from receiver, and geometry of bearing intersections (Withey et al. 2000). To account for telemetry error, we will exclude all locations with error ellipses larger than 2.0 ha or the mean size of grass-brush patches within each dancing ground complex, whichever is larger.

To evaluate habitat selection, we will use a type 2 use-availability design where individual sharp-tailed grouse hens are identified and use is measured for each hen and availability is measured at the population level (Thomas and Taylor 1990, Alldredge and Griswold 2006, Thomas and Taylor 2006). We define the population as each individual dancing ground complex; therefore, availability will be defined as the extent of a minimum convex polygon around the collection of breeding season home ranges (Lesmeister et al. 2009) of hen sharp-tailed grouse within each dancing ground complex. Home range characteristics will be compared to those of the study area (i.e., dancing ground complex; second-order selection; Johnson 1980) because managers manage for large blocks of habitat. The experimental unit is an individual sharp-tailed grouse hen. We will define use as the proportion of time spent within individual cover types based on the number of observations per hen. We will not include successive locations of hens at nest sites in these analyses. We will use the Johnson rank-order method (1980) to rank the difference in proportional use of habitats and proportional availability of habitats for each sharp-tailed grouse hen using Hotelling's *T*-square or equivalent approximate *F* statistic (Johnson 1980, Alldredge and Griswold 2006). Johnson's procedure allows for inference only at the population level and does not require independence of animal relocations, which allows for the continuous observation of individuals (Alldredge and Griswold 2006).

Because we determined the number of non-nest points and will constrain sharp-tailed grouse selection to within the same habitat area as the nest, this represents a discrete-choice type analysis (Keating and Cherry 2004). We consider sharp-tailed grouse nests uncommon given the amount of available nesting cover types within study sites and we will search non-nest points to be sure a nest was not present; therefore, we will meet assumptions for case-control or discrete-choice analyses (Keating and Cherry 2004). To assess nest-site selection, we will employ case-control conditional logistic regression (Allison 1999, SAS Institute 2004 [PROC LOGISTIC with STRATA statement]) to compare nests to paired non-nest points. Each matched set will consist of the nest (case) and 2 random points (controls) located within 55-120 m of the nest.

To assess nest survival, we will use the logistic-exposure method (Shaffer 2004) to model the effects of habitat variables from our nest-site selection analyses and time-specific variables (e.g., date, nest stage) to estimate nest survival. The logistic-exposure method allows values for time-dependent explanatory variables, such as nest stage, to change among nest-observation intervals, but assumes they are constant within an interval (Schaffer 2004). We will fit logistic exposure models with the GENMOD procedure (SAS Institute 2004) by selecting a binomial response distribution (i.e., nest failed or survived the interval) and supplying the user defined link function described by Schaffer (2004).

We will use an information-theoretic framework (Burnham and Anderson 2002) to evaluate support for our sharp-tailed grouse nest-site selection and nest survival hypotheses Nest-site selection will evaluate habitat measured at multiple scales while nest survival will evaluate temporal and habitat effects. For the nest-site selection analyses, we will plot predicted probabilities that a site was a nest site as a function of covariates that had effects on nest site selection. We also will report descriptive statistics for nest-sites and non-nest points. For the nest survival analysis, we will first determine the most supported temporal effects (e.g., date of season, nest stage) and then include these in all habitat models; holding these nuisance effects constant in each habitat model will reduce the total number of models we fit and let us focus on habitat effects in the second stage of the analysis (Grant et al. 2005). For nest-site selection and nest survival analyses, we will examine each global model for multicollinearity by calculating tolerance values for variables (Allison 1999) and evaluate the goodness-of-fit with the Hosmer and Lemeshow (2000) goodness-of-fit test and will examine the overdispersion parameter (Burnham and Anderson 2002).

We will evaluate support of candidate models based on Akaike's Information Criteria for small sample sizes (AIC_c) and report likelihood values, AIC_c, ΔAIC_c, and Akaike weights (w_i) for each model (Burnham and Anderson 2002); we will use the effective sample size (Rotella et al. 2004) to compute AIC_c. Because of the potential for substantial model selection uncertainty, we will present model-averaged coefficients, unconditional standard errors, odds ratios and 95% confidence intervals from models with $\Delta AIC_c \geq 2$ (Burnham and Anderson 2002). We will limit interpretation of effects to those with odds-ratio confidence intervals that did not overlap 1. We will estimate the relative importance of covariates from averaged models by summing the Akaike weights (w_i) from all competing models in which the variable appeared (Burnham and Anderson 2002).

Survival over the breeding season will be measured from 15 April to 15 July annually. We will estimate breeding season survival using known-fate models in Program MARK (White and Burnham 1999) and use model selection to evaluate hypotheses about differences in survival during this time-period. We will calculate survival probabilities for radio-marked hens using the Kaplan-Meier product limit estimator (Kaplan and Meier 1958) with the staggered entry design (Pollock et al. 1989). We will code each encounter as live, dead, or censored. Because sharp-tailed grouse may need to adjust to radiotransmitters after capture, we will exclude any deaths within 1 week of capture from survival modeling.

RESULTS

During spring 2009 we trapped and radiocollared 17 sharp-tailed grouse; 10 hens and 7 cocks from 4 dancing grounds. We caught 3 hens at Willowsippi WMA and 7 hens at Rono dancing ground; we did not capture hens at Sherman nor Gun Lake dancing grounds. As of July 6, 2009, unknown predators killed 3 hens and 1 cock.

We found 8 nests all within a variety of habitats ranging in succession from wooded bog edge to willow brush-grass, to treeless residual grass within a burned meadow. The mean distance from nearest dancing ground to nest locations was 973.97 m (min. 504.17, max. 1899.74). The mean distance from nearest dancing ground to nest locations was 1160.06 m at Rono and 663.82 m at Willowsippi; for all samples, the mean distance was 973.97 m. The maximum distance between nest locations at Rono was 3074.80 m; the minimum distance was 528.25 m. At Willowsippi, the maximum distance between nest locations was 1357.80 m; the minimum distance was 702.73 m. The mean distance between neighboring nest locations for all samples was 1546.80 m (min. 528.25, max. 3074.80).

Of the 8 nests, 1 was excluded from survival analysis because the hen died while foraging away from the nest; all 13 eggs remained in the nest. Of the 7 nests considered for analyses, predators destroyed 4 nests during late incubation and 3 nests successfully hatched at least 1 sharp-tailed grouse chick. For all nests, mean clutch size was 11.0 eggs (7 minimum, 13 maximum).

DISCUSSION

While these data have not been analyzed, we anticipate results from this study should provide managers in east-central Minnesota with new insight into their brushland management efforts, and provide the context to evaluate and adjust their management of brushlands for sharp-tailed grouse consistent with MNDNR's Strategic Conservation Agenda (MNDNR 2007). By evaluating results of management practices, specific habitat features that directly influence reproductive success should be identified, increasing our ability to effectively conserve habitats (Martin 1992). Our results will allow managers to set quantitative goals for vegetation management by providing the range and configuration of cover types that benefit reproduction and survival of sharp-tailed grouse. All information gained from this study will help formulate cost-effective strategies for sharp-tailed grouse management and provide information on habitat relationships needed to effectively manage sharp-tailed grouse.

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Table 1. Distance (meters) to nest locations from dancing grounds where hens were captured during 2009 breeding season in Aitkin County, Minnesota.

Hen	Dancing ground	Distance to nest (m)
166.139	Rono	1246.45
166.238	Rono	1899.74
166.434	Rono	746.18
166.445	Rono	804.43
166.395	Rono	1103.50
166.120	Willowsippi	784.49
166.161	Willowsippi	702.81
166.483	Willowsippi	504.17

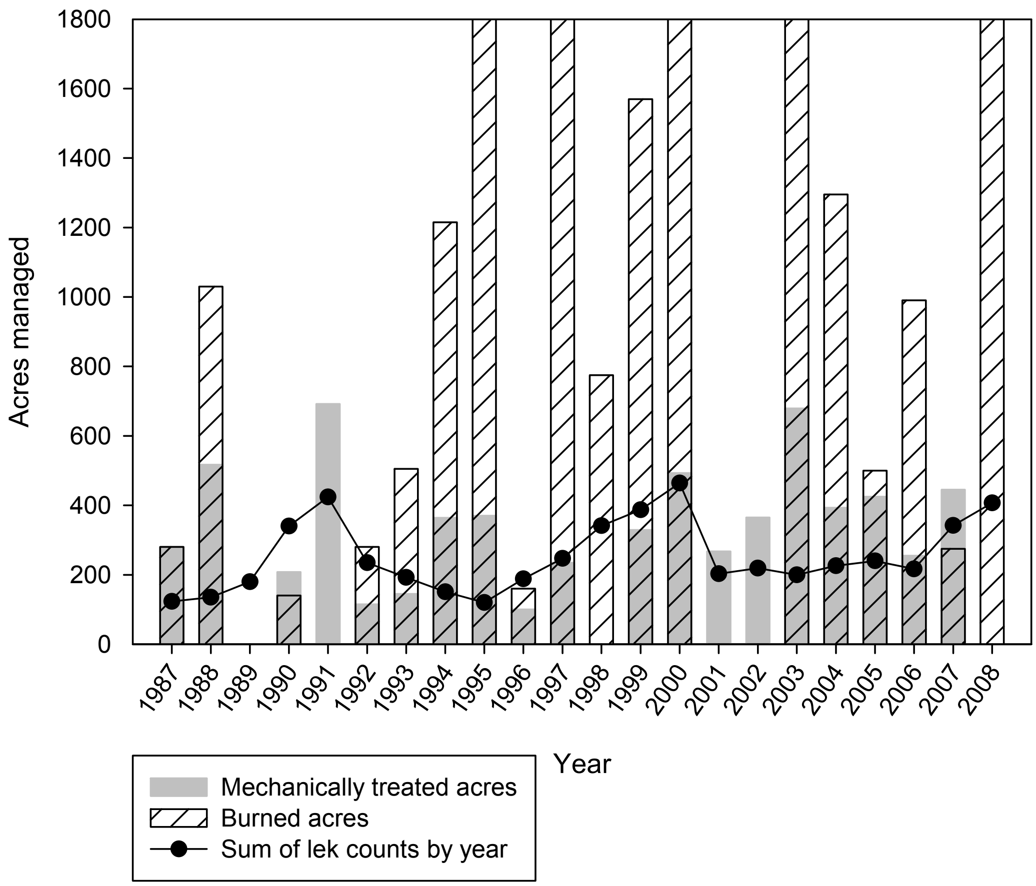

Figure 1. Relationship between mean count of male sharp-tailed grouse from annual dancing ground surveys and acres of managed brushland by treatment type within the Aitkin Wildlife Work Area during 1987-2008 in Aitkin County, Minnesota. Acres managed scaled to best show relationship; annual burned acres exceed 1800 in various years.

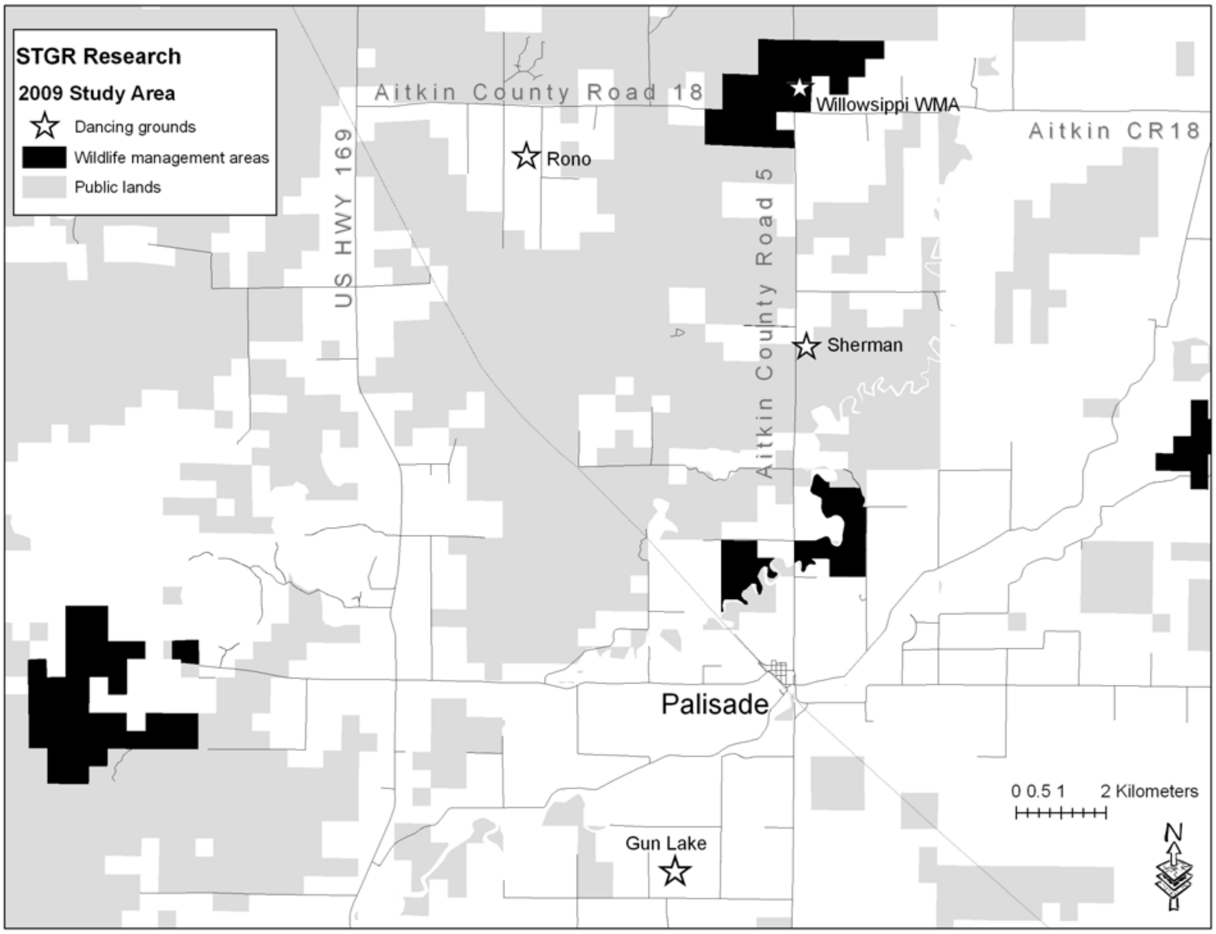

Figure 2. East-central study area in Aitkin County, Minnesota, during 2009 pilot season. Stars indicate locations of sharp-tailed grouse dancing grounds where trapping occurred and indicate focus areas for radio telemetry observations.

ASSESSING THE RELATIONSHIP OF CONIFER THERMAL COVER TO WINTER DISTRIBUTION, MOVEMENTS, AND SURVIVAL OF FEMALE WHITE-TAILED DEER IN NORTH CENTRAL MINNESOTA

Glenn D. DelGiudice and Barry A. Sampson

SUMMARY OF FINDINGS

The goal of this long-term (1991-2005) investigation is to assess the value of conifer stands as winter thermal cover/snow shelter for white-tailed deer (*Odocoileus virginianus*) at the population level. Over the course of this15-year study period, we radiocollared and monitored a total of 452 female deer, including 43 female newborn fawns. Data generated from this study provided the basis for scientific and popular articles addressing supplemental feeding effects on winter food habits of white-tailed deer; age-specific survival and reproduction; cause-specific mortality; seasonal migration; safe capture, chemical immobilization and handling; wolf predation; bait selection and capture success; and diseases of deer; as well as progress in applied geographic information system (GIS) technology. The focus of several of these papers was to explore new, more scientifically rigorous analytical approaches to viewing the diverse data sets we were accumulating. During the past year, we've completed the publication process for the last of these papers, addressing seasonal migration and approaches to analysis (i.e., time-scale and origin) of deer survival from birth to 18.5 years old. We have completed organization and quality control checks of several large data sets, and we have begun statistical analyses of relationships between environmental variation (e.g., severity of winter weather conditions, harvests of conifer cover), the physiological status of study deer, their use of habitat, and survival, focusing on nutrition as a mechanistic thread. Below we highlight some of these preliminary findings.

INTRODUCTION

The goal of this long-term investigation is to assess the value of conifer stands as winter thermal cover and snow shelter for white-tailed deer at the *population level*. Historically, the availability of conifer stands has declined markedly relative to the increasing numbers of deer in Minnesota and elsewhere in the Great Lakes region. The level of logging of all tree species collectively, and conifer stands specifically, has recently reached the estimated allowable harvest. Most land management agencies and commercial landowners typically restrict (to varying degrees) harvests of conifers compared to hardwoods, because of evidence at the *individual-level*, indicating the seasonal value of this vegetation type to white-tailed deer and other wildlife species. However, agencies have anticipated increased pressure to allow more liberal harvests of conifers in the future. Additional information is needed to assure future management responses and decisions are ecologically sound. Both white-tailed deer and the forests of the Great Lakes region have significant positive impacts on local and state economies, and they are highly regarded for their recreational value.

Hypotheses and Objectives

In proposing and planning this study in autumn 1990, we hypothesized that winter severity and the availability of moderately dense (40-69% canopy closure [Class B]) and dense (≥70% canopy closure [Class C]) conifer stands on winter range affects their use by female white-tailed deer as thermal cover or snow shelter, deer movements (i.e., migration) and distribution. Further, we hypothesized that nutrition is likely the mechanistic thread between this environmental variation (e.g., ambient temperature, snow depth, conifer availability) and the population performance (survival and reproduction) of deer. Relative to varying winter severities, the objectives of the comprehensive, quasi-experimental approach of this study have been to:

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

STUDY DESIGN AND PROGRESS

This study (1991-2005) employed a replicated manipulative approach, which is a modification of the Before-After-Control-Impact design (BACI; Stewart-Oaten et al. 1986; see DelGiudice and Riggs 1996). The study involves 2 control (Willow Lake and Dirty Nose Lake) and 2 treatment sites (Inguadona Lake and Shingle Mill Lake), a 5-year pre-treatment (pre-impact) phase, a 4-year treatment phase (conifer harvest served as the experimental treatment), and a 6-year post-treatment phase. The 4 study sites located in the Grand Rapids-Remer-Longville area of north-central Minnesota are 13.0-23.6 km^2 (5.0-9.1 mi^2) in area. The study began with the Willow and Inguadona sites during winter 1990-1991. The Shingle Mill and Dirty Nose sites were included beginning in winter 1992-1993. The objective of the experimental treatment (impact) was to reduce moderately dense and dense conifer stands (good and optimum thermal cover/snow shelter, respectively) to what is considered poor cover (< 40% canopy closure [Class A]).

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

Winter Severity and Nutritional Status of Study Deer

Winter nutritional restriction or deprivation of white-tailed deer, moose (*Alces alces*), and elk (*Cervus elaphus*) can be assessed by sequential collection and chemical analysis of fresh urine voided in snow (DelGiudice et al. 1988,1989, 1997, 2001; Ditchkoff 1994; and others). Collecting urine specimens associated with fresh deer tracks within 72 hours of a recent snowfall allowed us to associate urine chemistry values and nutritional assessments to known times (≤ 72 hours) with a high degree of certainty (DelGiudice et al. 1988).

Pooling across our 4 study sites, mean urea nitrogen:creatinine (UN:C) ratios of the snow-urine samples collected during winters of varying severities clearly shows that higher values, caused by starvation-level nutritional restriction and accelerated net catabolism of endogenous protein (DelGiudice et al. 1987,1991,1994), occurred during the 3 winters with the highest winter severity index values (1995-96, 1996-97, and 2000-01; Figure 1). The winter severity index (WSI) is calculated by accumulating 1 point for each day with an ambient temperature ≤ -17.7°C and 1 point for each day with snow cover ≥38 cm during November-May. During these 3 severe winters, maximum snow depths were 86-97 cm, more than twice as deep as threshold depths that cause deer to engage in energetically-costly bounding to move about (Kelsall and Prescott 1971, Moen 1976) and when forage availability is markedly diminished. When we examined the proportion of snow-urine specimens collected during each winter with UN:C ratios indicative of moderately severe ($3.0 \leq x < 3.5$ mg:mg) and severe or starvation-level (≥3.5 mg:mg) nutritional restriction, a clear pattern was revealed in each of the 4 study sites (Figure 2). Generally, severe nutritional restriction was most evident during winter 1995-96, 1996-97, and 2000-01, when winter conditions were most severe, but patterns varied somewhat among the 4 sites (Figure 2). Severe nutritional restriction was most apparent throughout the period 1992-93 to 2004-05 at Shingle Mill Lake and least apparent at Inguadona Lake. Maximum WSI values were significantly (P = 0.057) related to the proportions of snow-urine specimens with UN:C ratios indicative of severe nutritional restriction of deer during the study period; WSI values accounted for about one-third (r^2 = 0.35, y = 2.958 + 0.051x) of the variation of these proportions (Figure 3). Of these WSI values, temperature-days (r^2 = 0.39, y = -1.982 + 0.190x, P = 0.039,) exhibited a stronger relationship to these indicators of severe nutritional restriction than snow-days (r^2 = 0.30, y = 5.031 + 0.065x, P = 0.080). Additionally, this indicator of severe nutritional restriction in our study deer was related to percent winter mortality (r^2 = 0.52, y = 3.942 + 0.381x, P = 0.013, Figure 4) and percent mortality by wolf predation (r^2 = 0.38, y = 5.161 +0.415x, P = 0.044). We have reported wolf predation as the primary source of natural winter mortality for these deer (DelGiudice et al. 2002, 2006). Additional statistical analyses of snow-urine chemistry profiles will focus on specific relationships within control and treatment sites relative to winter severity, experimental harvests of conifer cover, and use of habitat

Winter Severity and Use of Conifer Cover

The 4 study sites are mosaics of various vegetative types (polygons) that we classified by the dominant 2-3 tree species, tree height, and for conifers, by canopy closure class as well. Using ArcMap of ArcGis (Version 9.3), we measured the nearest distance (m) of diurnally radio-located female deer (Dec-May) to conifer stands with moderately dense (Class B) and dense (Class C) canopy closures, which based on findings in the literature, serve as good to optimal thermal cover and snow shelter, respectively, for deer. In our preliminary examination of deer location and habitat data from 3 winters ranging in maximum WSI values from 60 to 195, mean nearest distances of deer were closer to dense (Class C) conifer stands and to moderately dense or dense (Class B or C) stands *within* each site during the more severe winter, except at Dirty Nose Lake (Figure 5). At Dirty Nose Lake deer were located relatively close to conifer stands of Classes B or C (≤ 75 m) during all 3 winters, regardless of severity, and this may be related to the overall vegetative composition of this site, as well as the availability and arrangement of these conifer classes. We will be studying this and associated issues in more detail. On the other 3 sites (Willow, Inguadona, and Shingle Mill), there were significant ($P \leq$ 0.05) differences in the mean nearest distance to moderately dense or dense conifer cover during the mildest (1994-95) versus the most severe (1995-96) winters; distances tended to be intermediate during the moderately severe winter (1993-94). Among all 4 sites, mean nearest distances of female deer to dense conifer stands were inversely related (r^2 = 0.30, y = 323 – 0.978*x*, *P* = 0.028) to WSI values (Figure 6), with a slightly stronger association with snow-days (r^2 = 0.31, y = 285 – 1.376*x*, *P* = 0.026) than with temperature-days (r^2 = 0.28, y = 413-3.330*x*, *P* =0.036). Mean nearest distances of deer to moderately dense or dense conifer stands were also related (r^2 = 0.35, y = 158 –0.488*x*, *P* = 0.015) to WSI (Figure 6). Interestingly, we also observed a marginally-significant inverse relation (r^2 = 0.19, y = 134 –2.790*x*, *P* = 0.095) between mean nearest distances of deer to moderately dense or dense conifer stands and the proportion of snow-urine UN:C ratios indicative of severe nutritional restriction (Figure 7). This lends evidence to the notion that during increasingly severe winters of deep snow, deer spend more time close to and within moderately dense to dense conifer stands, where nutrition tends to be less available and nutritional restriction is more severe.

These promising preliminary findings suggest that the diverse data we've collected during our 15-year study period will allow us to more closely assess potential relations between environmental variation, deer behavior (e.g., habitat use), and the vital rates that most strongly affect population performance. Additional analyses will focus more specifically on the potential effects of the experimental reduction of conifer cover on the 2 treatment sites versus little to no reduction on the 2 control sites.

ACKNOWLEDGMENTS AND PROJECT COOPERATORS

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

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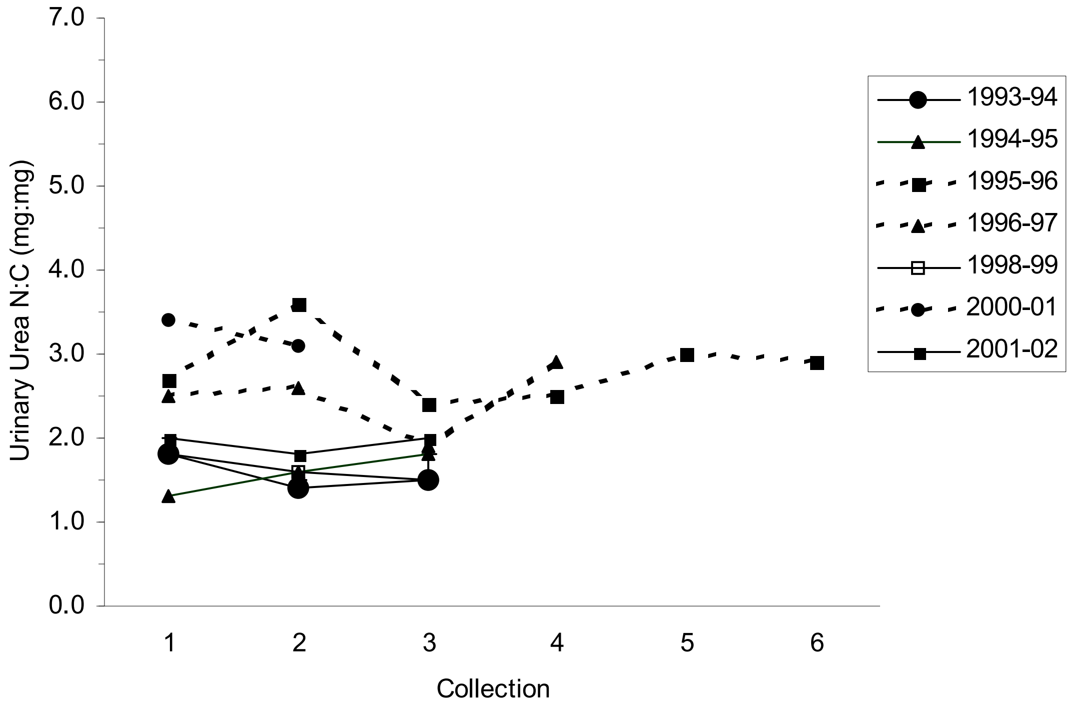


Figure 1. Mean urea nitrogen (N):creatinine (C) ratios in urine recently (≤ 72 hours) voided in snow (snow-urine) by free-ranging white-tailed deer, all 4 study sites pooled, north-central Minnesota, January-March 1993-94 to 2001-02 (as indicated in the legend) . Samples sizes per collection ranged from 94 to 143. Maximum winter severity index (WSI) values were 126, 61, 195, 159, 46, 153, and 45 (see text for definition).

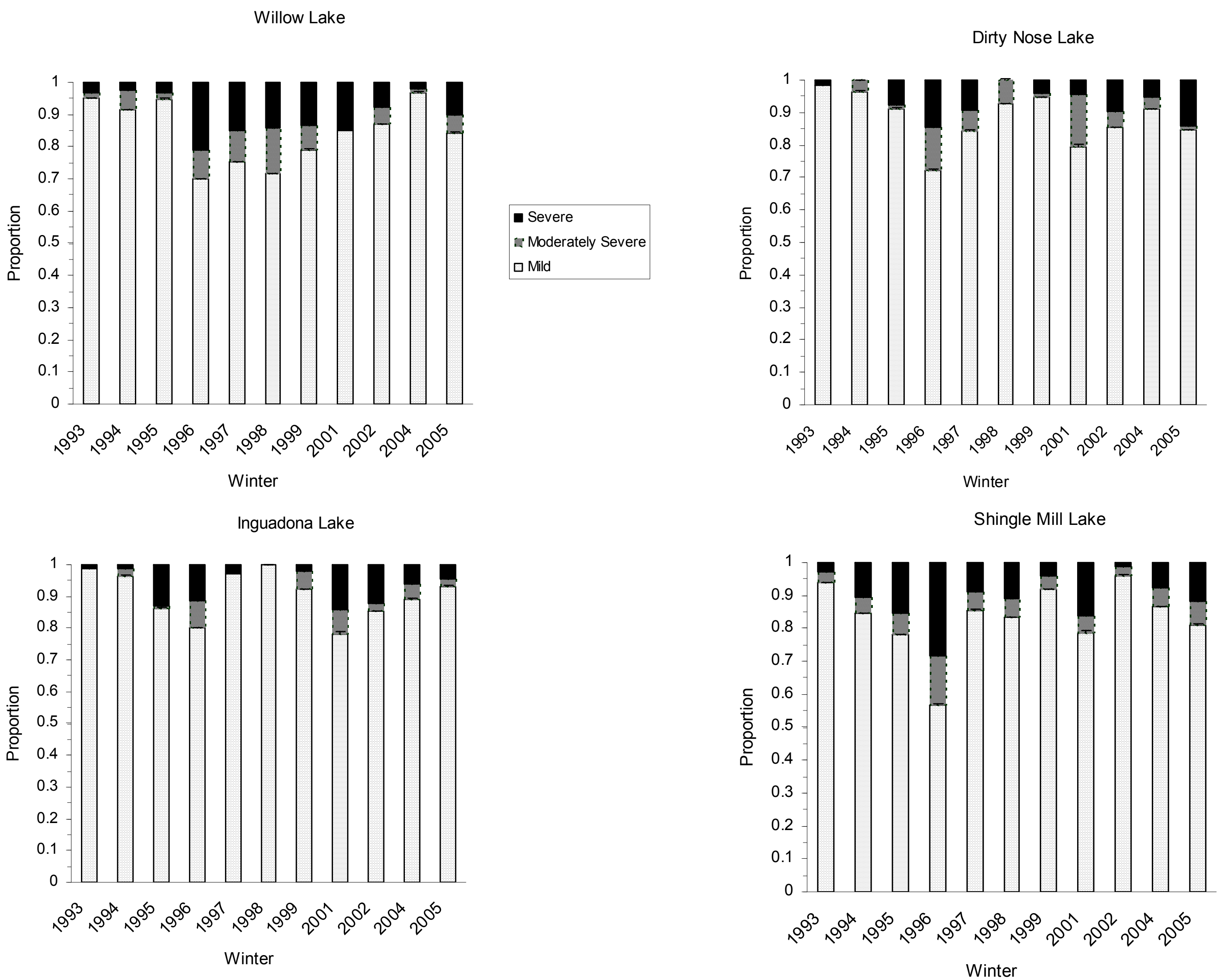


Figure 2. Proportion of urine specimens in snow (snow-urine) with urea nitrogen:creatinine ratios indicative of mild (< 3.0 mg:mg), moderately severe (3.0 ≤ x < 3.5 mg:mg), and severe (≥3.5 mg:mg) nutritional restriction in white-tailed deer on 2 control sites (Willow Lake, Dirty Nose Lake) and 2 treatment sites (Inguadona Lake, Shingle Mill Lake), north-central Minnesota, January-March 1992-93 to 2004-05 (as indicated on the x-axis).

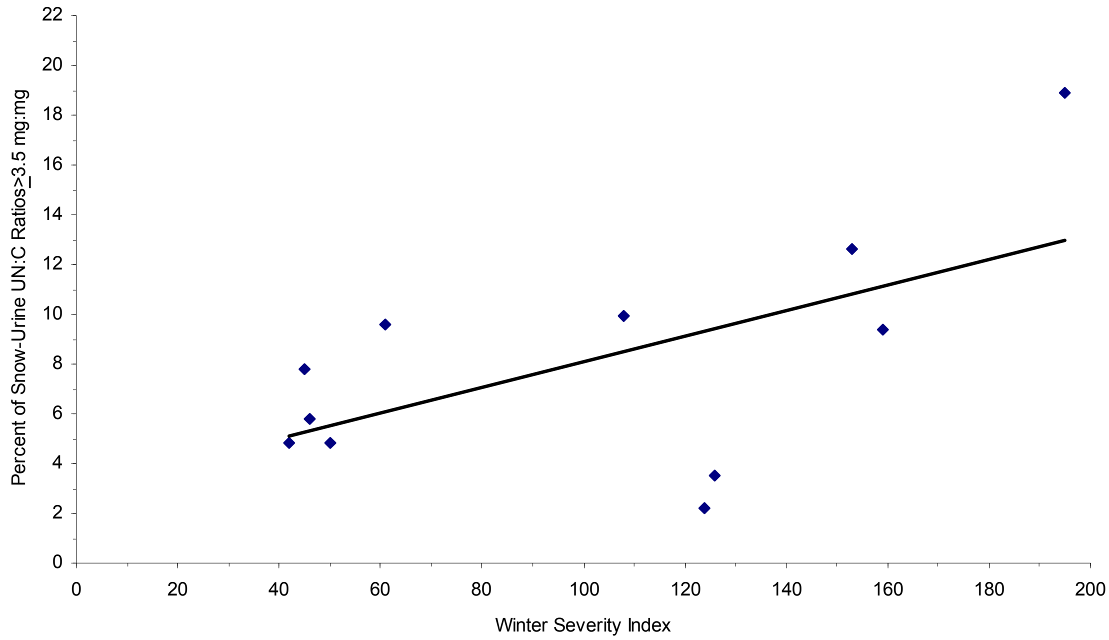


Figure 3. Relationship (r^2 = 0.35, y = 2.958 + 0.051*x*, *P* = 0.057) of the annual maximum winter severity index (see text for definition) to the percent of urine samples in snow (snow-urine) of white-tailed deer with urea nitrogen: creatinine (UN:C) ratios indicative of severe nutritional restriction (≥3.5 mg:mg), all 4 study sites pooled, north-central Minnesota, winters 1992-93 to 1998-99, 2000-01, 2001-02, 2003-04, and 2004-05.

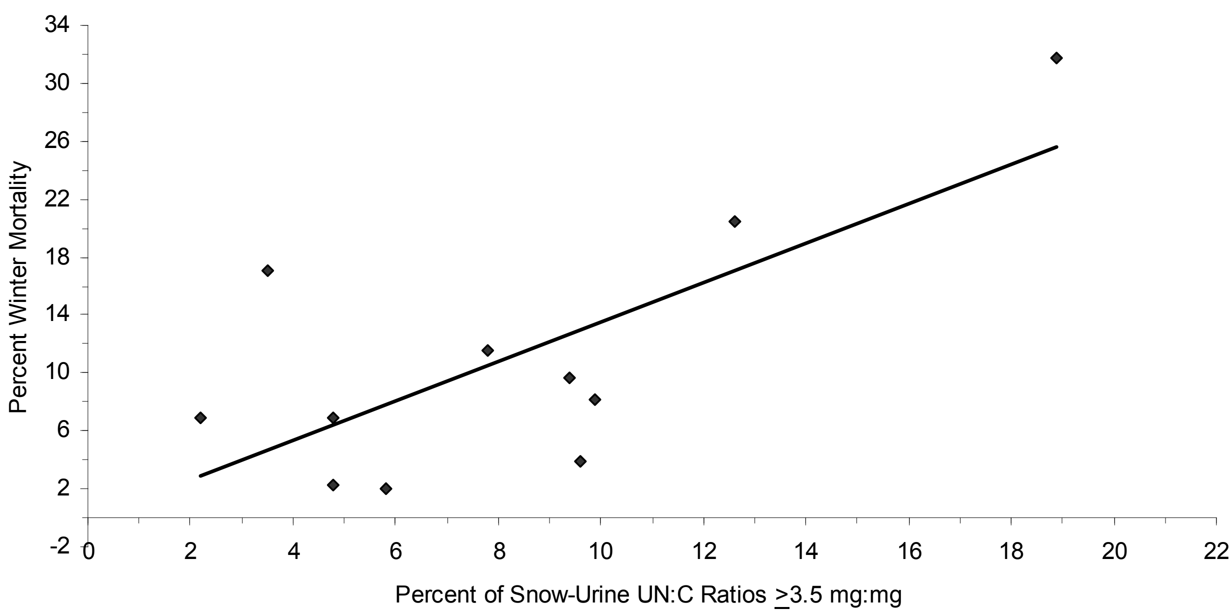
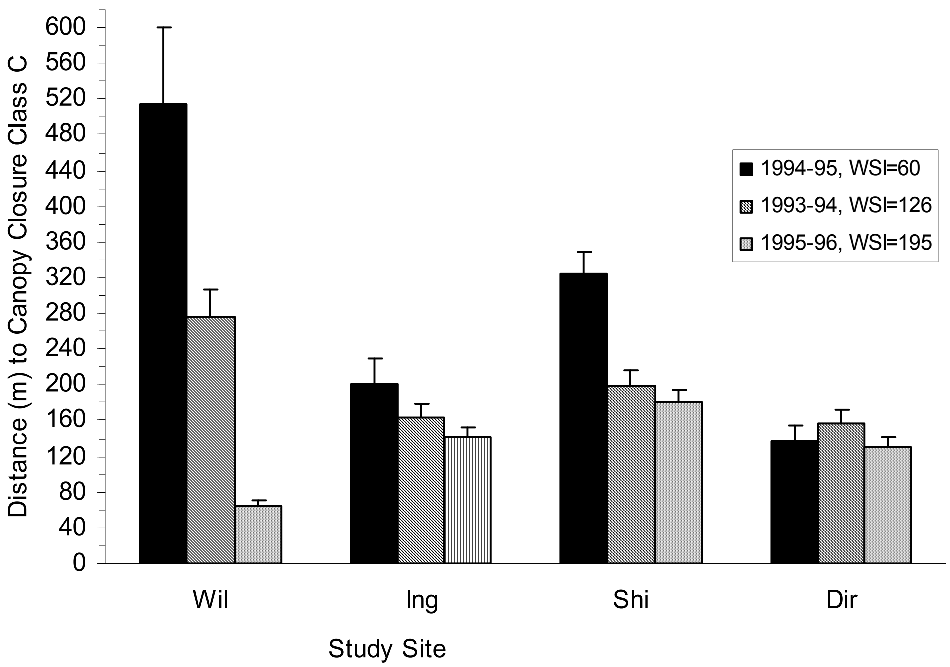


Figure 4. Relationship (r^2 = 0.52, y = 3.942 + 0.381*x*, *P* = 0.013) of the annual percent of urine samples in snow (snow-urine) of white-tailed deer with urea nitrogen:creatinine (UN:C) ratios indicative of severe nutritional restriction (≥3.5 mg:mg) to percent winter mortality, all 4 study sites pooled, north-central Minnesota, winters 1992-93 to 1998-99, 2000-01, 2001-02, 2003-04, and 2004-05.



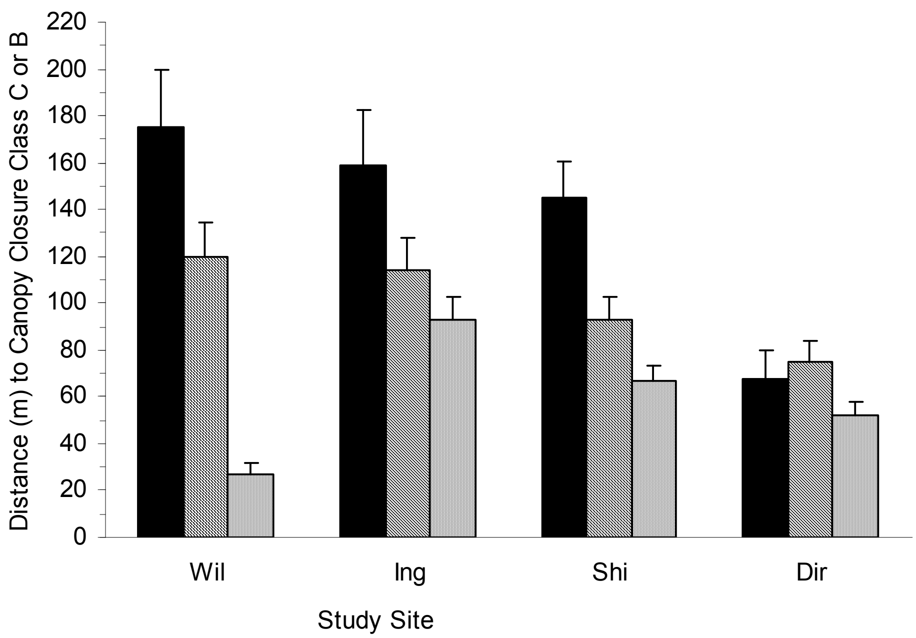


Figure 5. Mean nearest distance of radiocollared, female white-tailed deer to conifer stands classified as canopy closure (cc) Class C (top, ≥70% cc) and to those classified as canopy closure Class B (40% < x < 70% cc) or C (bottom) on the Willow Lake (Wil), Inguadona Lake (Ing), Shingle Mill Lake (Shi), and Dirty Nose Lake (Dir) study sites, north-central Minnesota, winters 1993-94, 1994-95, and 1995-96 (winter severity indexes [WSI, see text for definition] appear within the legend).

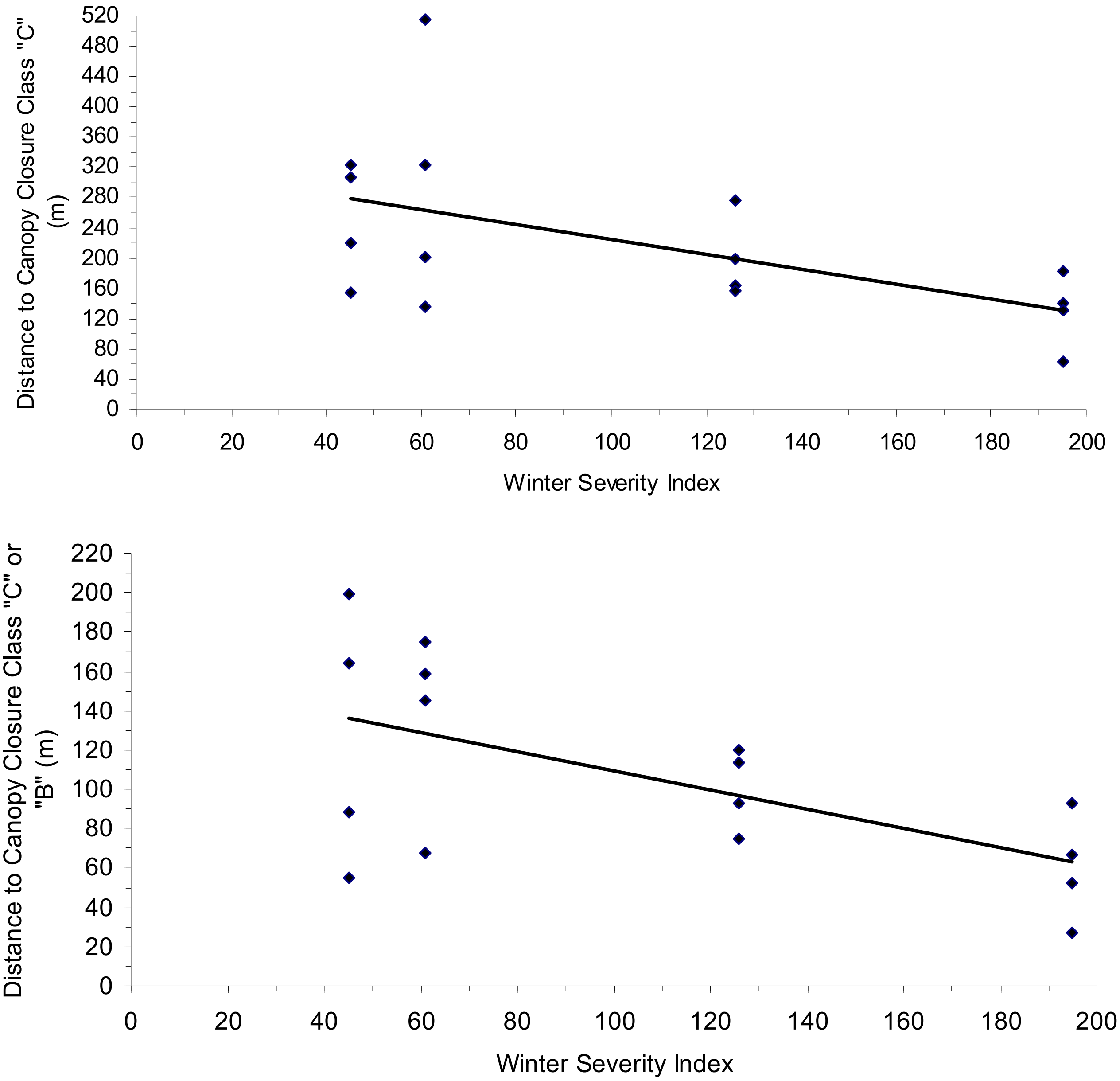


Figure 6. Relationship of the annual maximum winter severity index (see text for definition) to the mean nearest distance of radiocollared, female white-tailed deer to conifer stands classified as canopy closure (cc) Class C (top, ≥70% cc; r^2 = 0.30, y = 323 – 0.978x, P = 0.028) and to those classified as canopy closure Class B (40% < x ≤ 70% cc) or C (bottom, r^2 = 0.35, y = 158 –0.488x, P = 0.015), all 4 study sites pooled, north-central Minnesota, winters 1993-94, 1994-95, 1995-96, and 2001-02.

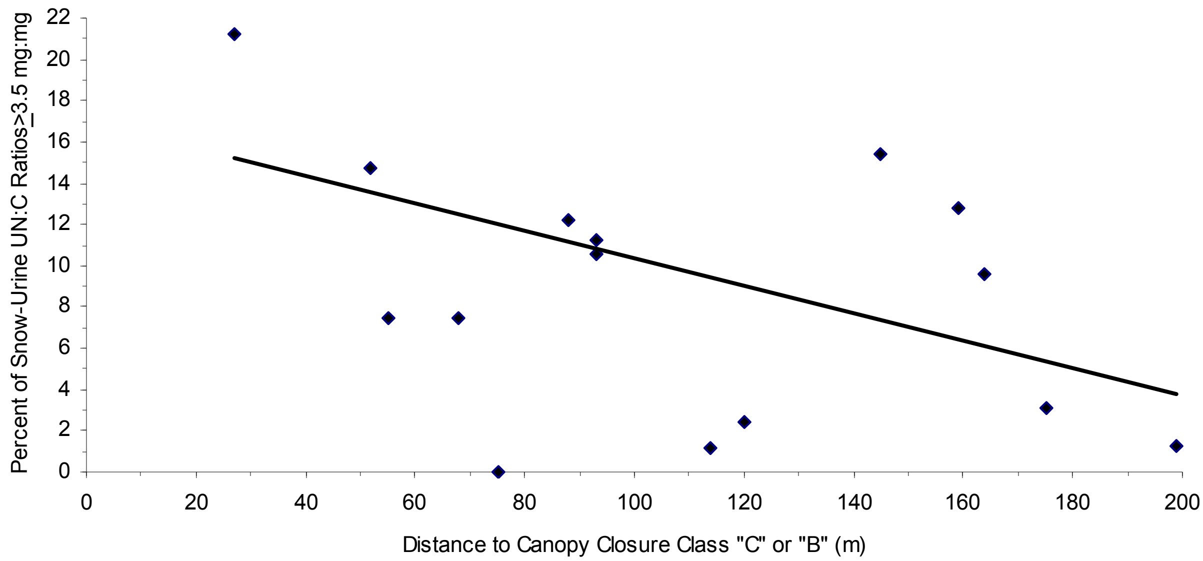


Figure 7. Relationship (r^2 = 0.19, y = 134 –2.790x, P = 0.095) of the mean nearest distance of radiocollared, female white-tailed deer to conifer stands classified as canopy closure (cc) Class C (≥70% cc) or B (40% < x ≤ 70% cc) to the percent of urine samples in snow (snow-urine) of white-tailed deer with urea nitrogen:creatinine (UN:C) ratios indicative of severe nutritional restriction (≥3.5 mg:mg), all 4 study sites pooled, north-central Minnesota, winters 1993-94, 1994-95, 1995-96, and 2001-02.

UNDERSTANDING DIEL WINTER MOVEMENTS OF NORTHERN WHITE-TAILED DEER[1]

Christopher O. Kochanny[2] and Glenn D. DelGiudice

ABSTRACT

Northern deer (*Odocoileus* spp.) have evolved physiological, behavioral, and morphological adaptations for survival during winter. Among them is voluntary restriction in movement, which contributes to energy conservation and limits the impact of negative energy balance. To better understand the diel movements of white-tailed deer in winter, we deployed 14 global positioning system (GPS) collars on adult (≥1.5-year old) female deer during February–March 1999 and 2000. Collars collected 1 location per hour. A total of 10,329 (n = 11) 1-h movement segments were used to calculate hourly diel movement distances and rates. Deer were relatively active 24 h/day with mean daily movements of 2.9 ± 0.13 (SE) km and no difference (P = 0.45) between mean hourly diurnal (0600–1759 h, 153 ± 7 m) and nocturnal (1800–0559 h, 142 ± 12 m) distances moved. Mean total daily diurnal and nocturnal distances moved (1.7 ± 0.1 vs. 1.4 ± 0.1 km) were different (P = 0.02). We observed no relations of ambient temperature or snow depth to mean movements during these 2 mild winters with minimal snow cover (mean weekly snow depths of 8–15 cm). A 70-kg doe requires an estimated 0.049 kcal/ m of energy to travel in ≤ 18 cm of snow and 0.110 kcal/ m (or 2.25 times more energy) to travel in 40 cm of snow. For GPS-collared deer in our study making estimated daily minimum movements of 2.9 km, the daily energy cost for travel alone in shallow snow (≤ 18 cm) was 142 kcal, but would have been an estimated 319 kcal during a more severe winter with 40 cm of snow. Over an entire winter (1 Dec–31 Mar), the difference in cumulative energetic impact would have amounted to an estimated 21,417 kcal. The relatively low cost of movement during winters 1999 and 2000 would largely explain the relatively high activity of deer diurnally and nocturnally. Despite reasonably similar diurnal and nocturnal movements, deer may show temporal variation in the use of space and winter habitat, particularly relative to the nocturnal thermal benefits of conifer cover. Knowledge of the relation of winter cover on the landscape to the winter diel movements of deer is essential to a fuller understanding of their activity patterns and habitat requirements.

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COMPARING GLOBAL POSITIONING SYSTEM AND VERY HIGH FREQUENCY TELEMETRY HOME RANGES OF WHITE-TAILED DEER[1]

Christopher O. Kochanny[2], Glenn D. DelGiudice, and John Fieberg

ABSTRACT

Use of Global Positioning System (GPS) collars on free-ranging ungulates overcomes many limitations of conventional very high frequency (VHF) telemetry and offers a practical means of studying space use and home range estimation. To better understand winter home ranges of white-tailed deer (*Odocoileus virginianus*), we evaluated GPS collar performance, and compared GPS- and VHF-derived diurnal home ranges (for the same animals) and GPS-derived home range estimates for diurnal and nocturnal locations. Overall, the mean fix success rate of our GPS collars was 85% (range = 14–99%). Kernel density estimates of home range (using the 95% probability contour) derived from GPS and VHF locations were generally similar, as were GPS-derived diurnal and nocturnal home ranges. Overlap indices between GPS and VHF utilization distributions (UDs) ranged from 0.49 to 0.78 for the Volume of Intersection (VI) index and from 0.67 to 0.94 for Bhattacharyya's Affinity (BA); overlap indices for GPS-diurnal and nocturnal UDs ranged from 0.29 to 0.81 for VI and from 0.56 to 0.94 for BA. Despite similarities of home ranges estimated from GPS versus VHF locations and GPS-diurnal versus nocturnal locations, our data also indicate that differences may have important implications for studies focused on deer use of space, habitat, and resources at a finer scale.

[1]Journal of Wildlife Management (2009) 73: *In press*.
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COMPARISON OF NATIVE GRASSLAND MANAGEMENT TREATMENTS TO SPRING PRESCRIBED BURNS, A PILOT STUDY

David Rave, Kevin Kotts, and John Fieberg

SUMMARY OF FINDINGS

We conducted a pilot study to measure the response of restored native grasslands to: (1) grazing; (2) fall biomass harvest; and (3) spring prescribed burning. Some fields included both a fall biomass harvest and a spring controlled burn within different portions of the same field. Fields were located on Wildlife Management Areas (WMAs) or Waterfowl Production Areas (WPAs) in Working Land Initiative Focus Areas of Grant, Kandiyohi, Pope, and Stevens Counties. We conducted visual obstruction measurements, Daubenmire frame analysis, and we measured litter depth and vegetation height in all study fields. We also examined temporary and seasonal wetlands in bioharvested fields and recorded wetland type, and waterfowl presence. Vegetation characteristics varied considerably from field to field, and treatments (grazing, biomass harvest, burning) were not randomly assigned. Thus, comparisons involving fields undergoing a single treatment (e.g., grazing or fall biomass harvest only) were less informative than within-field comparisons involving biomass harvest and burn treatments. In these latter fields, biomass harvested and burned subplots appeared similar in most vegetative characteristics. We intend to survey vegetation in additional biomass harvest/burn field combinations in 2009, and will continue to survey all fields through summer 2011.

INTRODUCTION

Minnesota's Draft Grassland Biomass/Bioenergy Harvest on WMAs and Aquatic Management Areas (AMAs) states, "Grassland biomass harvest from WMAs and AMAs shall be in concert with fish and wildlife habitat management activities, consistent with the habitat or wildlife species management goals and habitat management objectives for each individual WMA/AMA." Further, Sample and Mossman (1997) found that differences in habitat structure are likely more important to bird communities than differences in vegetative species composition. They recommend that the following features of grassland habitat are important to grassland nesting birds: vegetation height and density, height and cover of woody vegetation, litter depth and cover, standing residual (dead) and live herbaceous cover, and ratio of grass vs. forb cover. However, the response of native grassland stands on WMAs and AMAs to grassland biomass harvest is unknown. We conducted a pilot study during the summer of 2008 with the following objectives:

- to establish and test protocols for sampling vegetation; and
- to determine sample sizes necessary to have a high probability of detecting meaningful differences in vegetative characteristics among treatment groups in a subsequent 3-year study.

STUDY AREA

The pilot study was primarily conducted in Working Land Initiative (MNDNR unpublished brochure;http://files.dnr.state.mn.us/assistance/backyard/privatelandsprogram/working-lands-ini.pdf) Focus Areas within Grant, Stevens, and Pope Counties. Following the pilot year, the proposed study area will include Working Land Focus Areas in much of the prairie portion of Minnesota (Figure 1). Fields sampled during the pilot study were all located on state managed Wildlife Management Areas (WMAs) or federally managed Waterfowl Production Areas (WPAs). Study sites consisted of 4 with grazed, 5 with bioharvested, and 3 with both bioharvested and burned fields.

METHODS

We compared the response of restored native grasslands to (1) grazing, (2) fall biomass harvest (hayed) and (3) spring prescribed burning (control) Where possible, the fall biomass harvest versus spring prescribed burn comparison was made within different portions (subplots) of the same field. Visual obstruction measurements (VOMs, Robel et al. 1970) were taken every 2 weeks from early June through mid-August in grazed, hayed, and spring burned portions of each field following methods described by Zicus et al. (2006). Three VOM sample stations were established at the 3 quarter points along the longest straight-line diagonal across each field. GIS locations were permanently marked with stakes to define starting and sampling points. Each station had 4 sampling points located 20 m north, east, south, and west of a starting point. At each field sampling point, vegetation height and density was measured in each cardinal direction. This provided 48 VOMs from grazed, hayed and spring burned portions of each field on a given date.

A Daubenmire square (Daubenmire 1959) was used to determine coverage by various species across grazed, hayed, and burned fields. To sample, one corner of each field was randomly selected and, using a compass, a transect was walked across the field. The $1m^2$ Daubenmire frame was placed on the ground every 25 paces, and each plant species (and % coverage within the frame) that comprised $\geq$ 10% of the total number of individual plants within the frame was recorded. This procedure was repeated 10 times in each treated field every 2 weeks.

Litter depth (nearest 1mm) and vegetation height (nearest 0.5 dm) were also measured at 10 locations along the VOM transect in grazed, hayed, and burned portions of each field every 2 weeks. While walking the VOM transect, all exotic and woody species present were recorded, and the amount of these species in each field will be estimated using distance sampling (Buckland et al. 2004).

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

RESULTS

During the pilot study year, we measured vegetation on 4 sites with grazed fields, 5 sites with hayed fields, and 3 sites with both hayed and burned fields. Vegetative characteristics varied considerably among fields in the same treatment group (e.g., see Figures 2 and 3 for grazed fields), but were largely similar in hayed and burned subplots within the Eldorado and Grace Marsh fields (Figures 4-8). Vegetation was taller (with larger VOM readings), litter depth was greater, and a higher number of species were located in the hayed treatment subplot than the burned subplot at Klason.

We examined 12 seasonal and temporary wetlands in mid-April that had been at least partially harvested during the biomass treatment in fall 2007. One wetland remained closed (cover type 1), 4 had open scattered vegetation (cover type 2), 3 had an open central expanse of water (cover type 3), and 4 were completely open (cover type 4). Six wetlands had dabbling duck pairs present when visited in spring 2008.

DISCUSSION

The Minnesota Department of Natural Resources acquires and manages Wildlife Management Areas primarily to establish and maintain optimal population levels of wildlife while maintaining ecological diversity; maintaining or restoring natural communities and ecological processes; and maintaining or enhancing populations of native species (including uncommon species and state- and federally-listed species; The Draft Grassland Biomass/Bioenergy Harvest on WMAs & AMAs directive, unpublished MNDNR publication). Prior to settlement and implementation of agriculture, natural disturbance in the form of fire and grazing maintained native grassland diversity and productivity (Anderson 1990). Wildlife managers have traditionally used spring prescribed burns to simulate these natural disturbances (K. Kotts, personal communication). However, there are a variety of management options available to wildlife managers to create disturbances in native grass stands. These options are not typically the first choice of managers; likely because there is little known about the response of native grass stands to these treatments. Our study is designed to compare the vegetative response of 3 management options for disturbing native grass stands.

Historically, the major factors influencing grassland ecosystems were fire, grazing by herbivores, and climatic variations (Kirsch et al. 1978). 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 burned versus non-burned 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 indicated the positive relationship between upland nesting birds and grassland disturbance from 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.

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

Management of native grass stands has become an important component of wildlife management in prairie portions of Minnesota (Kotts, personal communication). Historically, spring prescribed burning has been the management option most often 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 Conservation Reserve Program or Wetland Reserve Program, 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, soil type) among grazing treatment fields proved challenging. Vegetative characteristics varied considerably among fields, treatments were not randomly assigned, and there was no within field comparison in grazed fields. Based on results from this pilot, it is unlikely we can learn much about the effects of grazing on vegetation. Therefore, we will likely drop the grazing component of this study in future years, and concentrate our efforts on the difference between bioharvested fields and spring controlled burn fields.

Differences in vegetation characteristics between bioharvested and burned fields were greatest at Klason WMA, possibly as a result of fields being burned at a later date at Klason (May 18, 2008) than at Grace Marsh WMA (April 25, 2008), or Eldorado WMA (May 9, 2008). The biomass harvest treatments on all 3 WMAs occurred in the fall of 2007, after vegetative growth had finished. However, vegetation began growing in the spring shortly after snow melt. Burning removes all old and new vegetation from the burned field. Therefore, the later in the spring the treatment field is burned, the larger the difference in vegetative growth between the hayed and burned portions of the same field would be expected.

The removal of wetland vegetation in the fall is a promising way to open choked wetlands, making them available to waterbirds such as dabbling ducks, geese, swans and shorebirds. Fall wetland conditions play an important role in determining how successful this technique will be. Wetlands must be fairly dry when the haying occurs to allow equipment to harvest vegetation within the wetland basin. We will continue to monitor the basins that were harvested in 2007 to document the duration of benefit from fall biomass harvest.

ACKNOWLEDGEMENTS

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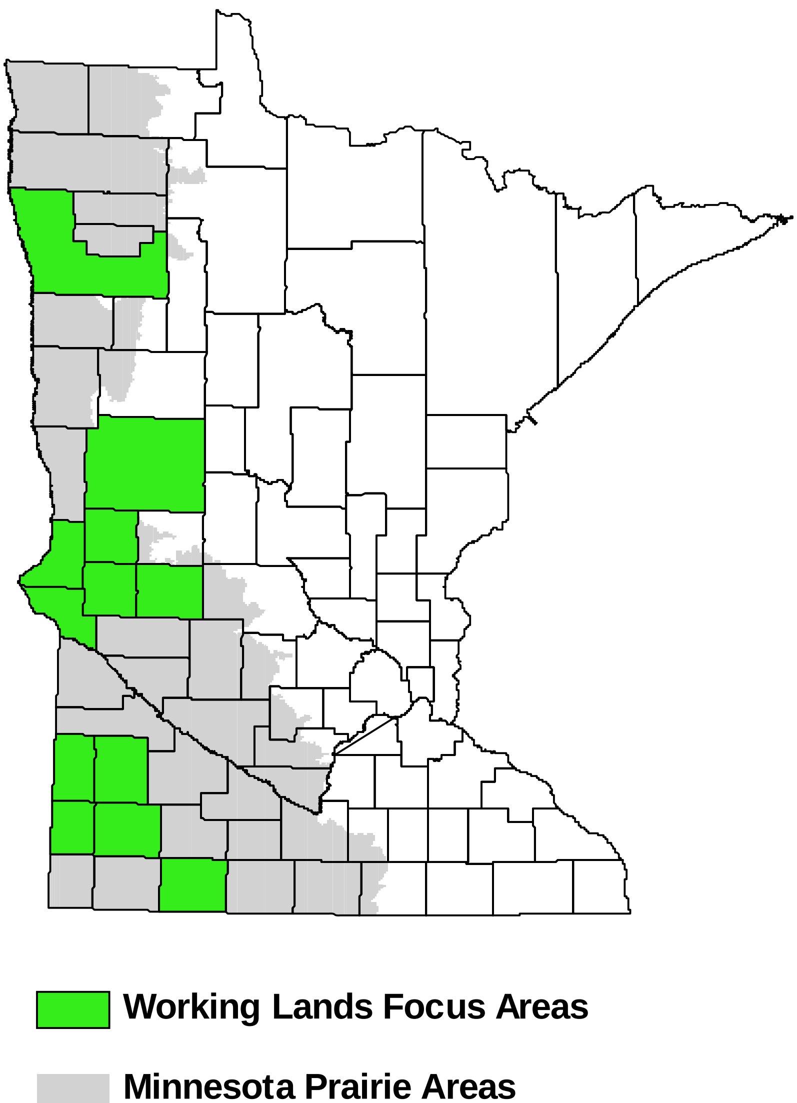

Figure 1. Minnesota Counties showing prairie areas and Working Lands Initiative focus areas, 2008.

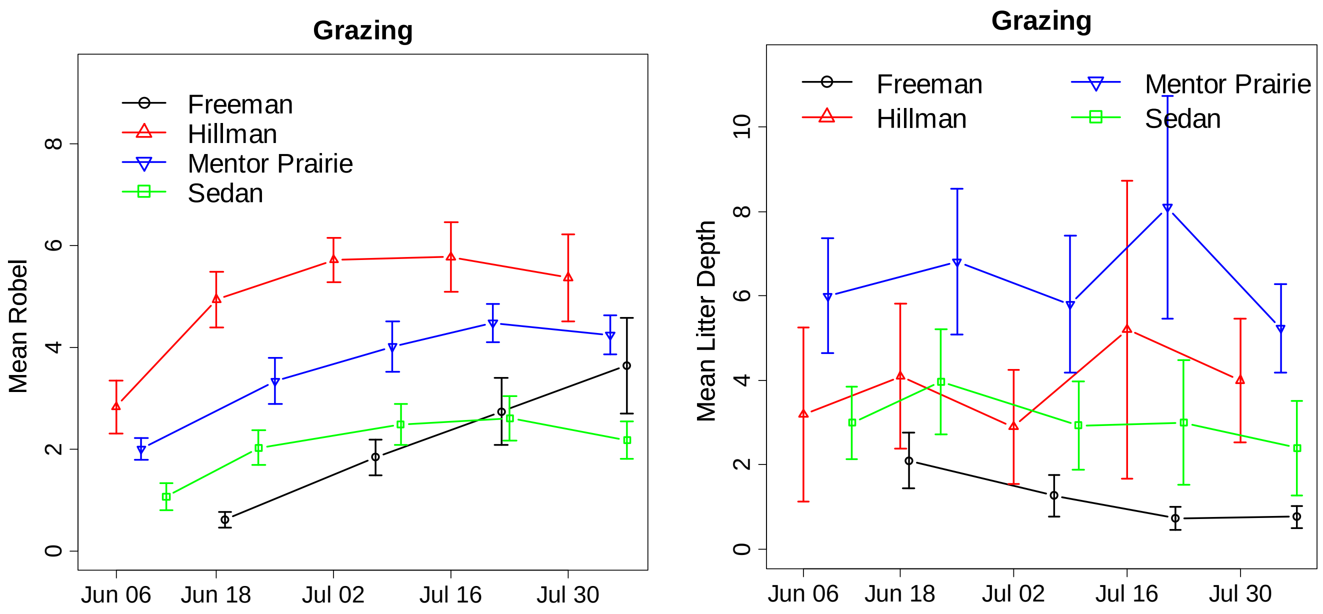

Figure 2. Comparison of mean Robel measurements (dm) and mean litter depth measurements (cm) across 4 units (Federal Waterfowl Production Areas and State Wildlife Management Areas) grazed in summer 2007 in west-central Minnesota.

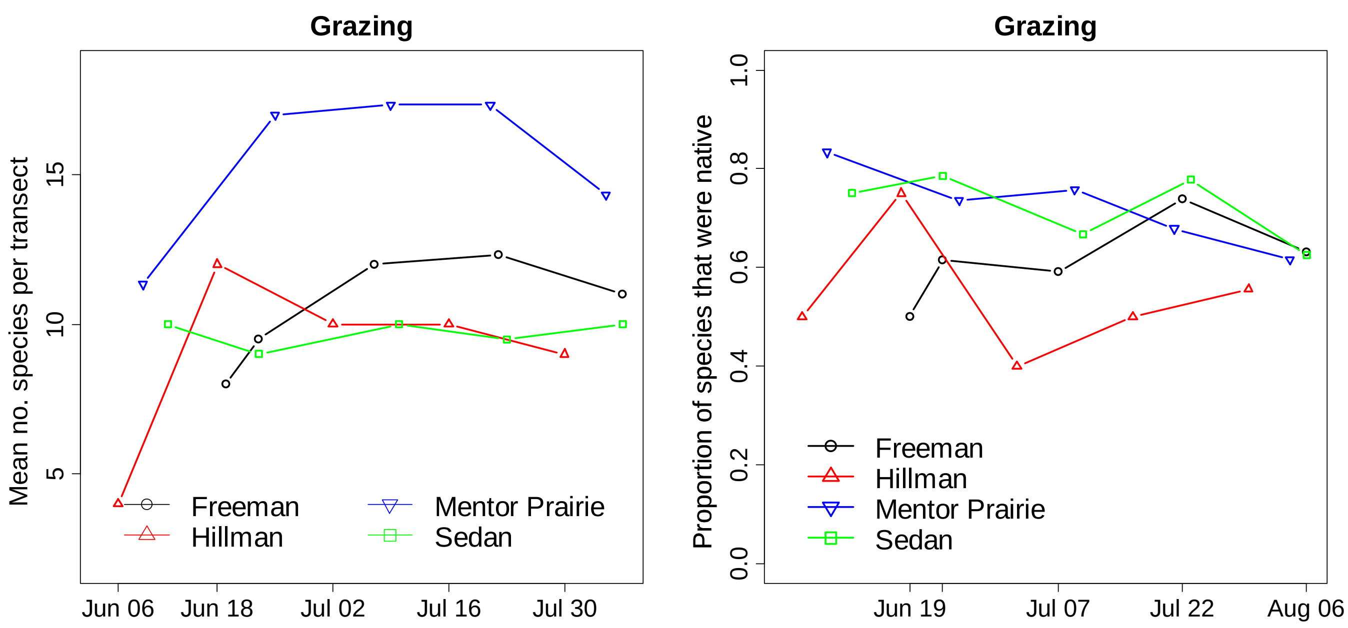

Figure 3. Comparison of mean number of species per transect and proportion of species that were native across 4 units grazed in summer 2007 (Federal Waterfowl Production Areas and State Wildlife Management Areas) in west-central Minnesota, summer 2008.

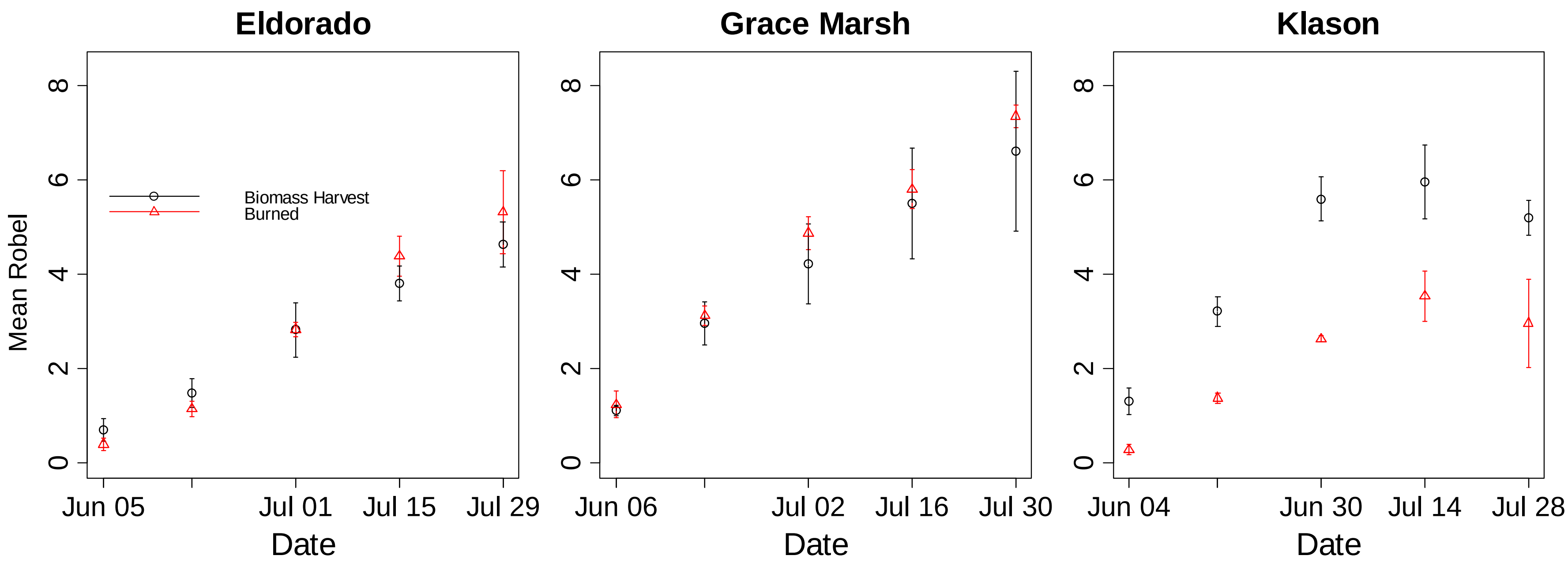

Figure 4. Comparison of mean Robel measurements (dm) between 2 treatments (a fall 2007 Biomass harvest and a spring 2008 prescribed burn) in the same restored native grass field on 3 State Wildlife Management Areas in west-central Minnesota, summer 2008.

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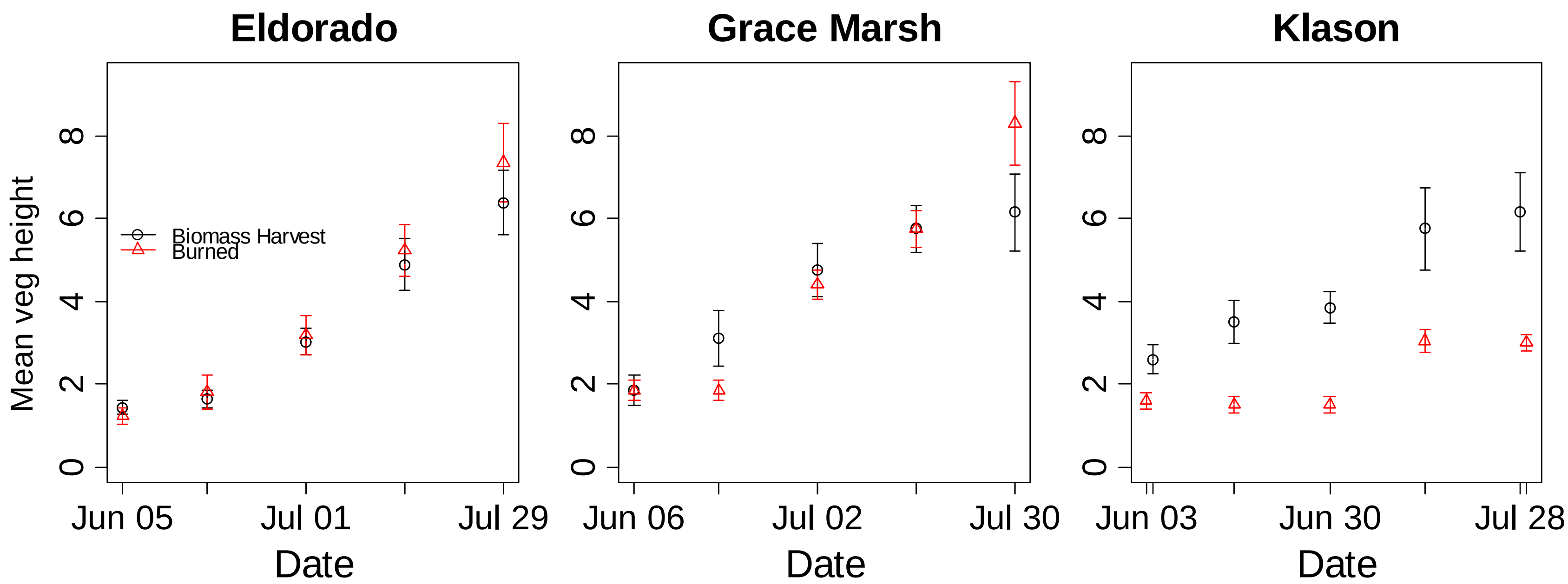

Figure 5. Comparison of mean vegetation height (dm) between 2 treatments (a fall 2007 Biomass harvest and a spring 2008 prescribed burn) in the same restored native grass field on 3 State Wildlife Management Areas in west-central Minnesota, summer 2008.

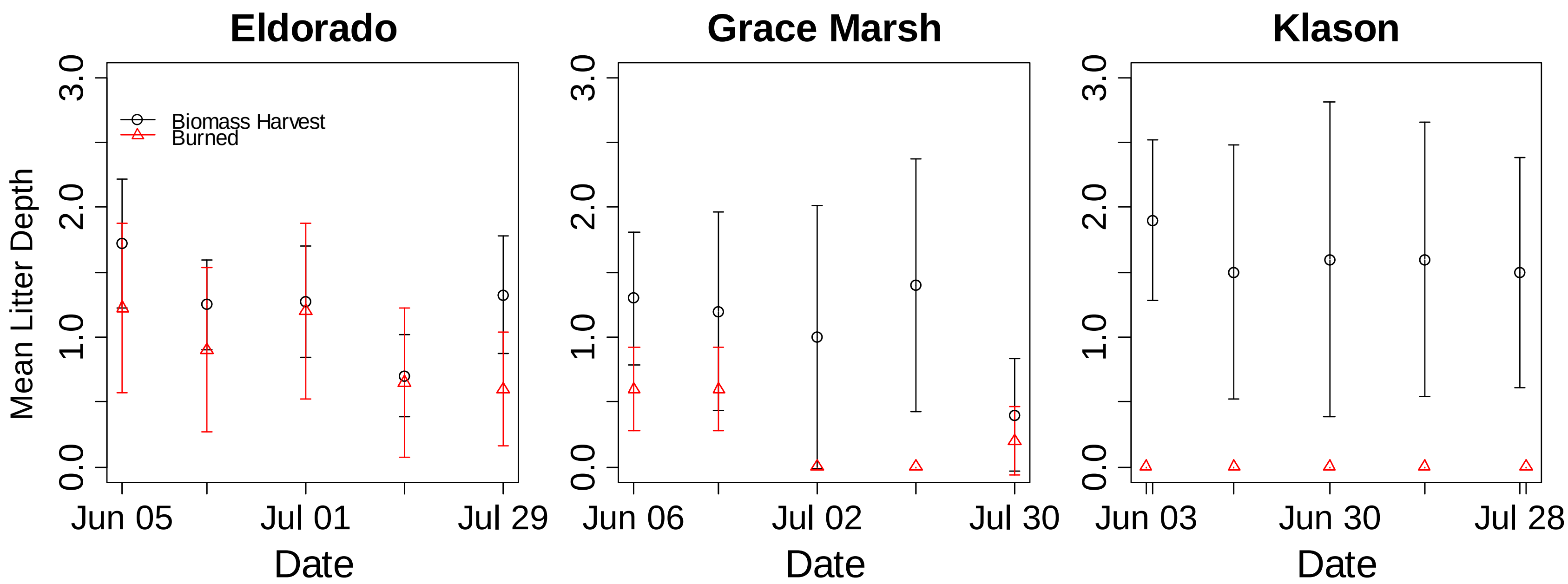

Figure 6. Comparison of mean litter depth (cm) between 2 treatments (a fall 2007 Biomass harvest and a spring 2008 prescribed burn) in the same restored native grass field on 3 State Wildlife Management Areas in west-central Minnesota, summer 2008.

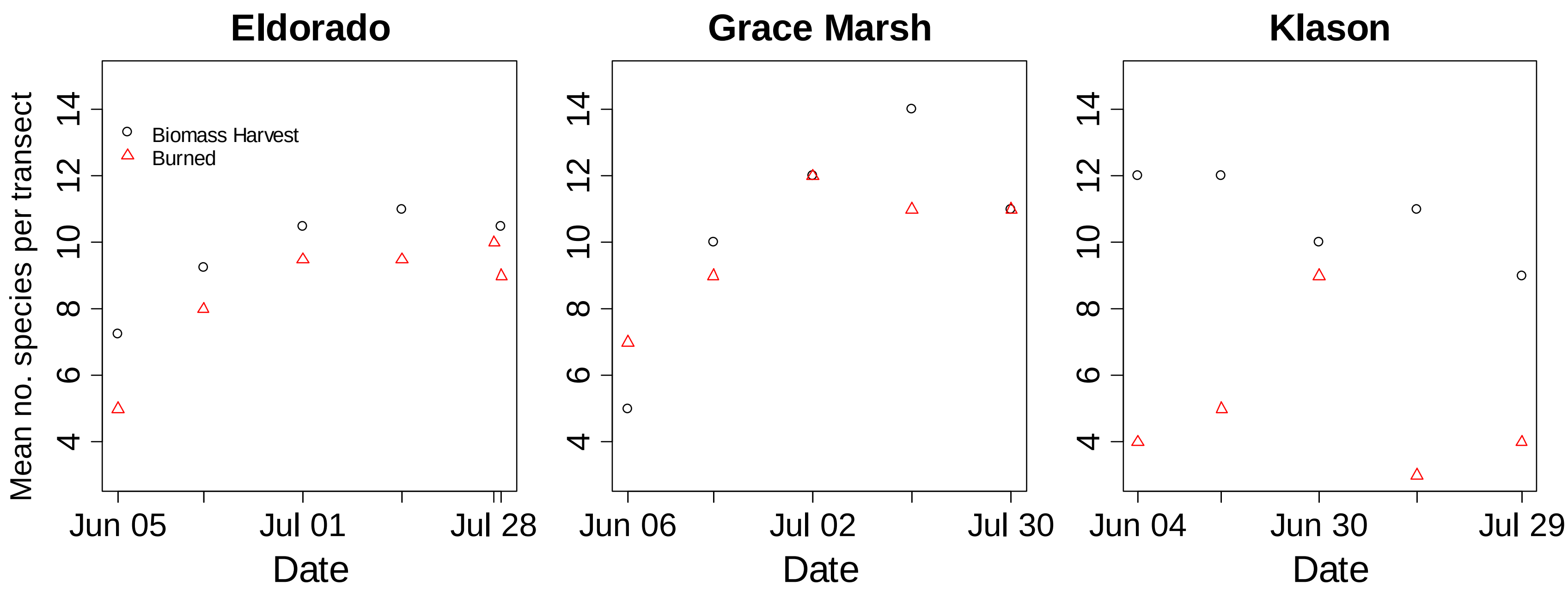

Figure 7. Comparison of mean number of plant species per transect between 2 treatments (a fall 2007 Biomass harvest and a spring 2008 prescribed burn) in the same restored native grass field on 3 State Wildlife Management Areas in west-central Minnesota, summer 2008.

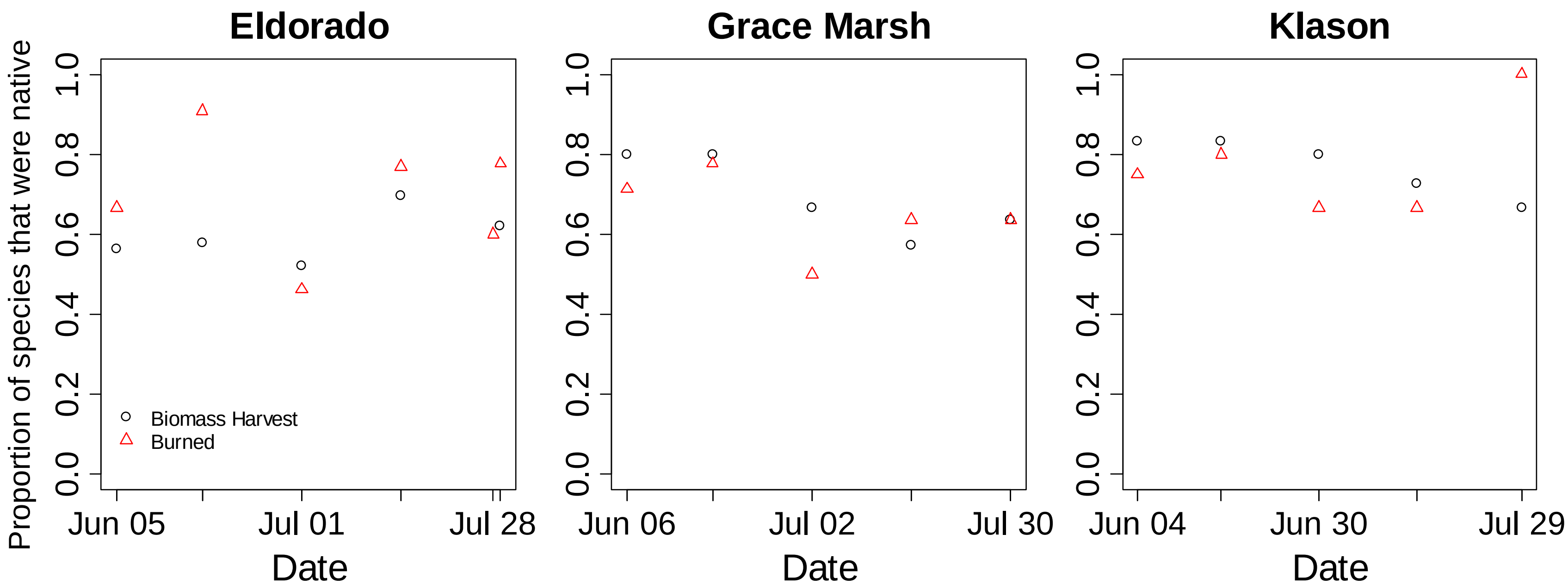

Figure 8. Comparison of the proportion of native plant species between 2 treatments (a fall 2007 Biomass harvest and a spring 2008 prescribed burn) in the same restored native grass field on 3 State Wildlife Management Areas in west-central Minnesota, summer 2008.

2008 RING-NECKED DUCK BREEDING PAIR SURVEY

Christine M. Sousa, David P. Rave, Michael C. Zicus, John R. Fieberg, John H. Giudice, and Robert G. Wright

SUMMARY OF FINDINGS

A pilot study was conducted in 2004-2006 to develop a survey for Minnesota's ring-necked duck (*Aythya collaris*) breeding population, because little was known about its distribution and relative abundance. We employed the survey design and methods developed during the pilot study (Zicus et al. 2006) to estimate the size of the population in 2007. In 2008, the survey was conducted again but we surveyed only 3 of 6 geographic strata due to budget limitations. The helicopter-based counts entailed 6 survey-crew days from 9-17 June totaling ~35 hrs of flight time. The survey included the portion of Minnesota considered primary breeding range. The 2008 breeding population was estimated to be ~9,500 indicated breeding pairs and ~19,500 birds, which are similar estimates to 2006 and 2007 for the 3 geographic strata.

INTRODUCTION

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

A 3-year pilot study was initiated to develop a ring-necked duck breeding pair survey (Zicus et al. 2006), and 2007 represented the first year of an operational survey. In 2008, the survey was conducted again but was reduced in scope due to budget limitations. The primary objectives of this survey are to estimate breeding pair numbers and monitor population trends.

METHODS

Similar to the pilot study, we estimated number of breeding pairs and population size within a stratified random sample of survey plots using 2 stratification variables: (1) Ecological Classification System (ECS) sections; and (2) presumed nesting-cover availability (i.e., a surrogate for predicted breeding ring-necked duck density, Zicus et al. 2006). Surveys were restricted to an area believed to be primary breeding range of ring-necked ducks for logistical efficiency (Zicus et al. 2005). Public Land Survey (PLS) sections (~2.6-km^2 plots, range = 1.2 – 3.0 km^2) were used as primary sampling-units. The PLS sections at the periphery of the survey area that were <121 ha in size were removed from the sampling frame to reduce the probability of selecting these small plots. From 2004-2007, 6 ECS sections were surveyed (Western and Southern Superior Uplands [sections combined for the survey]; Northern Superior Uplands; Northern Minnesota and Ontario Peatlands; Northern Minnesota Drift and Lake Plains; Minnesota and Northeast Iowa Morainal; and Lake Agassiz, Aspen Parklands). In 2008, 3 of the ECS sections were dropped from the survey; ECS sections sampled for the survey included:

(1) Northern Minnesota Drift and Lake Plains; (2) Minnesota and Northeast Iowa Morainal; and (3) Lake Agassiz, Aspen Parklands, (Figure 1).

ArcInfo and ArcView software (Environmental Systems Research Institute, Inc., Redlands, California) were used to assign each PLS section to 1 of 4 model-based habitat classes (Zicus et al. 2006). We used the same habitat class definitions that were used for stratification in the last pilot year (i.e., 2006, Table 1). Plots with at least the median amount of nesting cover were assumed to have high potential (habitat class 1) for ring-necked ducks with plots having some nesting cover but less than the median amount assumed to have moderate potential (habitat class 2). Plots with no nesting cover but with near-shore water were assumed to have low potential (habitat class 3), and those without breeding habitat were assumed to have no potential (habitat class 4). In 2008, a stratified sampling design was used to estimate breeding ducks in the best ring-necked duck habitat (habitat class 1 and 2 plots), and the sampling frame consisted of 6 strata (i.e., 3 ECS sections x 2 habitat classes). We proportionally allocated 174 plots to the 6 strata (Zicus et al. 2005). In previous surveys we also used a 2-stage simple random sampling design to estimate population size in the remainder of the survey area (habitat class 3 and 4 plots). However, these areas were dropped from the survey due to limited funds.

As in previous years, we used a helicopter for the survey because visibility of ring-necked ducks from a fixed-wing airplane is poor in most ring-neck breeding habitats. For each plot, location, date, and time were recorded on data sheets as were all ring-necked ducks observed on study plots from the helicopter and their sex and social status (lone, paired, single-sex flock, mixed groups). Locations of these birds were also plotted on aerial photos. We considered pairs, lone males, and males in flocks of 2–5 to indicate breeding pairs (IBP; J. Lawrence, MNDNR, personal communication). The total breeding ground population in the survey area was considered to be twice the IBP plus the number of lone females, flocked females, mixed sex groups, and single-sex groups >5 birds. We used R programming language (R Development Core Team 2007) to estimate IBP and breeding population totals for habitat class 1 and 2 plots in each ECS section and the entire survey area.

RESULTS

In 2008, plots were well distributed throughout the study area (Figure 1). Most plots (108 plots) were located in the Northern Minnesota Drift and Lake Plains section, while the fewest plots (13 plots) were located in the Lake Agassiz, Aspen Parklands section (Table 2). The sampling rate was higher in the Lake Agassiz, Aspen Parklands section than the other 2 ECS sections (3.8% versus 1.5%; Table 2).

The survey was conducted 9–17 June and entailed 6 survey-crew days totaling ~35 hrs of flight time. A total of 296 ring-necked ducks were observed in 58 of 174 plots (Table 3). Overall, counts on occupied plots ranged from 1 to 25 birds (median = 3 birds/plot). Numbers of IBP on occupied plots ranged from 0 to 10 (median = 2 IBP/plot). Numbers of birds on occupied plots ranged from 0 to 28 ducks (median = 4 breeding birds/plot). Of the birds observed, 65% were classified as pairs, 14% flocked males, 12% lone males, 6% groups, 2% lone females, and <1% flocked females. Of IBP, 56% were classified as pairs, 24% flocked males, and 20% lone males. These IBP ratios suggest that survey timing was reasonably good for estimating the local breeding population. Observed pairs represented 56% of the IBP tallied during the 2008 survey, which was similar to the 2004 and 2007 surveys and slightly higher than the 2005 and 2006 surveys (Figure 2).

Estimated IBP in the survey area was 9,439 pairs (SE = 1,582 pairs; Table 4, Figure 3A). The estimated breeding ground population of ring-necked ducks in the survey area was 19,488 birds (SE = 3,240 birds; Table 4, Figure 3B). Because of sampling frame changes in 2008, estimates from 2006 and 2007 were re-calculated with a 3 ECS sampling frame. Data from 2004 and 2005 were not re-calculated, because habitat classifications have also changed since those surveys were conducted. Estimates (IBP and breeding population) from 2008 were slightly higher than 2007 and slightly lower than 2006 but was within the error of both prior surveys. The breeding population ranged from a high of 4,948 pairs and 10,264 breeding birds in the Northern Minnesota Drift and Lake Plains section to a low of 803 pairs and 1,846 breeding birds and in the Lake Agassiz, Aspen Parklands section (Table 5).

In 2008, pair densities on habitat classes 1 and 2 ranged from 0.04 pairs/km^2 in the Lake Agassiz, Aspen Parklands section to 0.23 pairs/km^2 in the Northern Minnesota Drift and Lake Plains section. Densities of breeding birds ranged from 0.09 birds/km^2 in the Lake Agassiz, Aspen Parklands section to 0.48 birds/km^2 in the Northern Minnesota Drift and Lake Plains section. Compared to previous years, densities of birds and IBP were slightly lower in the Northern Minnesota Drift and Lake Plains section, higher in the Minnesota and Northeast Iowa Morainal section, and within the range of past years in the Lake Agassiz, Aspen Parklands section (Figure 4).

The survey was not designed explicitly to describe the distribution of breeding ring-necked ducks, but observations accumulated thus far have improved our knowledge of ring-necked duck distribution in the survey area (Figures 5 and 6). Most of the IBP and breeding population to date have been located along the north and northwest margin of the Northern Minnesota Drift and Lake Plains section. Another concentration of breeding ring-necked ducks is found at Agassiz National Wildlife Refuge in the center of the Lake Agassiz, Aspen Parklands section. Very few ring-necked ducks have been observed along the southern margin of the study area, although there have been a number of survey plots in this area.

DISCUSSION

Survey timing is considered optimal when most birds are counted as pairs and not in flocks (Smith 1995). Survey dates in 2008 appeared appropriate, because 56% of the indicated pairs were counted as paired birds. The breeding population appears to be relatively stable in the few years that this population has been surveyed, remaining between 18,000 and 22,000 breeding birds in the 3 ECS sampling frame. The Northern Minnesota Drift and Lake Plains section continues to have the majority of the ring-necked duck breeding population. These surveys are planned to continue in 2009.

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

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Table 1. Habitat classes assigned to Public Land Survey section plots in the Minnesota ring-necked duck breeding pair survey area, June 2004 – 2008.

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

[a]Plots are Public Land Survey sections. MNGAP = Minnesota GAP level 4 land cover data. Class 10 = lowlands with <10% tree crown cover and >33% cover of low-growing deciduous woody plants such as alders and willows. Class 12 = lakes, streams, and open-water wetlands. Class 13 = water bodies whose surface is covered by floating vegetation. Class 14 = wetlands with <10% tree crown cover that is dominated by emergent herbaceous vegetation such as fine-leaf sedges. Class 15 = wetlands with <10% tree crown cover that is dominated by emergent herbaceous vegetation such as broad-leaf sedges and/or cattails.

[b]Habitat class definitions in 2005, 2006, 2007, and 2008 were the same, but MNGAP class 10, 14, and 15 cover associated with lakes having a General or Recreational Development classification under the Minnesota Shoreland Zoning ordinance was not considered nesting cover in 2006, 2007, and 2008.

Table 2. Sampling rates in the habitat class 1 and 2 strata by Ecological Classification System (ECS) section for Minnesota's ring-necked duck breeding- pair survey, June 2004 – 2008.

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

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

Table 3. Survey results for habitat class 1 and 2 strata in the Minnesota ring-necked duck breeding pair survey area, 2004-2008.

			Birds			IBP			Breeding birds		
Year	No. of plots	No. plots with birds (%)	Total	Per plot	Per occupied plot	Total	Per plot	Per occupied plot	Total	Per plot	Per occupied plot
2004	200	50 (25)	278	1.39	5.56	160	0.58	3.20	353	1.77	7.06
2005	230	37 (16)	147	0.64	3.97	92	0.63	2.49	218	0.95	5.89
2006	200	50 (25)	279	1.40	5.58	167	0.60	3.34	375	1.88	7.50
2007	200	52 (26)	152	0.76	2.92	137	0.90	2.63	296	1.48	5.69
2008	174	58 (33)	296	1.70	5.10	173	0.58	2.98	364	2.09	6.28

Table 4. Estimated indicated breeding pairs (IBP) and breeding ground population size in the habitat class 1 and 2 strata in the Minnesota ring-necked duck breeding pair survey area, 2004-2008.

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

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

Table 5. Estimated indicated breeding pairs (IBP) and breeding ground population by Ecological Classification System (ECS) section in the habitat class 1 and 2 strata in the Minnesota ring-necked duck breeding pair survey area, June 2005-2008.

	IBP (CV [%])				Breeding ground population (CV [%])			
ECS section	2005	2006	2007	2008	2005	2006	2007	2008
W & S Superior Uplands[b]	444 (99.5[c])	669 (59.1)	671 (99.6)	-	889 (99.5[c])	1,338 (59.1)	1,342 (99.6)	-
Northern Superior Uplands	1,169 (46.8)	2,679 (33.7)	2,694 (46.5)	-	2,339 (46.8)	5,357 (33.7)	5,388 (46.5)	-
N Minnesota & Ontario Peatlands	239 (54.1[c])	1,572 (34.7)	717 (46.5)	-	477 (54.1[c])	4,076 (42.3)	1,434 (46.5)	-
N Minnesota Drift & Lake Plains	3,490 (33.0)	6,334 (31.5)	5,686 (26.0)	4,948 (24.6)	6,981 (33.0)	14,816 (29.6)	11,651 (25.4)	10,264 (24.3)
Minnesota & NE Iowa Morainal	918 (43.6)	2,102 (53.9)	2,118 (38.8)	3,689 (26.0)	4,122 (56.4)	4,204 (53.9)	4,236 (38.8)	7,377 (26.0)
Lake Agassiz, Aspen Parklands	1,235 (40.1[c])	1,414 (35.2)	902 (40.9)	803 (38.4)	2,471 (40.1[c])	2,829 (35.2)	1,976 (42.3)	1,846 (41.4)

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

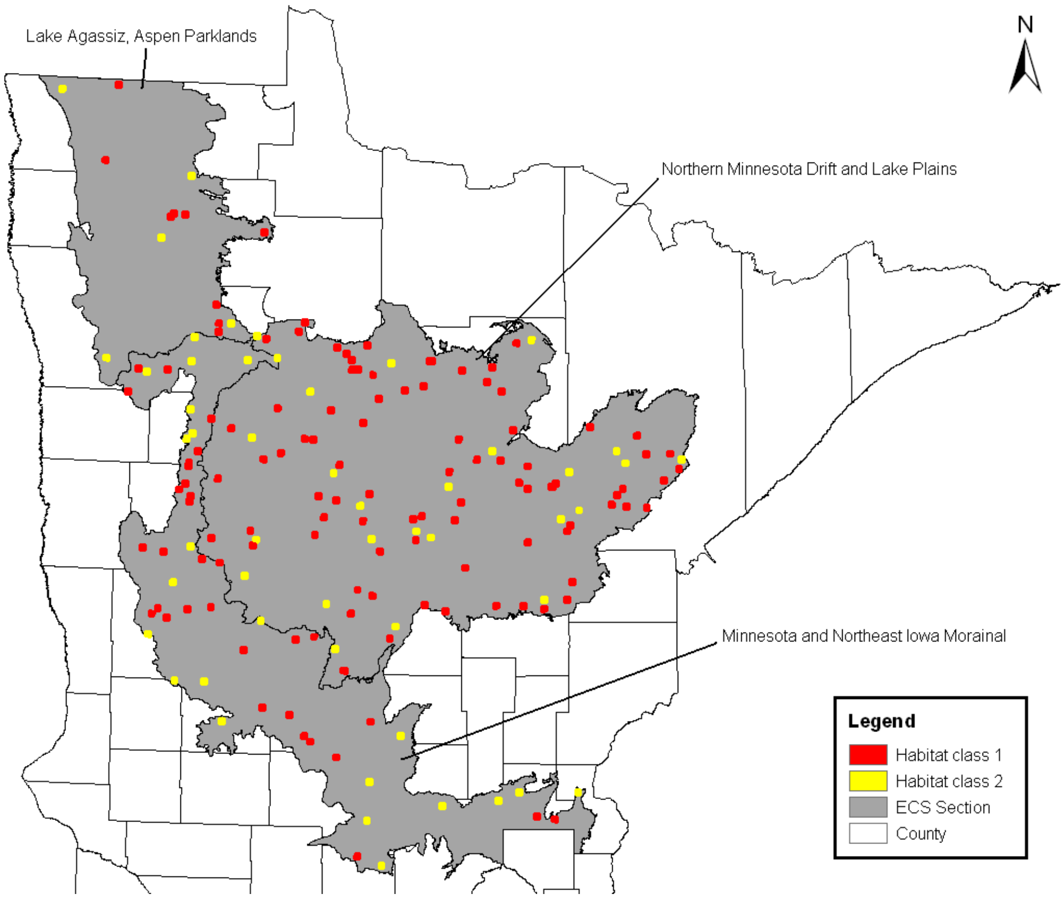


Figure 1. Study area with survey plots indicated by habitat class for Minnesota's 2008 ring-necked duck breeding pair survey.

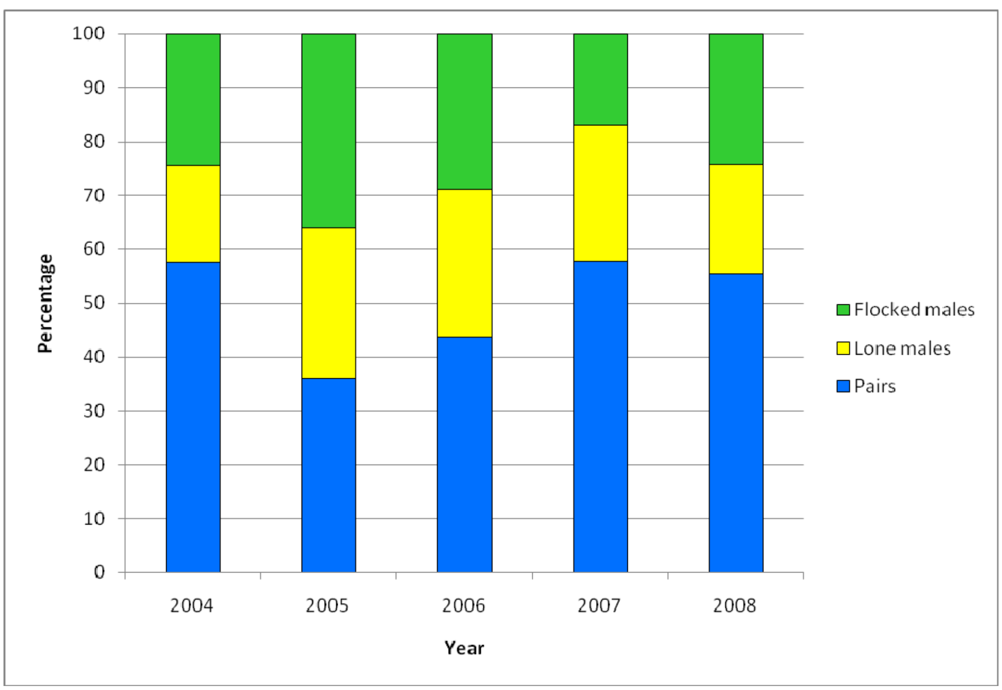
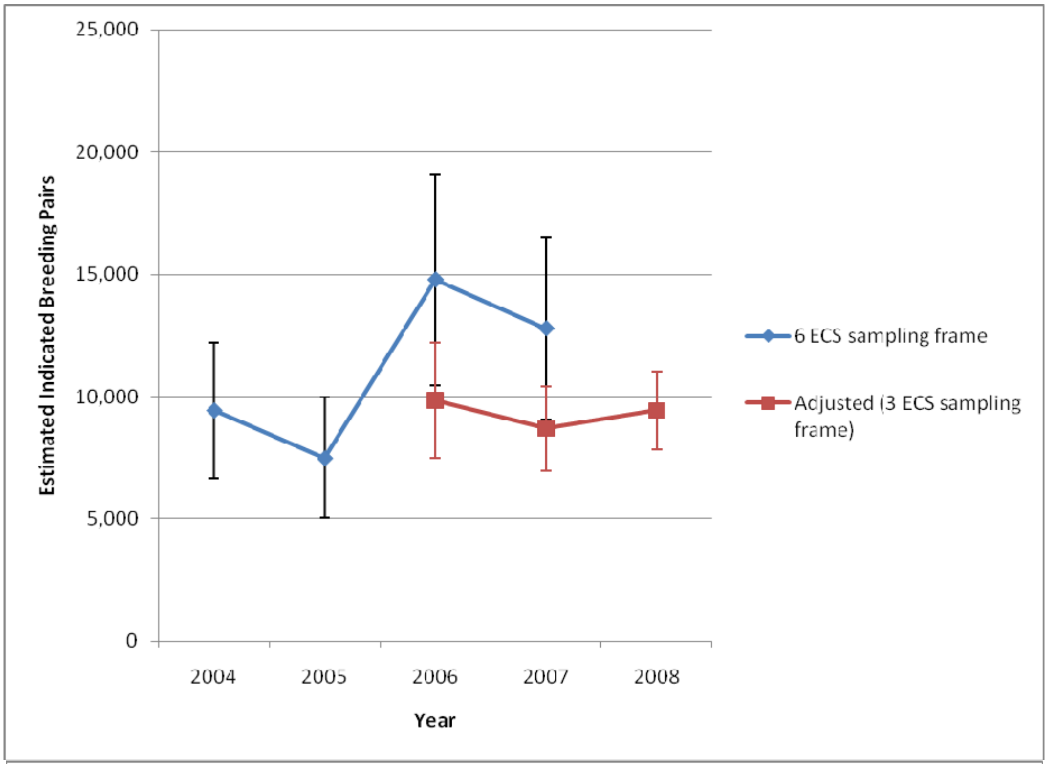


Figure 2. Social status of the indicated breeding pairs observed in the Minnesota ring-necked duck breeding pair survey area, June 2004-2008. Surveys were conducted 6 – 17 June 2004, 12 – 24 June 2005, 6 – 16 June 2006, 5 – 13 June 2006, and 9 - 17 June 2008.

A



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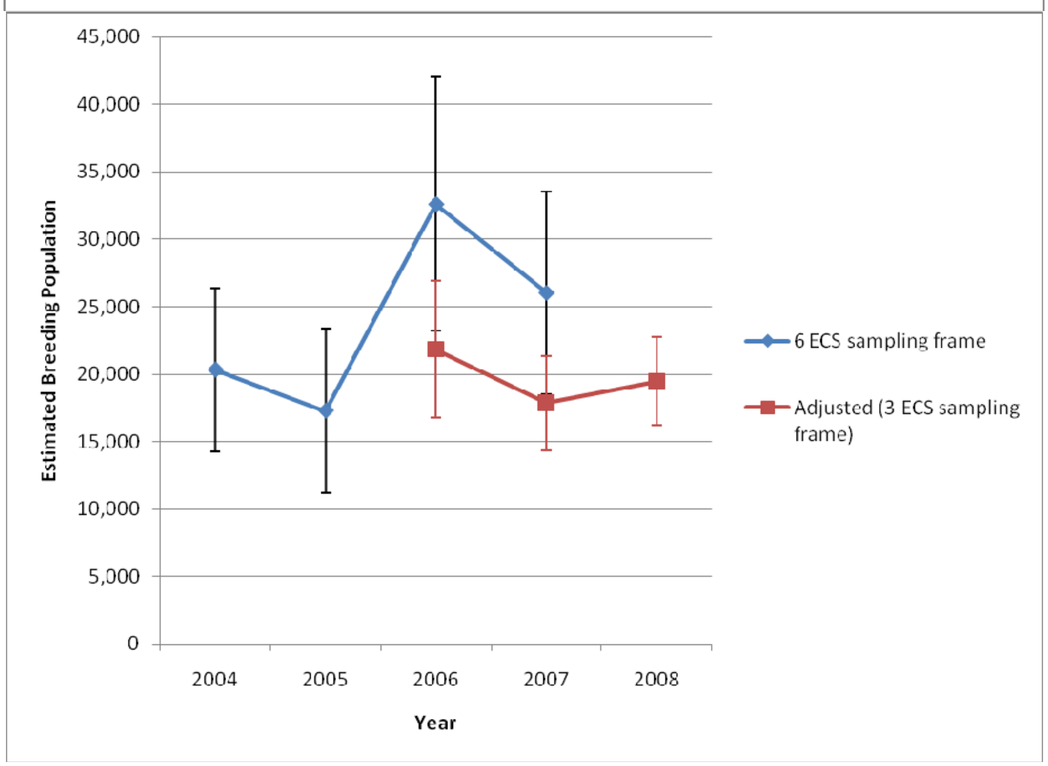
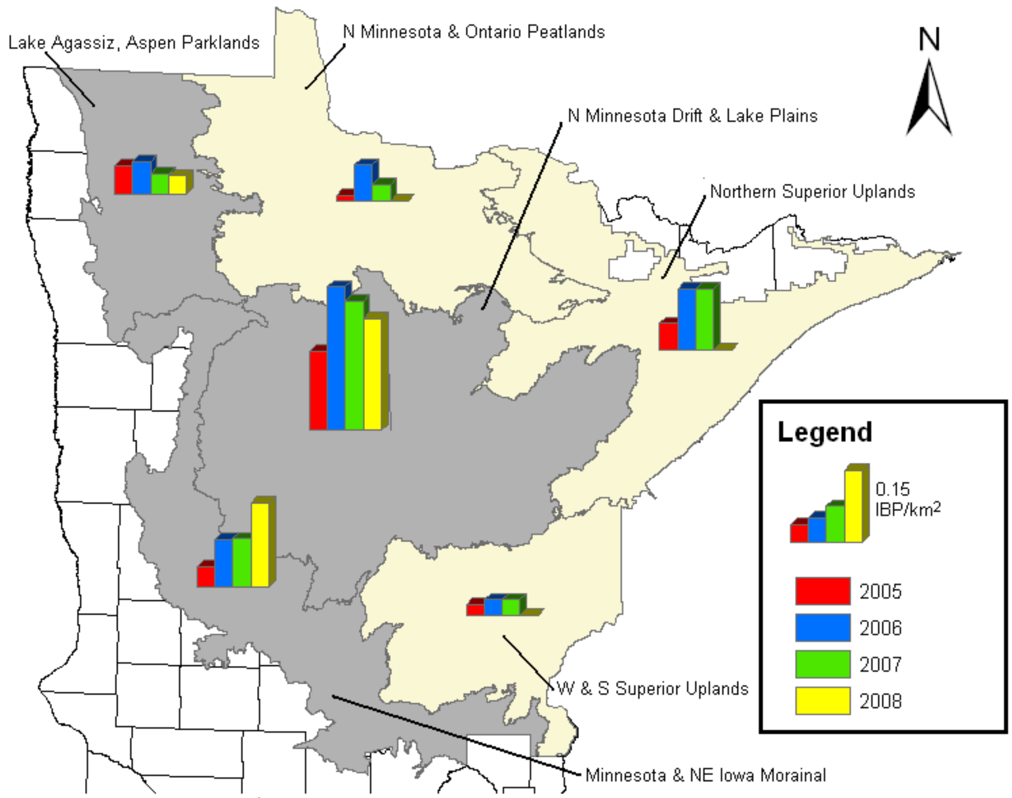


Figure 3. For the habitat class 1 and 2 strata (A) estimated indicated breeding pairs with SE bars and (B) estimated ring-necked duck breeding ground population with SE bars in the Minnesota ring-necked duck breeding pair survey area, June 2004-2008. Estimates were based on a stratified random sample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

A



B

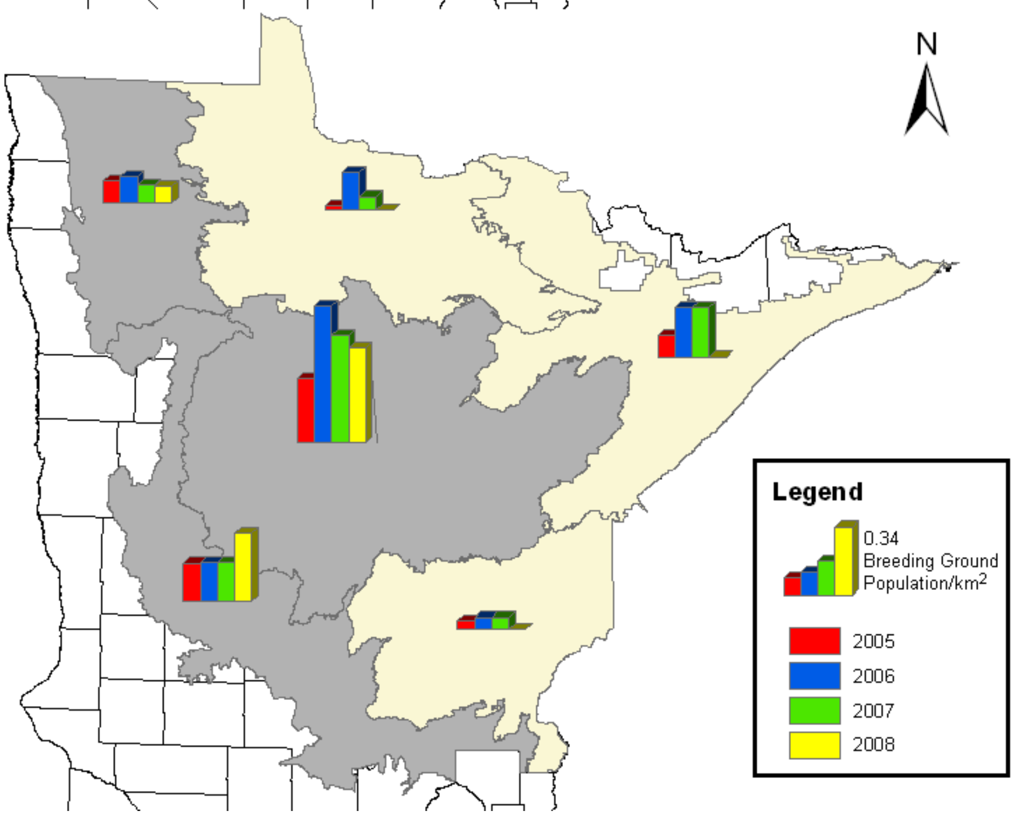


Figure 4. For the habitat class 1 and 2 strata (A) estimated indicated breeding pairs (IBP) per km^2 and (B) estimated breeding ground population per km^2 in the Minnesota ring-necked duck breeding pair survey area, June 2005-2008. Western and Southern Superior Uplands sections combined due to the small area of the Southern Superior Uplands occurring in the survey area.

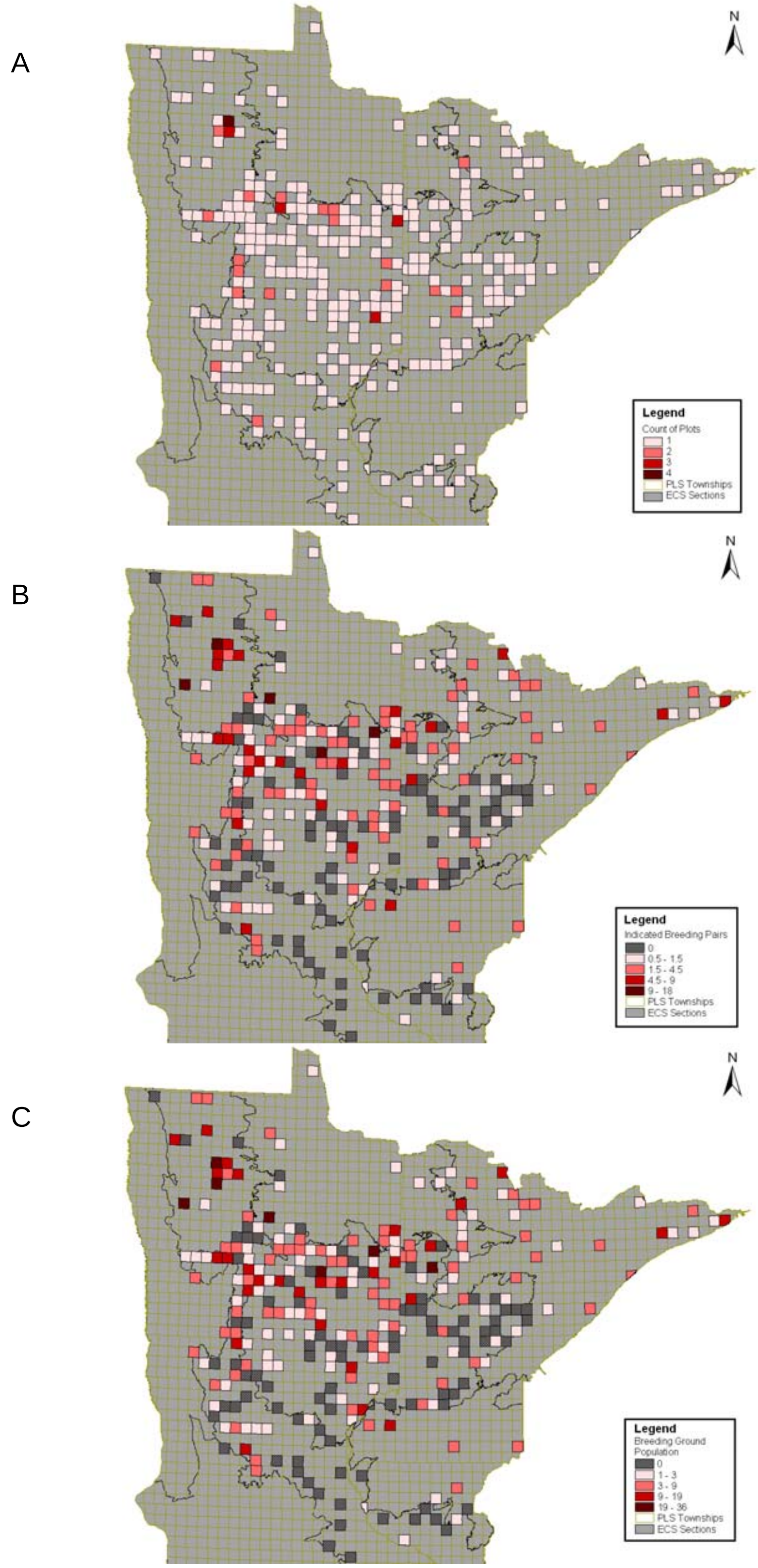


Figure 5. Maps are colored to indicate (A) total number of plots surveyed, (B) average number of indicated breeding pairs (IBP), and (C) average breeding ground population in each Public Land Survey (PLS) Township within the Minnesota ring-necked duck breeding pair survey area during 2004-2008. The Ecological Classification System (ECS) sections are also shown.

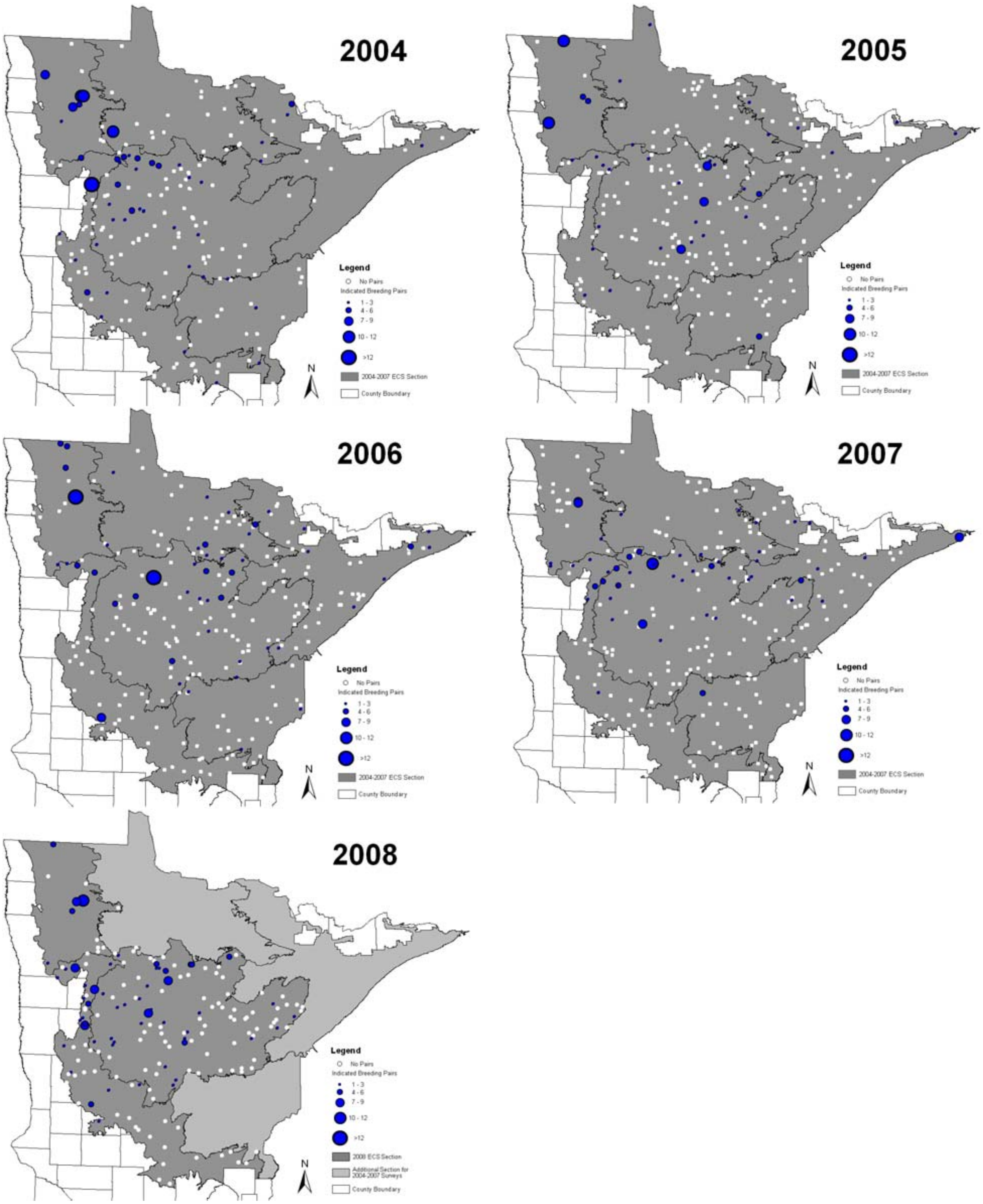


Figure 6. Plot locations and numbers of indicated breeding pairs observed on survey plots in the Minnesota ring-necked duck breeding pair survey area in June - 2008 (bottom left). White circles indicate plots where no indicated pairs were seen. Maximum number of indicated breeding pairs per plot was 10 pairs in 2008 (13 in 2004; 11 in 2005; 16 in 2006; 11 in 2007). The Ecological Classification System (ECS) sections are also shown.

NESTING ECOLOGY OF RING-NECKED DUCKS IN NORTHERN MINNESOTA

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

SUMMARY OF FINDINGS

The first field season focused on data collection and refining methods to increase the efficiency of nest searches. We relied on 2 datasets to identify ring-necked duck (*Aythya collaris*) nesting habitat: 1) ring-necked duck breeding survey data from 2004-2008 (Zicus et al. 2006); and 2) GAP data layers indicating the juxtaposition of emergent herbaceous vegetation or low woody plants with water and <10% tree crown cover. After lakes were identified, we flushed ring-necked duck hens off nests using multiple methods, including nest dragging, disturbing vegetation with bamboo poles on foot and from canoes, and with low flights in a helicopter (Heyland and Munro 1967, Johnson 1977, Kaminski 1979). We found 18 ring-necked duck nests in this first year. We will improve effectiveness of finding nests next year by scouting for pairs and lone males on wetlands before conducting nest searches.

OBJECTIVES

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

STUDY AREA

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

METHODS

We identified lakes with suitable habitat using data on ring-necked duck use in the spring. These data were obtained from a helicopter survey conducted in 2004-2008 and from ground surveys conducted on 14 lakes in the Bemidji area beginning in 1969. We then used these data to identify lakes with similar land cover attributes (GAP types 12 and 13 surrounded by GAP types 10, 14, and 15). We also attempted to search lakes in our study area with pairs or lone males in the 2008 ring-necked duck survey.

We searched emergent vegetation along wetland margins using bamboo poles and nest drags to locate nests. We also searched with a helicopter to determine whether search efficiency could be improved from that of efforts on the ground (Heyland and Munro 1967, Johnson 1977, Kaminski 1979). This method involved flying low in a helicopter to blow vegetation and flush hens off nests. We documented lakes in which hens, lone males, and pairs were observed. We also recorded nests of other species encountered during nest searches from the ground.

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. Nests were monitored weekly to determine fate (abandoned, depredated, or successful). At each nest, and at a random point located 25 m from each nest, we determined water depth, concealment using a Daubenmire frame and Robel pole (Daubenmire 1959, Robel et al. 1970), predominant vegetation (e.g.,

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cattail, sedge), distance to dry land, and distance to open water. Wetland size, distance to roads and dwellings, wetland class, and disturbance variables will be determined in GIS for use in models of nest survival.

Late in incubation, we trapped ring-necked duck females on nests with Weller traps (Weller 1957) to attach radiotransmitters. Because a surgical transmitter attachment method might be disruptive to incubating hens, we tried a bib-type transmitter attachment method with which we had had previous success in wood ducks (*Aix sponsa*, Montgomery 1985). This attachment method is faster and less invasive than surgical methods. Briefly, hens received a transmitter fastened to a Herculite® fabric bib with dental floss and superglue (total weight of 11 g). We modified the method used unsuccessfully by Sorenson (1989) by securing the bib more tightly and by preening the bib into the breast feathers as in Montgomery (1985). After the transmitter was in place, we trimmed any excess fabric so that feathers concealed the transmitter. We released birds at the edge of the wetland.

After the nest hatched, we monitored broods every 3-4 days. We conducted behavioral observations of hens with transmitters and hens without transmitters to determine whether behavior was altered by our methods. We continued to monitor hens after the brood-rearing period to examine hen survival until migration.

RESULTS

Nest Survival

We searched for nests from the ground on 39 wetlands a total of 73 times between 22 May-22 July 2008. We located 18 ring-necked duck nests on 10 of these wetlands. We also located nests of 6 mallards (*Anas platyrhynchos*), 4 American coots (*Fulica americana*), 2 blue-winged teals (*Anas discors*), a common loon (*Gavia immer*), a pied-billed grebe (*Podilymbus podiceps*), a red-necked grebe (*Podiceps grisegena*), a sora rail (*Porzana carolina*), and a Canada goose (*Branta canadensis*). Two black tern (*Chlidonias niger*) colonies were located, and many nests were associated with each colony. All nests were located on foot or from canoes. Helicopters flushed 16 hens in 4 hours, but many of these areas could not be searched due to difficulties obtaining landowner permission or because the bog mats could not withstand our weight without being submerged.

Of the 18 ring-necked duck nests, 8 hatched, 4 were depredated when found, 3 were depredated after they were found, and 3 nests were flooded by rising lake levels following rain events. Average clutch size was 9.1 $\pm$ 0.6 (range: 7-15) and 87.1 $\pm$ 0.1% of eggs hatched. We put transmitters on 8 hens late in incubation. Apparent nest success was 8/18, or 44%. Mayfield nest success for a 26-day period (Mendall 1958) was 37.3%. More thorough nest survival analyses will be conducted at the conclusion of the study, when sample sizes are larger. We will also analyze characteristics at each nest in subsequent reports.

Hen Survival

Of 8 birds radiomarked, 2 birds died during the brood-rearing period; one had a brood and the other lost her nest late in incubation. 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 continued to nest and rear broods after transmitter attachment, with the exception of one bird that lost her nest to flooding. Two more birds died during hunting season, but unfortunately we could not determine whether these mortalities were associated with hunting.

Brood and Duckling Survival

Of the 8 broods monitored (7 radio-marked, n = 66 ducklings), 1 brood survived to fledge 5 ducklings. Other broods dwindled slowly, with total brood loss at the IA (1), IB (1), IC (1), and IIA (2) stages. The fate of one brood could not be determined because the hen died when the brood was at the IIA stage. Another brood made it to the IC stage, but we did not trap the hen in time to give her a transmitter, so their fate was uncertain. One hen with a transmitter lost her nest to flooding just before the expected hatch date, and thus did not have a brood.

DISCUSSION

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 (30% flooding and 70% depredation) were also similar to those of other studies (flooding 16-24%, depredation 67-80%, and desertion 5%, Mendall 1958, McAuley and Longcore 1989). Early estimates of hatching success appear to be slightly lower than those of Eberhardt's previous study in north central Minnesota (94%, Hohman and Eberhardt 1998), but the spring of 2008 was very cool and rainy, which may have chilled eggs and flooded nests.

We have identified ways to improve our nest searching methods and success rate next field season. In 2009, we will scout lakes with ring-necked duck habitat for pairs and lone males before conducting nest searches. This modification will focus our nest searching efforts on lakes with nesting pairs rather than on lakes with nesting habitat. A small percentage of the lakes with nesting habitat were actually used by ring-necked ducks; our success rate finding nests at lakes with good nesting habitat was 25%. However, when pairs or lone males were observed on lakes, our success finding nests was 90%. Although the number of nests we found was comparable to that in other studies of ring-necked ducks (45 nests in 3 years, Maxson and Riggs 1996; 35 nests in 2 years, Koons and Rotella 2003, 188 nests in 6 years by R. T. Eberhardt), we expect to have even greater success in 2009 with these modifications to our methods.

ACKNOWLEDGMENTS

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

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

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

SUMMARY OF FINDINGS

The Minnesota Department of Natural Resources (MNDNR) is conducting a study that examines use and survival benefits of waterfowl refuges to locally produced ring-necked ducks (*Aythya collaris*). During 2007 and 2008, we captured and implanted 108 flightless ring-necked ducks with radiotransmitters. Ducklings were tracked weekly by aircraft and from telemetry receiving stations located on 14 waterfowl refuges. The distance between weekly locations averaged ~8 km in both 2007 and 2008. Birds departing from their natal lakes did not exhibit strong directionality in movements. Young ring-necked ducks used state and federal waterfowl refuges, but this use was not evenly distributed among refuges; 2 refuges received the majority of use and 4 refuges have yet to be used by marked birds. Refuge use also increased markedly during hunting season. Additional data collection in 2009 will be aimed at increasing sample sizes to address survival benefits of refuge use to young birds.

INTRODUCTION

The MNDNR Fall Use Plan recognized sizable populations of resident breeding ducks as a cornerstone to improving fall duck use. Although breeding ring-necked duck populations have been increasing continentally, they may be declining in Minnesota (Zicus et al. 2005). Furthermore, hunter harvest of ring-necked ducks has declined markedly in Minnesota in the last 20 years, even as numbers of these birds staging on most traditional ring-necked duck refuges in the fall have increased in the state (Wetland Wildlife Populations and Research Group, unpublished data). Efforts to better understand population status began in 2003 with development of a ring-necked duck breeding-pair survey.

The Fall Use Plan identified the need to better understand the role of refuges in duck management. The influence of north-central Minnesota refuges on the distribution and welfare of resident ring-necked ducks is unknown. Factors influencing resident populations of ring-necked ducks are also poorly understood, as is the influence that the distribution of resident ring-necked ducks might have on that of migrant ring-necked ducks staging in the fall.

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

OBJECTIVES

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

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

STUDY AREA

The study area lies in the heart of the Laurentian mixed forest province of Minnesota. This area is characterized by mixed coniferous and hardwood forest pocked with lakes, many of which are dominated by wild rice (*Zizania palustris*). The study area is ~200 x 135 km in size and encompasses a significant portion of the core of ring-necked duck breeding range in Minnesota, as well as 14 important ring-necked duck refuges which are not open to public hunting (Figure 1, Table 1).

METHODS

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

By early September, radiotelemetry stations were established at each refuge as a means of quantifying refuge use. These stations consisted of a tower with a four-element yagi antenna pointed toward the primary waterfowl use areas within the refuges. In some cases, more than 1 antenna was used so that a greater area could be covered. The receivers were programmed to scan all transmitter frequencies each hour and were equipped with data loggers to store the data (Advanced Telemetry Systems, Incorporated DCC II Model # D5041 and Model # R4500). Data were downloaded weekly from data-loggers from mid-September through early November, and examined to determine presence/absence of radiomarked birds. Reference radiotransmitters were stationed permanently at each refuge to ensure that receivers and data loggers functioned properly. Flights were also conducted once weekly throughout the fall to document the locations and survival of radiomarked birds within the study area. Additional location and survival information came from USGS Bird Banding Lab banding and harvest reports. These reports include the hunters' names and the dates and locations of harvest.

RESULTS

We captured 52 ducklings with night-lighting techniques between 4 August and 3 September 2007. In 2008, we captured 56 ducklings between 29 July and 26 August. Capture locations were distributed throughout the study area (Table 2 and Figure 2).

Birds moved in all directions from their natal lake. All but 1 bird left its natal lake before hunting opened over the 2 years. Success locating birds from aerial flights was higher before hunting season than the week hunting opened in both years (87% before and 66% after in 2007, 95% before and 83% after in 2008). Success locating birds also declined as birds began moving more in preparation for migration. For the tracking period, average weekly movements were 8.5 $\pm$ 1.9 km in 2007 and 8.3 $\pm$ 2.1 km in 2008. Average weekly movements tended to increase as the season progressed until mid to late October when birds started leaving the study area. Average weekly movements prior to the start of hunting (6.4 $\pm$ 1.1 km and 6.8 $\pm$ 1.6 km, in 2007 and 2008 respectively) were shorter than after hunting season opened (14.5 $\pm$ 3.0 km and 16.6 $\pm$ 3.5 km) in both years.

At the conclusion of the 2007 tracking season, 24 radiomarked birds were known to have died, of which 8 were harvested by hunters. Four of the 8 hunter harvested birds were shot during the first 2 days of the season (29 and 30 September). Two were harvested in Louisiana and 1 in Illinois. Natural sources of mortality based on evidence at the site where the transmitter was found included predation by mink (*Mustela vison*) and other mammals (7) and birds including great-horned owls (*Bubo virginianus*) or other raptors (3). Six radios were thought to have dehisced because they were retrieved from open water. Losses to predation prior to hunting season (7) were similar or slightly higher than those during hunting (3). Formal survival analyses have yet to be performed.

At the conclusion of the 2008 tracking season, 33 radiomarked birds were known to have died, of which 11 were harvested by hunters. Four (36%) of these birds were shot during opening weekend. Birds were shot predominantly in Minnesota (8), but losses also occurred in Louisiana (2) and South Carolina (1). We attributed mortality to predation by mammals (5), raptors (1), and unknown sources (5). Four radios were thought to have dehisced, because they were retrieved from open water, and a definitive cause of mortality could not be determined for 7 birds. Five of these cases might have been ducklings that were crippled and later scavenged. Before hunting season, predators took 10 marked birds. After hunting opened, 1 bird was lost to predation and 5 others were lost to unknown causes.

Refuges were rarely used before hunting season, but use increased markedly with the onset of hunting (Figure 3). Refuge use was documented for 18 radiomarked birds on 8 refuges during the fall of 2007 (Table 1), and 12 radiomarked birds used 8 refuges in 2008. The most heavily used refuges in 2007 were Mud-Goose (7 birds), Tamarac National Wildlife Refuge (6 birds), and Fiske and Blue Rock Lakes (5 birds). In 2008, Mud-Goose (6 birds) and Gimmer Lake (3 birds) were used the most by radiomarked birds. Rice Lake NWR was not used in either year, but we expected use of this refuge by radiomarked birds to be less than that of refuges located within the capture area. Nevertheless, Rice Lake NWR is an important staging area for ring-necked ducks in the fall, so we will continue to monitor this refuge next year.

DISCUSSION

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

ACKNOWLEDGMENTS

M. Zicus was instrumental in planning, development, and initiation of this project. J. Berdeen helped with planning, surgeries, night-lighting, and hiring interns. N. Besasie, S. Bischof, P. Christensen, B. Ferry, R. Hanauer, J. Kennedy, T. Peterson, and E. Zlonis, helped set up remote towers, capture, and radiotrack birds. S. Cordts, J. Lawrence, K. Noyce, and B. Sampson helped with surgeries. J. Lawrence and S. Cordts also helped in the field, especially with aerial telemetry. J. Heineman flew telemetry flights, and A. Buchert flew us to Drumbeater Lake and helped set up the remote tower. J. Fieberg provided statistical advice. Dr. A. Piller and M. Kelly of Bemidji Veterinary Hospital ordered surgical equipment and gave technical advice. Dave Brandt, USGS Northern Prairie Research Center, loaned us 2 pairs of crimping pliers. John Finn, Leech Lake DNR, guided us into Drumbeater Lake, and retrieved equipment from Drumbeater Lake. R. Lego allowed access to the Leech River, and D. Barrett, North Country Regional Hospital Surgery Department, sterilized transmitters.

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Table 1. National Wildlife Refuges and Minnesota State Refuges included in the study area, approximate location of the refuges, peak numbers of ring-necked ducks during fall migration, number of recording telemetry stations established on each refuge, and the use of each refuge by radiomarked post-fledging ring-necked ducks during 2007-2008.

Refuge	Location	~Peak numbers	Stations	Refuge use	
				2007	**2008**
National Wildlife Refuge					
Rice Lake	5 mi SSW of McGregor	120,000	4	No	No
Tamarac	16 mi NE Detroit Lakes	10,000	3	Yes	Yes
State Waterfowl Refuge/State Game Refuge					
Donkey Lake	6 mi SW Longville	350	1	Yes	No
Drumbeater Lake	2 mi N of Federal Dam	160,000	1	Yes	Yes
Fiske and Blue Rock Lakes	8 mi SE Northhome	40,000	1	Yes	No
Gimmer Lake	10 mi SE Blackduck	200	1	No	Yes
Hatties and Jim Lakes	13 mi SE Blackduck	0	1	No	No
Hole-in-Bog Lake	2 mi SW Bena	4,000	1	No	No
Mud-Goose	4 mi SSW of Ballclub	2,100	1	Yes	Yes
Lower Pigeon Lake	4 mi S Squaw Lake	700	1	Yes	Yes
Pigeon River Flowage	6 mi S Squaw Lake	700	1	No	Yes
Preston Lakes	22 mi ENE of Bemidji	535	1	No	Yes
Round Lake	8 mi N Deer River	7,000	1	No	No
Rice Pond	9 mi E of Turtle River	15	1	Yes	Yes

Table 2. Ring-necked duckling captures per county in Minnesota during 2007 and 2008.

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

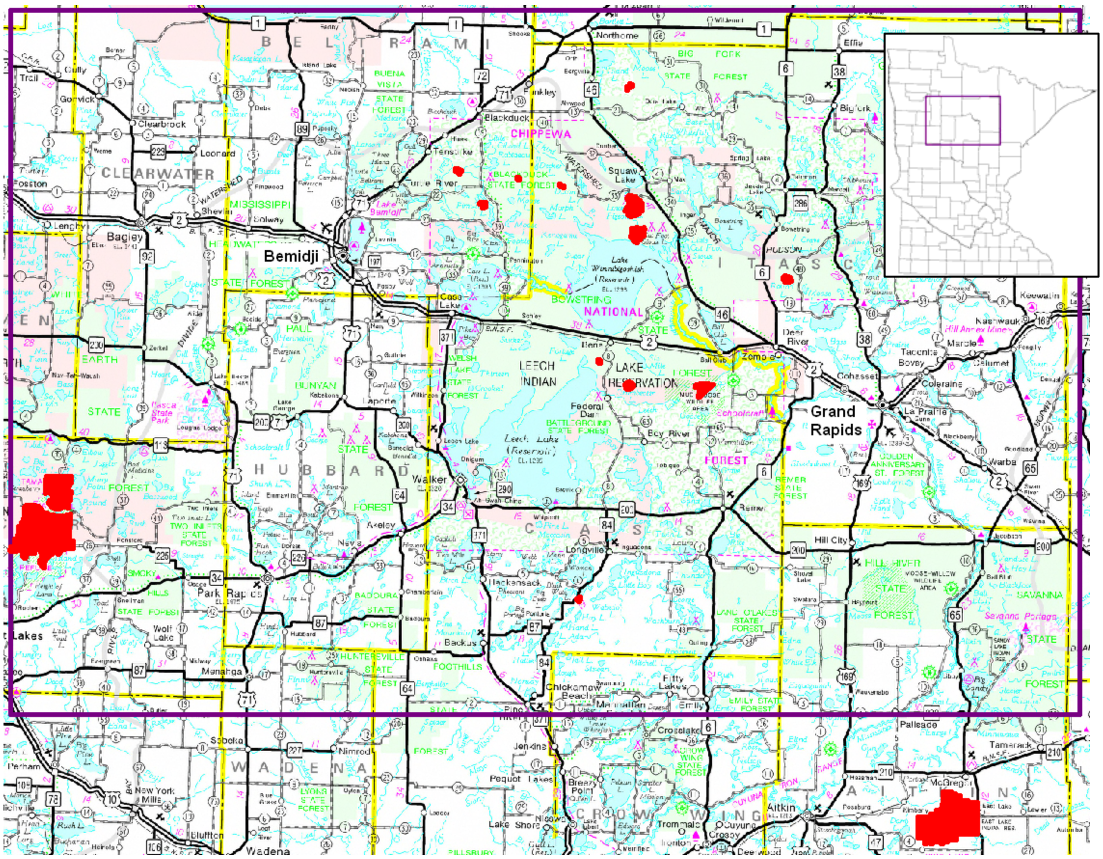

Figure 1. Ring-necked duck study area in Minnesota during 2007-2008 with 12 state waterfowl refuges/state game refuges and 2 National Wildlife Refuges depicted in red.

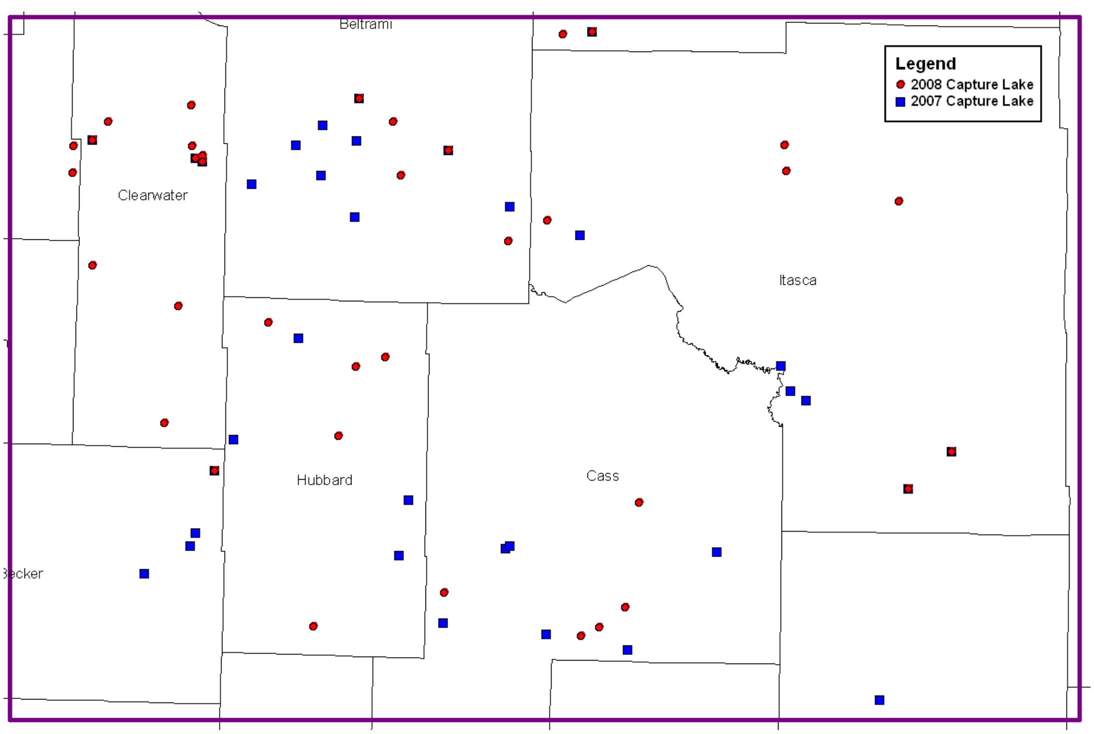

Figure 2. Capture locations for ring-necked duck ducklings in Minnesota during 2007 and 2008.

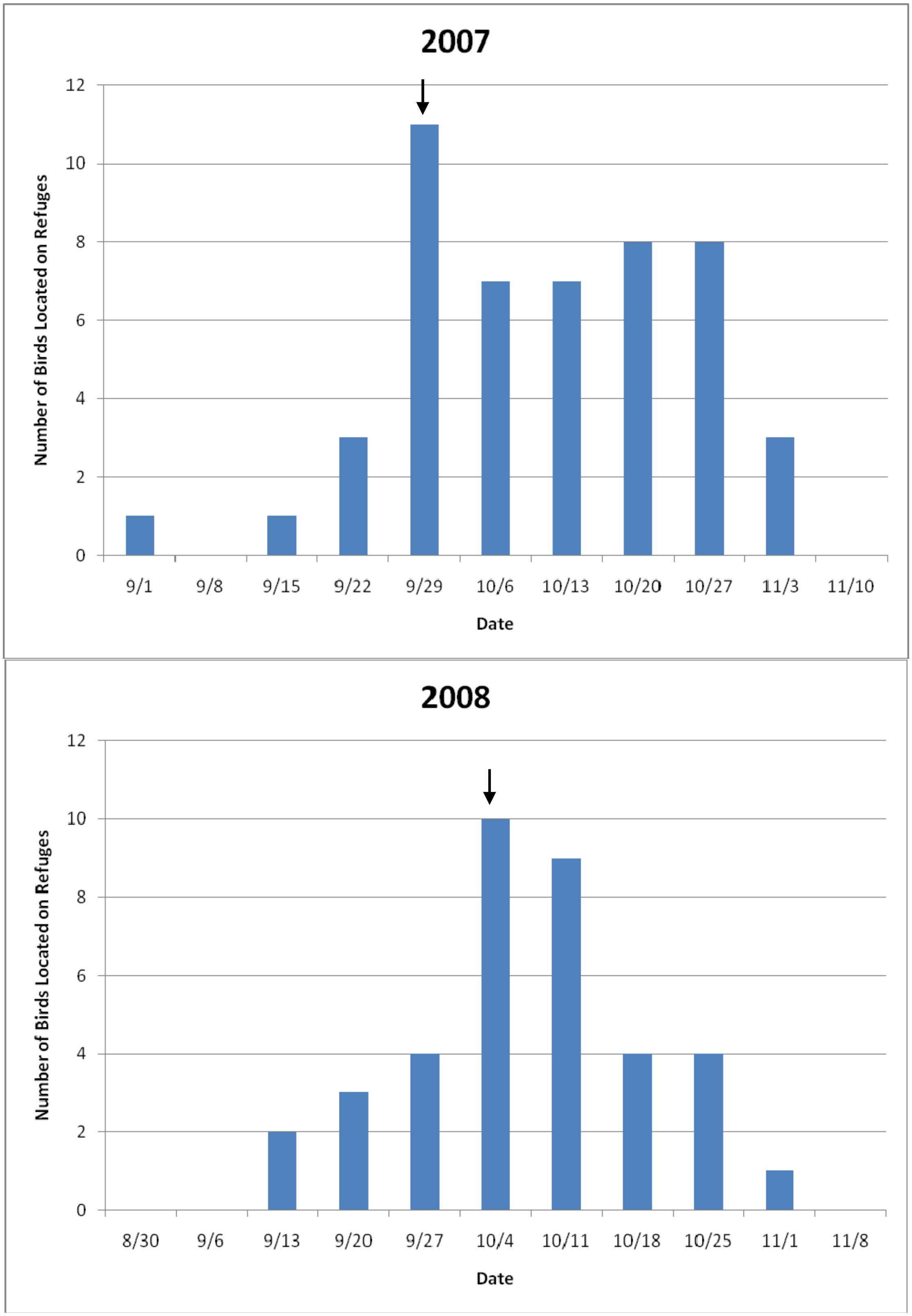

Figure 3. Weekly use of refuges by post-fledging ring-necked ducks before and during hunting season in 2007 and 2008 in Minnesota. Weeks are from Saturday through Friday with the Saturday date shown. Arrows indicate the week hunting opened.

ASSESSING CHARACTERISTICS OF KENOGAMA LAKE, A SHALLOW WATERFOWL LAKE IN NORTHERN MINNESOTA: TWO YEAR SUMMARY

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SUMMARY OF FINDINGS

Kenogama Lake (Kenogama) is a shallow lake in western Itasca County, MN, contained within the boundaries of the Laurentian Mixed Forest. The lake is believed to be of considerable importance to migrating diving ducks, especially Lesser Scaup (*Aythya affinis*). During the past 15 years, anecdotal evidence indicates that fall use of Kenogama by diving ducks has diminished. Mechanisms responsible for these declines are unknown but may include changes in duck migration patterns, weather and precipitation dynamics, or changing availability of aquatic invertebrates or other food resources important in diets of migrating Lesser Scaup and other ducks. Of particular interest is whether historical use of Kenogama as a site for rearing of walleye (*Sander vitreus*) fry is related to changes in lake characteristics and habitat suitability for migrating ducks. During 2007 and 2008, we monitored relative abundance of fish and aquatic invertebrates, water transparency, phytoplankton abundance, major nutrients, submerged macrophytes, and other characteristics of Kenogama. Fish were abundant, with golden shiners (*Notemigonus crysoleucas*) and walleyes comprising most biomass in our samples in 2007; however, adult walleyes appeared to be absent from the lake during 2008. We observed sparse populations of macroinvertebrates such as aquatic insects and amphipods. Zooplankton were abundant, but only small taxa were numerous, probably reflecting high predation by zooplanktivorous fish. Water quality data and relative abundance of submerged aquatic plants were indicative of a shallow lake in a “clear-water state” with a lighted substrate and rooted aquatic plants present in most areas throughout the lake. Adult walleye stomach contents indicated considerable consumption of aquatic invertebrates. It is not known whether this consumption is responsible for a low density of macroinvertebrates throughout the lake, but planktivorous fish may have been equally important. During 2008, plants and nutrients showed some evidence of a trend toward turbid conditions. We plan additional monitoring efforts at Kenogama and other forest lakes during 2009-2011. Current and future data should help clarify influences of various fishes and other factors on shallow lake characteristics and suitability for waterfowl.

INTRODUCTION

Kenogama Lake holds considerable interest to wildlife managers in north central Minnesota due to its history of fall use by migrating diving ducks. Located in the Laurentian Mixed Forest, Kenogama also represents a type of shallow lake that has received little study in North America. In Minnesota and elsewhere, shallow lakes are believed to exhibit a bimodal distribution of characteristics, tending toward opposite regime conditions along a continuum of water clarity and extent of submerged macrophyte development (Scheffer 2004). These “alternative states” are typically characterized by clear-water lakes containing abundant submerged macrophytes, and alternatively, by lakes with turbid water and sparse submerged macrophytes. In each alternative regime, shallow lakes are believed to exhibit stability and resist changes toward the opposite extreme, especially at either very high or very low levels of background nutrients. However, at intermediate nutrients, either regime is possible and lakes can quickly shift in response to water level changes, winter hypoxia and resulting fish “winterkill”, chemical fish kills, introduction of fish, and other perturbations. For example, turbid shifts sometimes follow increased density of planktivorous/benthivorous fish populations, prolonged increases in water depth, or increased nutrient loading (although examples of the latter are rare). Complete removal of fish from shallow Minnesota lakes has been shown to

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induce transitions toward clear-water states (Hanson and Butler 1994a, Zimmer et al. 2001), but in such cases, regime shifts may be temporary.

Mechanisms influencing characteristics of shallow lakes in forested regions of Minnesota and elsewhere are poorly documented. At least some north temperate shallow lakes seem to follow a pattern of alternative regimes (Bayley and Prather 2003, Zimmer et al. in press). Minnesota's shallow lakes program has compiled data from 375 shallow lakes statewide, yet these efforts target relatively few lakes in the Laurentian forest. Data from Minnesota also indicate that patterns of shallow lake characteristics differ dramatically between prairie and transition ecoregions, perhaps indicating importance of different structuring mechanisms across regional gradients (Herwig et al. 2006).

Previous studies demonstrated that shallow lakes in the Minnesota parkland often support diverse fish communities (Herwig et al. 2006). Thus, although Kenogama is in the forest, we expected that it might also contain a rich fish community. This seemed especially likely given the lake's size, history of angler interest, and the recent pattern of mild winters. Limited reports from Kenogama indicated that water clarity was good, that abundance of submerged aquatic plants was relatively high, and that plants were not limited by poor water clarity (Hansel-Welch et al., unpublished data). Kenogama has been used to rear walleye since 1983 (MNDNR, unpublished data). Walleye fry are stocked in spring; juveniles (age-0) are removed during fall. Some unharvested walleye are known to survive over-winter because, summer and winter angling was sometimes popular, at least during the past decade.

Recent research evaluating stocking of walleye fry in shallow prairie lakes indicated that adding juvenile (age-0) walleye to sites containing dense fathead minnow (*Pimephales promelas*) populations sometimes bolster abundance of macroinvertebrates and zooplankton, and favors clear water shifts (Potthoff et al. 2008). However, it is currently not possible to predict long-term consequences of walleye fry stocking in shallow lakes with unknown fish communities, or where adult fish are well established.

During May 2007-September 2008, we monitored ecological characteristics and habitat suitability for waterfowl at Kenogama. Our objectives were to: (1) document current ecological conditions within the lake; (2) assess characteristics of the lake's current fish community; (3) describe the invertebrate community, with special emphasis on selected taxa known to be important for water quality and as waterfowl food; and (4) draw broad comparisons between Kenogama and other shallow MN lakes recently studied. Here, we summarize results of our efforts during May-September 2007 and 2008; we also discuss implications of our findings and offer some hypotheses about current characteristics and possible changes in the lake.

METHODS

During May 2007, we chose 6 transects by randomly selecting 6 compass bearings from the approximate center of the lake. All sampling for aquatic invertebrates and fish was conducted along these transects during the study.

Fish Community

Relative abundance and species composition of Kenogama's fish population was assessed using 3 gill nets and 12 mini-fyke (small trap) nets during June and August 2007 and 2008. For each sampling, a single mini-fyke net was deployed along the shore, or at the deep margin of emergent vegetation along each of the 6 transects. Gill nets were set concurrently, 1 each at the deepest location along transects 2, 4, and 6. Sampling gear was deployed in the morning and checked approximately 24 hours later. Fish were identified to species, and wet weights (g) and total lengths (mm) were determined in the field. During 2007, random samples of stomachs and otoliths were taken from walleye. Because we were especially concerned with population characteristics and functional influences of walleye, we also examined walleye length at age distribution, length frequency, relative weights (Wr, Pope and Carter 2007), and stomach contents. During late May 2008, DNR Fisheries staff stocked approximately 7 million walleye fry into Kenogama. Adult walleye were absent in 2008, apparently due to winter hypoxia (winterkill), but stomach analysis was conducted on young of the year (age-0) walleyes, yellow perch (*Perca flavescens*), and golden shiners.

Aquatic Invertebrates

Two sampling stations were established along each of the 6 transects, one 20 meters from the edge of the emergent vegetation and a second at the midpoint between the shoreline and the lake center. Aquatic invertebrates were sampled at the 12 stations using column samples (CS, Swanson 1978) and vertical activity traps (AT, after the design of Muscha et al. 2001) at approximately 2-3 week intervals (6-8 times each summer). AT samplers were used at deep and shallow locations; CS were taken at deep sites only. AT were deployed for approximately 24 hrs, then samples were condensed by passage through an 80 μm-mesh funnel. CS were concentrated by passage through a 64 μm-mesh funnel. Both CS and AT samples were preserved in 70% ethanol.

Aquatic macroinvertebrates were sampled during August and September 2008 using 500 μm mesh D-frame sweep net. Sweep nets were gathered concurrently from deep and shallow locations from 3 of the 6 transects. Substrate samples were taken by skimming the net along the bottom sediment for a distance equal to the depth at the sampling station described by Anteau and Afton (2008). Contents of the sweep nets were then concentrated with a 243-μm mesh wash net to reduce the amount of decaying organic material. Contents were put in plastic bags, refrigerated and later identified.

Invertebrates from CS, AT, and sweeps were identified to the lowest feasible taxonomic group (mostly family, sometimes genus) and were counted in a lab at Bemidji State University. We pooled organisms into 11 groups: all insects, all Diptera (Chaoboridae, Chironomidae, Culicidae), Corixidae, Ephemeroptera, Amphipoda, large cladocera (mainly *Daphnia, Ceriodaphnia, Simocephalus,* and Sididae), small cladocera (Chydoridae, Bosminidae, *Diaphanosoma* and *Eurycercus*), cyclopoid copepods, calanoid copepods, and *Leptodora*. We combined results of all CS and AT samples on each sampling date to develop a relative abundance estimate for each of the 11 groups. Organisms captured using sweep nets (2008 only), were pooled to form the following groups: Chironomidae, Hydracarina, *Leptodora,* Oligochaeta, *Hyalella*, Physidae, other Insects (including Diptera, Ephemeroptera, Trichopetera, Anisoptera) and Hirundea. We assessed trends in major taxa graphically.

Plant Community

Relative abundance of submerged macrophytes was measured during August 2007 and 2008 using methods of Deppe and Lathrop (1992) (an approach generally similar to that used by MNDNR Section of Wildlife Shallow Lakes Program staff and by researchers from MNDNR Wetland Wildlife Group). For plant surveys, we selected 8 transects, with 5 sampling locations equidistant from one another and from shorelines. At each location, we collected plants using 2 casts of a weighted plant rake. We recorded presence/absence of individual submergent species retained on each cast. A maximum score of 40 would indicate that a particular species was collected on each cast at all sampling stations.

We compared Kenogama to other shallow lakes in Minnesota using 2 procedures. First, we constructed a plant relative-abundance matrix by combining Kenogama plant survey results with similar data from a recent (2006) study of 74 shallow lakes in MN (Herwig et al. 2006). We then used Non-metric Multidimensional Scaling (NMS) to assess similarity between plant communities of Kenogama and the other shallow lakes. Second, we plotted plant extent (% vegetated points) and water clarity (average Secchi/average lake depth) of Kenogama along with values from other shallow lakes in a data set supplied by the MNDNR Section of Wildlife Shallow Lakes Program (Nicole Hansel-Welch, unpublished data). These approaches should not be considered formal statistical tests, but they do allow visual contrasts of potential similarities or differences between Kenogama and other shallow MN lakes.

Chemical Properties and Water Quality Features

We also measured water clarity, phytoplankton abundance (indexed using chlorophyll *a* (Chl a)), and concentrations of major nutrients at approximately 2-3 week intervals during open water periods from 31 May-18 September 2007 and 15 May-17 September 2008. Chemical analyses were performed from surface-dip water samples collected at 3 central-lake locations. Secchi disk transparency was measured at these 3 sites using a standard (20-cm) circular disk. We measured turbidity directly using a LaMotte turbidity meter following transport of water samples back to the lab. Water samples collected for determination of total phosphorus (TP), total nitrogen (TN), and ammonium (NH_4) were frozen and transported to laboratory facilities. Water samples were also collected and later analyzed for total dissolved phosphorus (TDP) and phytoplankton abundance (Chl a). We also measured TDP because this is sometimes more useful than TP for evaluating ecological change in shallow lakes (Potthoff et al. 2008). TDP samples were prepared by filtering lake water through GF/F glass fiber filters (0.7 µm nominal pore size) and immediately freezing the filtered water. Chl a samples were prepared by filtering lake water through GF/F glass fiber filters which were wrapped in tin foil and immediately frozen. TDP concentrations were determined using high-temperature persulfate digestion followed by ascorbic-acid colorimetry. Chl a was measured via fluorometric analysis following a 24-h, alkaline-acetone extraction of photosynthetic pigments. All chemical procedures for analysis of Chl a, TP, TDP, TN, and NH_3 were performed using laboratory facilities at the University of St. Thomas (St. Paul, MN). Evaluation of data trends was done graphically.

Relative Water Depth

Relative water level readings were recorded approximately biweekly from May-September 2007 and 2008 by reading a depth gauge near the boat access. On 8 June 2007, using a Lowrance sonar unit, we also measured water depth at various locations around the lake.

RESULTS

Fish Community

Fourteen and 10 fish species were captured during 2007 and 2008, respectively (Tables 1, 2). In 2007, golden shiner, fathead minnow and walleye were the most abundant fishes (highest relative mass, Table 1). Golden shiners and brook stickleback (*Culaea inconstans*) were most abundant in 2008 (Table 2). Golden shiners from several size (year) classes were captured in mini-fyke nets during both years; gill nets also collected golden shiners, but only larger sizes (135-185 mm) representing older year classes. Gill net catches were generally lower during 2008, when very few golden shiners, yellow perch, and walleye were collected in gill net samples.

We observed 3 peaks in length frequency of walleye collected during 2007 and walleye length increased from June to August, reflecting summer growth (Figure 1). Age-assignment based on otoliths confirmed 3 (or more) year-classes indicated by length-frequencies (Figure 1). Randomly selected walleye ranging from 230–300 mm were age-1 (2006 year class), fish ranging from 360–430 mm were age-2 (2005 year class), and larger walleye ranged from ages 3-5. No adult walleye were collected during 2008, but young of the year walleye that were stocked in May were collected during August that year.

Walleye appeared to be in good condition during summer 2007, with an average relative weight (W_r) ranging between 0.8 and 1.0 (Figure 2). Smaller fish achieved highest W_r during June 2007, whereas larger fish appeared healthier during August. In general, W_r values were negatively associated with total length during June 2007, but an opposite (positive) correlation was evident by August.

Summer diet of adult walleye (2007) consisted primarily of aquatic invertebrates (Table 3). Amphipods comprised the major percentage of food found in walleye stomachs during June (32.7% wet mass, n = 8) and August (58.3% wet mass, n = 9). Minnows (cyprinids) were absent in walleye stomachs during June, but occurred in 16.7 % of stomachs examined in August (31.3% wet mass, n = 9). Other food items present in walleye stomachs were Decapoda (crayfish), Hirundea (leeches), and larval insects. Almost one-fourth of walleye stomachs were empty, and considerable proportions of stomach contents (19.4%) were decomposed, thus were unidentifiable. In August 2008, the walleye population consisted entirely of age-0 fish, and stomachs contained mostly of cyprinids (94.2% wet mass, n = 9) (Table 4). Forty percent of stomachs were found to be empty, and contents of 20% (n = 5) were unidentifiable. Stomach contents were also taken from age-0 yellow perch in August 2008. Yellow perch diets included zooplankton (such as cladocera, 25% wet mass, n = 4) and some vegetation (20.5% wet mass, n = 4), and a lesser quantity of minnows (cyprinids, 13.6% wet mass, n = 2). Approximately one-third of the stomachs examined contained unidentifiable material (31.8% wet mass, n = 5), and 28% were empty. In August 2008, golden shiner stomachs showed a high presence of zooplankton (38.5% wet mass, n = 5), along with some Diptera (15.4% wet mass, n = 1) and vegetation (15.4% wet mass, n = 2) present. Most of the golden shiner stomachs examined were empty (60%), and almost one-third of the total proportion of stomach contents was unidentifiable (30.8% wet mass, n = 2).

Aquatic Invertebrates

Zooplankton samples during late May–September 2007 were numerically dominated by small-bodied cladocerans and copepods, but large cladocerans were also abundant during early August 2007 and late May, June and September 2008 (Figure 3a,b). Small cladocerans were present during both years, but were especially abundant during early 2007 (Figure 3a). Density of small cladocerans was variable during 2008, but peaked in June and August (Figure 3b). Large cladocerans occurred in relatively low numbers during 2007 (except early August), but were much more abundant in 2008, with peaks in late May and mid-September (Figure 3a,b). *Leptodora* (large, predatory cladocera) were absent in samples from the first 3 dates in 2007, appeared in July, but returned to low densities by mid-September (Figure 3a). In 2008, *Leptodora* were present at low density, but peaked briefly during August (Figure 3b).

Calanoid copepods were abundant but variable throughout 2007–2008 (Figure 4a,b). Throughout most of 2007 and 2008, cyclopoid copepods were present and less abundant than calanoids except for a peak during May–June 2008 (Figure 4b). In general, calanoid copepods were more abundant than were cyclopoids in both study years (a pattern opposite what we usually observe in shallow lakes elsewhere; Figure 4a,b).

Amphipods (*Hyalella* only) were collected in low numbers throughout 2007, but increased briefly during early-summer 2008 (Figure 5a,b). Amphipods ranged from 0 (31 May and 1 August) to 4 individuals in late July 2007 and remained similarly low during August–September 2008. Diptera were the most abundant insects during all sampling periods (except late September 2007). Corixidae (water boatmen) were periodically collected in 2007, but densities remained very low (< 5 individuals, lake-wide) (Figure 6a). Corixidae densities were higher in 2008, but they were never abundant (Figure 6b). Ephemeroptera (mayflies) occurred periodically in 2007, ranging from 0 (7 June and 1 August 2007) to 18 individuals (20 June 2007), with a peak during June (Figure 6a). Ephemeroptera also occurred periodically in 2008, with numbers ranging from 0 (15 May and 19 August 2008) to 26 individuals (26 June 2008) (Figure 6b). Peak density was observed in early summer (late May to late June) and numbers decreased steadily (Figure 6b).

Some aquatic macroinvertebrates were captured in greater numbers in sweep nets than with column samplers and activity traps. For example, Chironomidae were predominant in the 2008 sweep net sampling, with a peak in August (394 individuals/net), then declining in

September (73 individuals/net). Mean *Hyalella* catches in sweeps ranged from 105 individuals in August to 6.83 in September (2008 only). Other taxa were present at very low densities in sweep net catches (e.g. Hydracarina, Oligochaetes, Figure 7) and these were not represented in AT or CS samples.

Water Clarity, Phytoplankton, and Major Nutrients

Water clarity, represented by the Secchi depth:mean depth ratios, followed typical summer patterns, reaching seasonal highs early, then decreasing to lower levels by fall (2007 and 2008; Figure 8a,b). Annual ratios of mean Secchi/depth remained well above values of 0.5 (a theoretical maximum for depth of the photic zone when light penetration equals depth, Figure 12b), indicating that most of the lake bed received sufficient sunlight to facilitate growth of submerged aquatic plants.

TP values in Kenogama remained relatively low and nearly constant throughout May–August 2007, ranging from slightly below, to slightly above, 25 *u*g L^{-1} (Figure 9a). We observed a very different pattern during 2008, when TP was increased by a factor of approximately 3 and fluctuated throughout the summer (Figure 9b). Dissolved phosphorus was present during both years, but with slightly higher values and greater variability during 2008. Throughout 2007, TDP comprised >50 % of the TP pool, but in 2008 TDP was only about 25-30% of the TP pool.

Phytoplankton abundance remained very low during May–August/September, with mean observed values ranging from 5.5–8.5 and 3.4–9.3 *u*g L^{-1} in 2007 and 2008, respectively (Figure 10a,b). Ratios of TP:Chl a (nutrients:phytoplankton abundance) were relatively high during both years, indicating that a considerable portion of the TP pool was probably not associated with phytoplankton (Figure 10a,b). During 2008, a considerable portion of the water-column phosphorus was unaccounted for by phytoplankton biomass (Figure 10b) or dissolved pools (Figure 9b).

Relative Water Depth

Relative water depth generally decreased as the sampling season progressed (by approximately 0.7 ft (0.21 m)) during May– mid-September 2007 (Figure 11a) and by approximately 0.8ft (0.24 m) during May–mid-September 2008 (Figure 11b). We assessed depths lake-wide on 8 June 2007, when depth ranged from 3.1 – 4.9 ft (0.94 – 1.49 m) in various locations.

Submerged Aquatic Plants

Submerged aquatic plants were collected from 100% of our sampling locations during 2007 and 2008, but subtle differences were evident between years. Seven species were present during 2007, but, only 3 were widespread. These included Robbins' pondweed (*Potamogeton Robbinsii*, collected at 100% of sites), bushy pondweed (*Najas flexilis*, collected at 65% of sites), and large-leaf pondweed (*Potamogeton amplifolius*, collected at 43% of sites). Flatstem pondweed (*Potamogeton zosterformis)*, whitestem pondweed (*Potamogeton praelongus*), 1 *Sagittaria* spp., and 1 unidentified pondweed (*Potamogeton spp.*) were also collected, but these were far less abundant.

In 2008, submerged aquatic plants were again present at 100% of the sampling sites, but only 5 species were found. Only Robbin's pondweed (collected at 98% of sites) and large-leaf pondweed (collected at 63% of sites) were widespread. Bushy pondweed (collected at 20% of sites), *Sagittaria* spp. (collected at 13% of sites), and variable pondweed (*Potamogeton gramineus*, collected at 8% of sites) were present, but far less abundant.

Kenogama Lake's plant community differed from that observed in other shallow lakes recently sampled in prairie and parkland areas of Minnesota (Figure 12). Most notable was the widespread occurrence of Robbin's pondweed, which occurs throughout Kenogama, but was not collected from other shallow lakes sampled by our research group during 2007-2008.

DISCUSSION

Shallow lakes in North America show dramatic contrasts in lake features, typically conforming to either clear- or turbid-water regimes (Hanson and Butler 1994a,b, Scheffer 2004, Zimmer et al. in press). During 2007-08, Kenogama showed evidence of transition between regimes, with characteristics of both clear- and turbid-water states.

Trends in water transparency were similar between the 2 years of study. Water transparency was initially high and then decreased during the warmer months, with relatively clear water returning by late August–September. Water levels decreased as the summer progressed in both 2007 and 2008. Water levels were consistently higher in 2008, probably due to heavy spring precipitation.

Kenogama supported a diverse fish community during both 2007 and 2008. Species richness was 14 and 10 during 2007 and 2008 respectively, with lower values in 2008 probably reflecting influence of winterkill. The fish community was comprised of mostly planktivorous species during both years, with walleye being a notable exception. Benthivorous fish were not abundant and were not sampled in 2008. In 2007 1 white sucker (*Catostomus commersoni*) and 1 yellow bullhead (*Ameiurus natailis*) were captured. We collected no adult walleyes (>age 0) during 2008, apparently due to hypoxia and winterkill. Winterkill and lack of adult walleyes may help explain other changes such as increased amphipods and large cladocera, along with increased relative abundance of yellow perch in 2008.

Kenogama's fish community continued to differ markedly from those observed in other Minnesota shallow lakes, mostly due to its high golden shiner population. This dense population of golden shiners may result, in part, from accidental loss and dumping of bait by anglers. Fish sampling during 2007 confirmed that the lake supported an extensive population of adult walleyes resulting from walleye rearing activities and the incomplete removal of these fish during fall netting. During 2008, walleye biomass was again high by August 2008, but only age-0 fish were sampled, almost certainly due to severe hypoxia in winter 2007-08.

The combination of dense golden shiners (planktivores) concurrent with an established walleye population (piscivores) seems to contradict recent evidence indicating that walleye stocking limits planktivore abundance (Potthoff et al. 2008). A partial explanation for coexistence of these species may be that the extensive stands of submerged macrophytes in Kenogama provide refuge areas for golden shiners and other planktivores, thus uncoupling predator and prey densities. Because golden shiners achieve larger body size (than fathead minnows, for e.g.), it is also plausible that walleye predation did little to limit abundance or recruitment of the shiners.

Walleye diets consisted of macroinvertebrates and fish, but differed between 2007 and 2008, perhaps due to our methods (we examined only adult walleyes in 2007, then only juveniles in 2008). During 2008, cyprinids comprised over 90% of age-0 walleye diet. In contrast during 2007, amphipods were important food items of adult walleyes (40.7% wet mass), along with Decapoda (crayfish), Hirudinea (leeches), and Odonata (dragonflies); minnows (9.7%) were less important than a combination of other macroinvertebrates. It seems counterintuitive that young walleye in 2008 were targeting cyprinids and the older, larger fish were consuming macroinvertebrates. During 2008, yellow perch stomachs from (age-0 fish) contained mostly Branchiopoda (zooplankton; 25% wet mass) along with a large amount of vegetation (20.5%). Vegetation was probably due to incidental digestion from feeding in and around aquatic plants.

We did not expect to find that minnows were a major food source for fingerling walleyes in Kenogama during 2008, especially after observing little predation on minnows by adult walleyes in 2007. Ward et al. (2008) observed a shift from zooplankton to fish to macroinvertebrates during seasonal walleye development. It is plausible that walleye predation constrains zooplankton and macroinvertebrates in Kenogama, similar to the influence of walleye reported for large prairie wetlands in west-central Minnesota (Reed and Parsons 1999). However, given abundance of golden shiners and yellow perch, their consumption of zooplankton and macroinvertebrates probably exceeded that of the walleye population during the 2 years of our study. It is also likely that the extreme annual variability (and perhaps prey consumption patterns) reflect our small sample sizes and examination of walleye fry and adults during separate years.

During 2007–2008, Kenogama exhibited characteristics of a clear-water regime with widespread (but not lush) submergent or emergent macrophytes. This is consistent with our observation that sunlight penetration exceeded average lake depth during both years. Kenogama supported a relatively sparse invertebrate community in 2007, with a high proportion of small-bodied cladocerans (inefficient filter-feeders on phytoplankton) and relatively sparse macroinvertebrates. This contrasts somewhat with 2008, when we observed increases in small amphipods (*Hyalella*), large-bodied cladocera, and some aquatic insects. These patterns may signal a transition to clear-water regime conditions, or they may simply reflect absence of adult walleye during 2008, which were evidently foraging on both amphipods and aquatic insects during 2007 (51% total stomach mass). During 2008, sweep nets were also used to assess organisms closely associated with the bottom substrate. These results suggest that some invertebrates including those important as waterfowl food items (*Hyalella*, Chironomidae) were substrate-associated, thus may be poorly represented in data from column samples and activity traps. During both 2007 and 2008, density of most organisms drastically decreased between August and September, which may reflect reduced activity of invertebrates as waters cool, high consumption by fish (following a season's growth), or even migrating waterfowl foraging on the lake.

Presently, some patterns at Kenogama reflect characteristics of both clear- and turbid-water regimes. For example, submerged macrophytes are widespread throughout the lake and phytoplankton abundance remains low. At the same time, planktivorous fishes are abundant, richness of the submerged plant community is quite low, and TP:Chl a ratios are high. These findings may indicate that the lake is balancing near a threshold between clear and turbid regimes, perhaps due to sustained high water levels and a persistent, high-density population of planktivorous fishes.

Presently in Minnesota, there is a need for better understanding of basic ecological characteristics of shallow lakes in forested regions and elsewhere. Managers need to know typical ranges of conditions for shallow lakes in forested landscapes, and whether these lakes always provide good habitat for invertebrates and wetland wildlife simply because they exhibit clear water and support abundant submerged plants. Staff from the MNDNR Wetland Wildlife Populations & Research Group, MNDNR Fisheries, and the University of St. Thomas (St. Paul, MN) plan studies at Kenogama, along with approximately 15 other Laurentian Forest lakes and 135 additional shallow lakes statewide during 2009–2011. We expect that this work will increase basic knowledge of ecological characteristics of shallow lakes throughout Minnesota. More specifically, we hope that resulting data will improve understanding of regional patterns of variability in shallow lakes and identify mechanisms responsible for triggering shifts to turbid regimes (with poor water quality and little wildlife use). We are hopeful that additional data from Kenogama and similar forest lakes will provide practical guidance to resource managers in the Laurentian Forest. This is especially important because the mechanisms controlling shallow lake characteristics and water quality remain poorly understood, limiting our ability to properly manage these areas for wetland wildlife.

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Table 1. Relative abundance (mean weight (g) of catch per overnight set, standard error in parentheses) of fishes in Lake Kenogama, Minnesota, during summer 2007.

		14-Jun-07			9-Aug-07		
		Trap net	Gill net	Activity trap	Trap net	Gill net	Activity trap
Species	Common name	*N* = 12	*N* = 3	*N* = 12	*N* = 12	*N* = 3	*N* = 12
Cyprinidae							
Hybognathus hankinsoni	Brassy minnow	0.4(0.4)	0.0(0.0)	0.0(0.0)	0.0(0.0)	0.0(0.0)	0.0(0.0)
Notemigonus crysoleucas	Golden shiner	3759.8(791.6)	277.3(17.3)	0.0(0.0)	26.5(9.7)	150.2(93.0)	0.0(0.0)
Notropis heterolepis	Blacknose shiner	23.5(5.1)	0.0(0.0)	0.0(0.0)	4.4(2.5)	0.0(0.0)	0.0(0.0)
Phoxinus eos	Northern redbelly dace	68.5(27.8)	0.0(0.0)	0.0(0.0)	0.0(0.0)	0.0(0.0)	0.0(0.0)
Phoxinus neogaeus	Finescale dace	11.3(5.4)	0.0(0.0)	0.0(0.0)	0.4(0.4)	0.0(0.0)	0.0(0.0)
Pimephales promelas	Fathead minnow	292.6(129.6)	0.0(0.0)	0.3(0.3)	6.9(2.7)	0.0(0.0)	0.0(0.0)
Catostomidae							
Catostomus commersoni	White sucker	0.0(0.0)	720.0(361.2)	0.0(0.0)	0.0(0.0)	412.3(296.7)	0.0(0.0)
Ictaluridae							
Ameiurus natalis	Yellow bullhead	48.3(48.3)	0.0(0.0)	0.0(0.0)	0.0(0.0)	0.0(0.0)	0.0(0.0)
Umbridae							
Umbra limi	Central mudminnow	3.0(1.6)	0.0(0.0)	1.0(0.8)	17.9(13.5)	0.0(0.0)	0.5(0.5)
Gasterosteidae							
Culaea inconstans	Brook stickleback	38.3(16.6)	0.0(0.0)	3.4(2.2)	0.2(0.2)	0.0(0.0)	0.2(0.1)
Percidae							
Etheostoma exile	Iowa darter	1.7(1.1)	0.0(0.0)	0.0(0.0)	0.0(0.0)	0.0(0.0)	0.0(0.0)
Etheostoma nigrum	Johnny darter	0.3(0.2)	0.0(0.0)	0.0(0.0)	0.0(0.2)	0.0(0.0)	0.0(0.0)
Perca flavescens	Yellow perch	0.0(0.0)	0.0(0.0)	0.0(0.0)	3.3(3.3)	303.7(303.7)	0.0(0.0)
Sander vitreus	Walleye	260.5(180.5)	9445.7(1304.6)	0.0(0.0)	214.2(94.2)	11447.7(2868.0)	0.0(0.0)

Table 2. Relative abundance (mean weight (g) of catch per overnight set, standard error in parentheses) of fishes in Lake Kenogama, Minnesota, during summer 2008.

		18-Jun-08		20-Aug-08	
		Trap net	Gill net	Trap net	Gill net
Species	Common name	*N* = 12	*N* = 3	*N* = 12	*N* = 3
Cyprinidae					
Notemigonus crysoleucas	Golden shiner	3270.8(761.3)	29.7(29.67)	312.1(66.3)	53.3(53.3)
Notropis heterolepis	Blacknose shiner	48.8(21.9)	0.0(0.0)	12.0(7.6)	0.0(0.0)
Notropos hudsonius	Spottail Shiner	6.0(2.3)	0.0(0.0)	0.0(0.0)	0.0(0.0)
Phoxinus eos	Northern redbelly dace	353.8(70.4)	0.0(0.0)	0.0(0.0)	0.0(0.0)
Phoxinus neogaeus	Finescale dace	243.3(93.0)	0.0(0.0)	185.4(20.7)	0.0(0.0)
Pimephales promelas	Fathead minnow	510.0(117.2)	0.0(0.0)	14.6(10.9)	0.0(0.0)
Umbridae					
Umbra limi	Central mudminnow	103.3(41.5)	0.0(0.0)	355.8(57.3)	0.0(0.0)
Gasterosteidae					
Culaea inconstans	Brook stickleback	2352.1(667.0)	0.0(0.0)	2781.3(535.0)	0.0(0.0)
Percidae					
Perca flavescens	Yellow perch	10.8(6.0)	450.3(391.87)	1290.0(442.2)	123.7(123.7)
Sander vitreus	Walleye	0.0(0.0)	0.0(0.0)	1095.0(191.7)	7.7(7.7)

Table 3. Percent by weight and prevalence (percent of stomachs containing a food item) of stomach contents of walleyes in Lake Kenogama, Minnesota, summer 2007.

	6/14/2007 *N* = 8 stomachs examined Empty stomachs = 1 (12.5%)		8/9/2007 *N* = 9 stomachs examined Empty stomachs = 3 (33.3%)		Overall *N* = 17 stomachs examined Empty stomachs = 4 (23.5%)	
Food	Percent by weight	Prevalence	Percent by weight	Prevalence	Percent by weight	Prevalence
Hirudinea	28.2	25			9.4	11.8
Crustacea						
Amphipoda	32.7	25	58.3	66.7	40.7	35.3
Decapoda	15.5	12.5			10.7	5.9
Insecta						
Odonata	9.3	12.5	2.1	16.7	7.1	11.8
Ephemeroptera			8.3	33.3	2.6	11.8
Diptera	0.1	12.5			0.3	5.9
Pisces						
Cyprinidae			31.3	16.7	9.7	5.9
Unidentified	28.2	37.5			19.4	17.6

Table 4. Percent by weight and prevalence (percent of stomachs containing a food item) of stomach contents of fish in Lake Kenogama, Minnesota, summer 2008.

	Walleye *N* = 25 stomachs examined Empty stomachs = 10 (40%)		Yellow Perch *N* = 25 stomachs examined Empty stomachs = 7 (28%)		Golden Shiner *N* = 25 stomachs examined Empty stomachs = 15 (60%)	
Food	Percent by weight	Prevalence	Percent by weight	Prevalence	Percent by weight	Prevalence
Hirudinea						
Crustacea						
Amphipoda			4.5	4		
Branchiopoda			25	16	38.5	25
Decapoda						
Insecta						
Odonata						
Ephemeroptera						
Diptera	0.9	4	4.5	4	15.4	4
Pisces						
Cyprinidae	94.2	36	13.6	8		
Unidentified	4.8	25	31.8	25	30.8	8
Vegetation			20.5	16	15.4	8

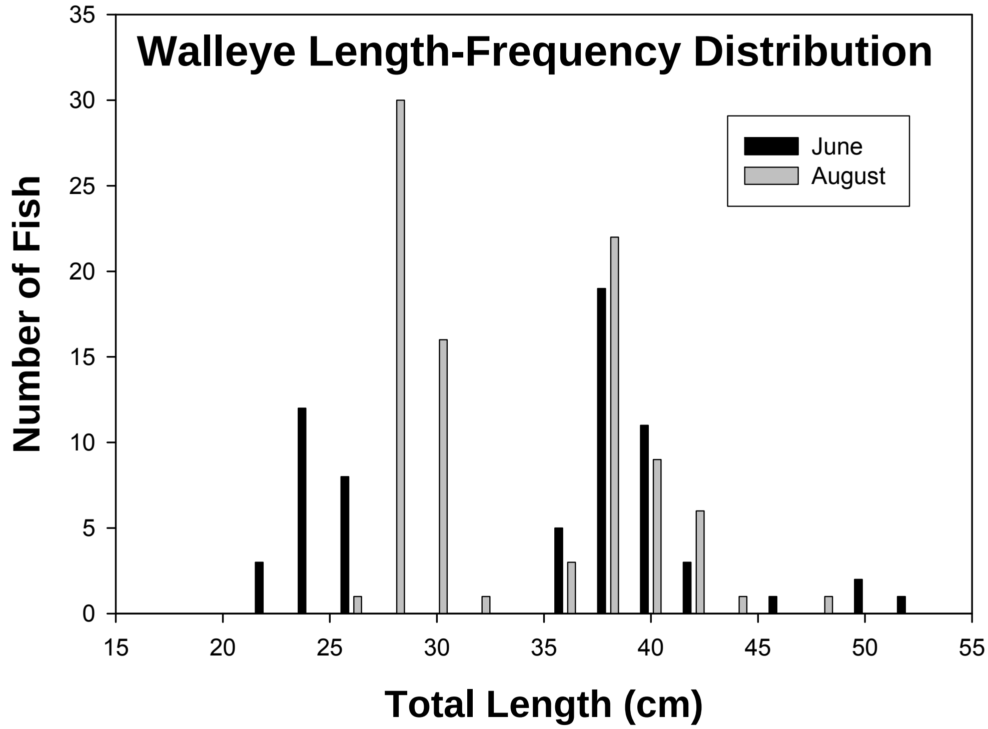


Figure 1. Length-frequency distribution of walleye captured in gill and mini-fyke nets during June and August 2007 in Kenogama Lake.

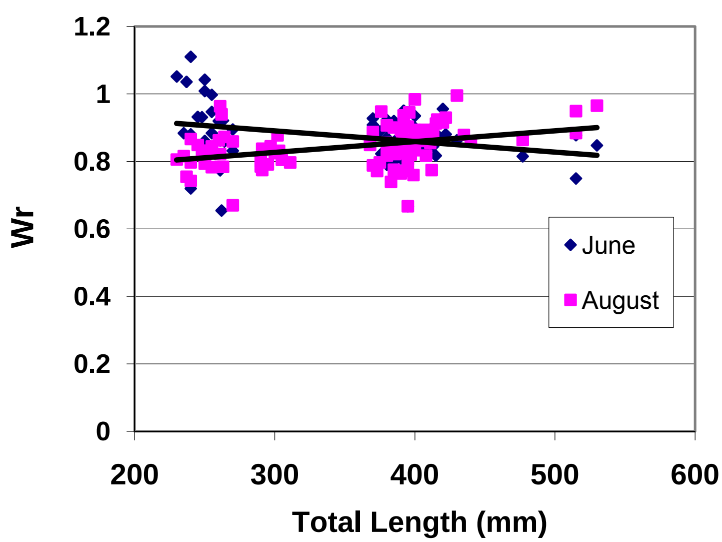


Figure 2. Relative weights (W_r) of walleyes sampled in Kenogama Lake, during June and August 2007 (June Wr = -0.0003x + 0.9853, R^2 = 0.098; August Wr = 0.0003x + 0.7308, R^2 = 0.137).

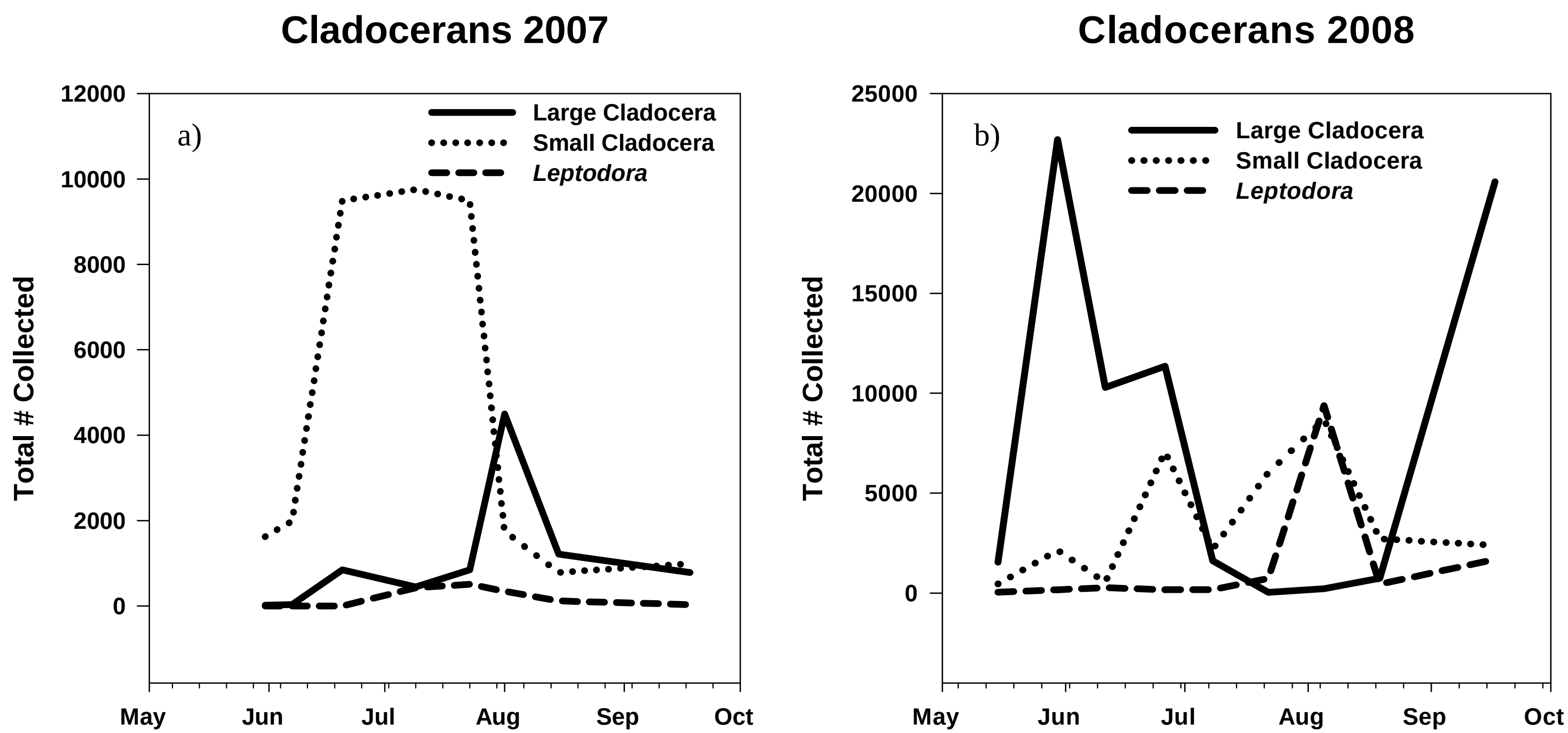
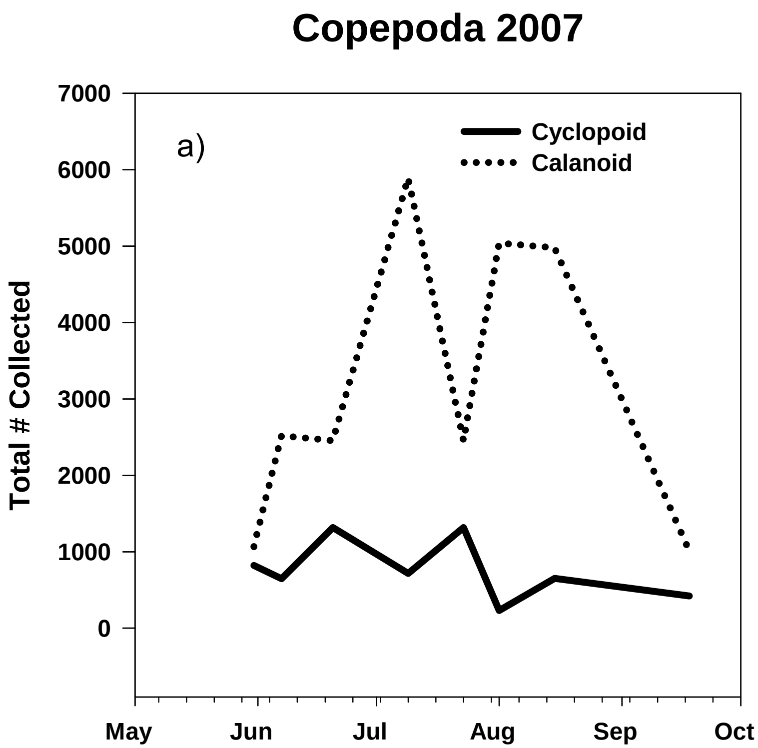


Figure 3. Seasonal patterns in total numbers of cladocerans captured in activity traps (n = 12) and vertical column samplers (n = 12) on each sampling date of 2007 (panel a) and 2008 (panel b). Note the scales differ by years. Large cladocerans include *Daphnia, Ceriodaphnia,* Sididae, and *Simocephalus*; small cladocerans include Bosminidae, Chydoridae, *Diaphanosoma,* and *Eurycercus*.



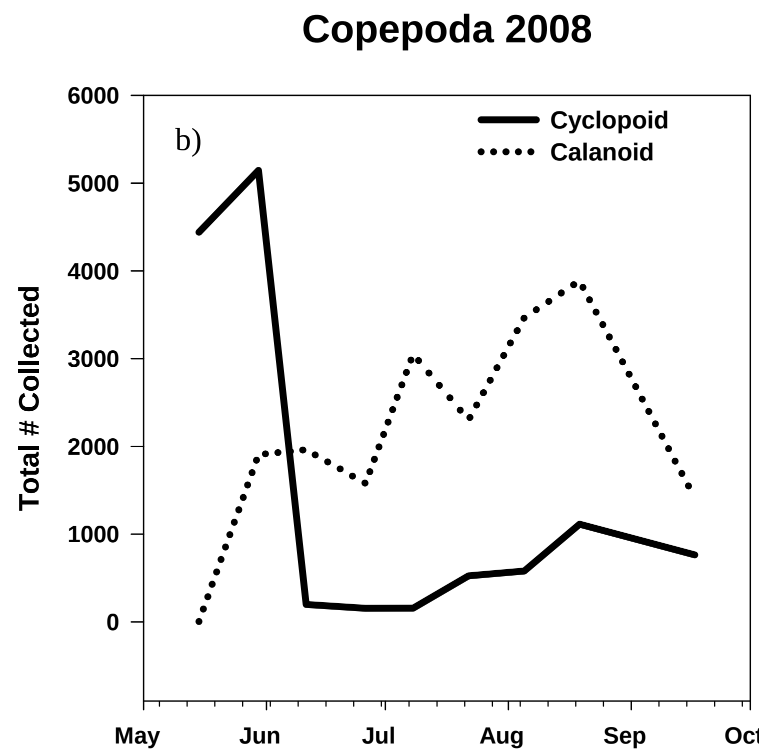


Figure 4. Seasonal patterns in total numbers of copepods captured in activity traps (n = 12) and vertical column samplers (n = 12) on each sampling date of 2007 (panel a) and 2008 (panel b). Note the scales differ by years. Copepoda were classified to only suborder.

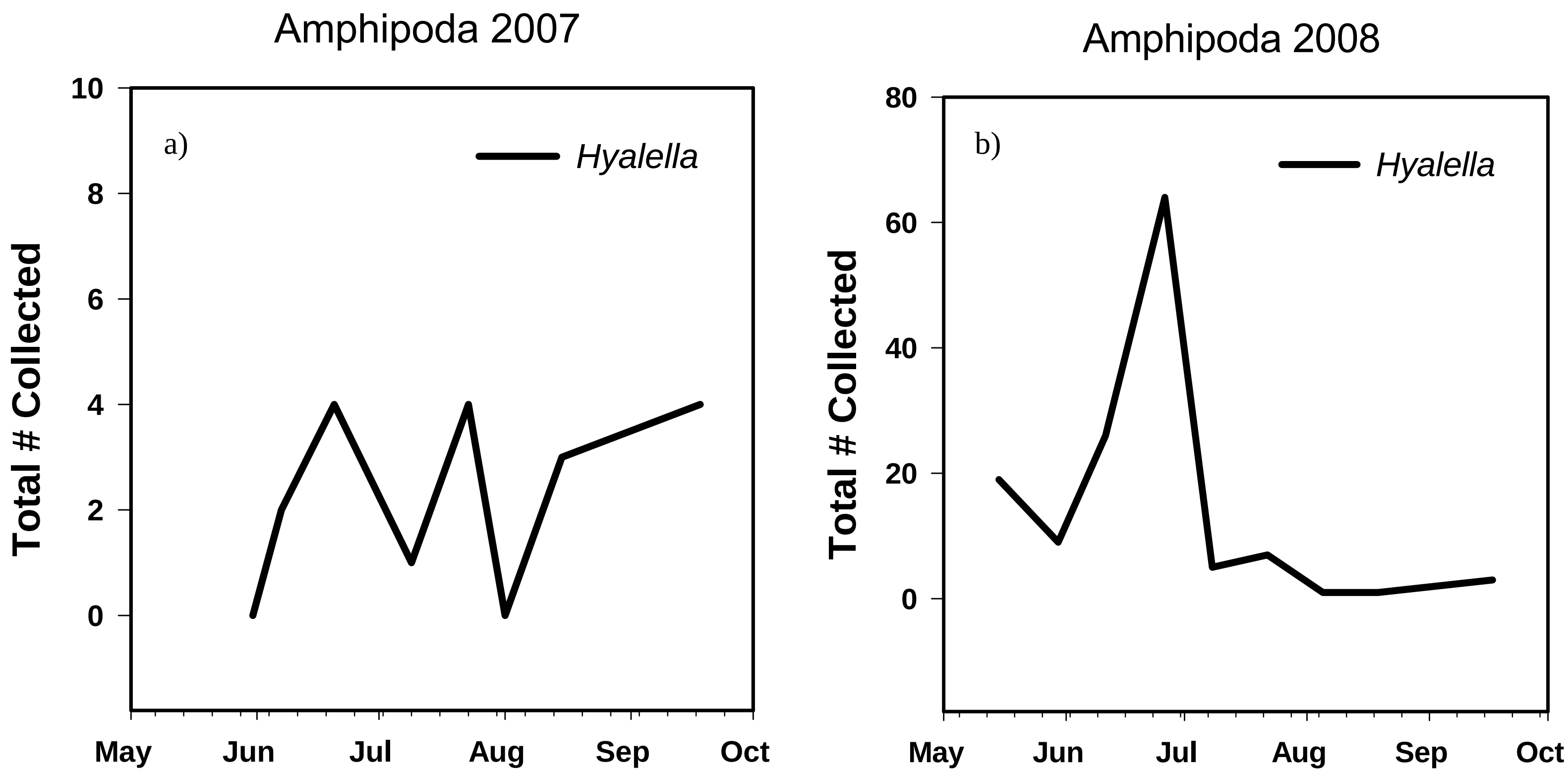


Figure 5. Seasonal patterns in total numbers of amphipods captured in activity traps (n = 12) and vertical column samplers (n = 12) on each sampling date of 2007 (panel a) and 2008 (panel b). Note the scales differ by years.

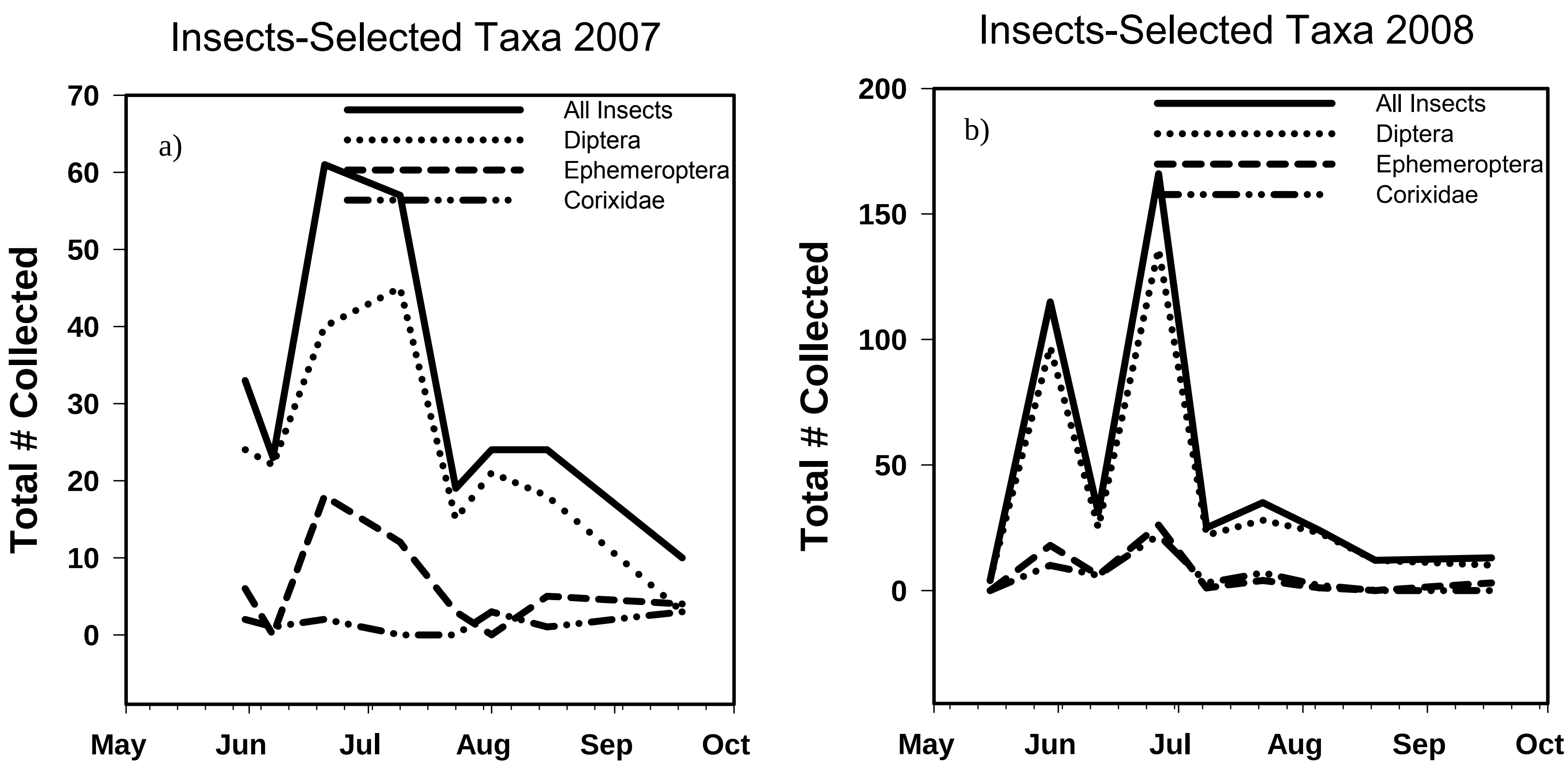


Figure 6. Seasonal patterns in total numbers of insects captured in activity traps (n = 12) and vertical column samplers (n = 12) on each sampling date of 2007 (panel a) and 2008 (panel b). Note the scales differ by years. Selected insect taxa groups of All Insects, Diptera, Ephemeroptera and Corixidae are shown.

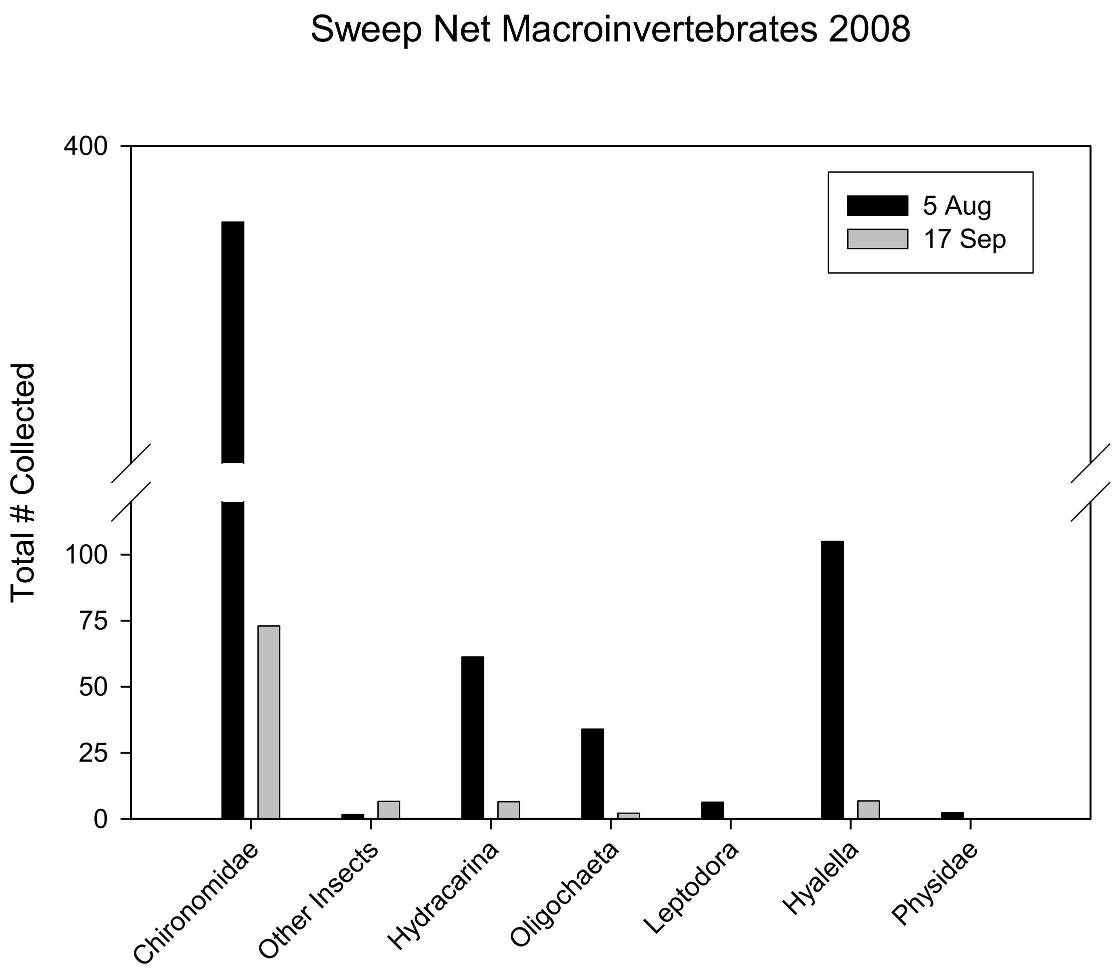


Figure 7. Mean density of aquatic macroinvertebrates collected in sweep nets in August and September 2008.

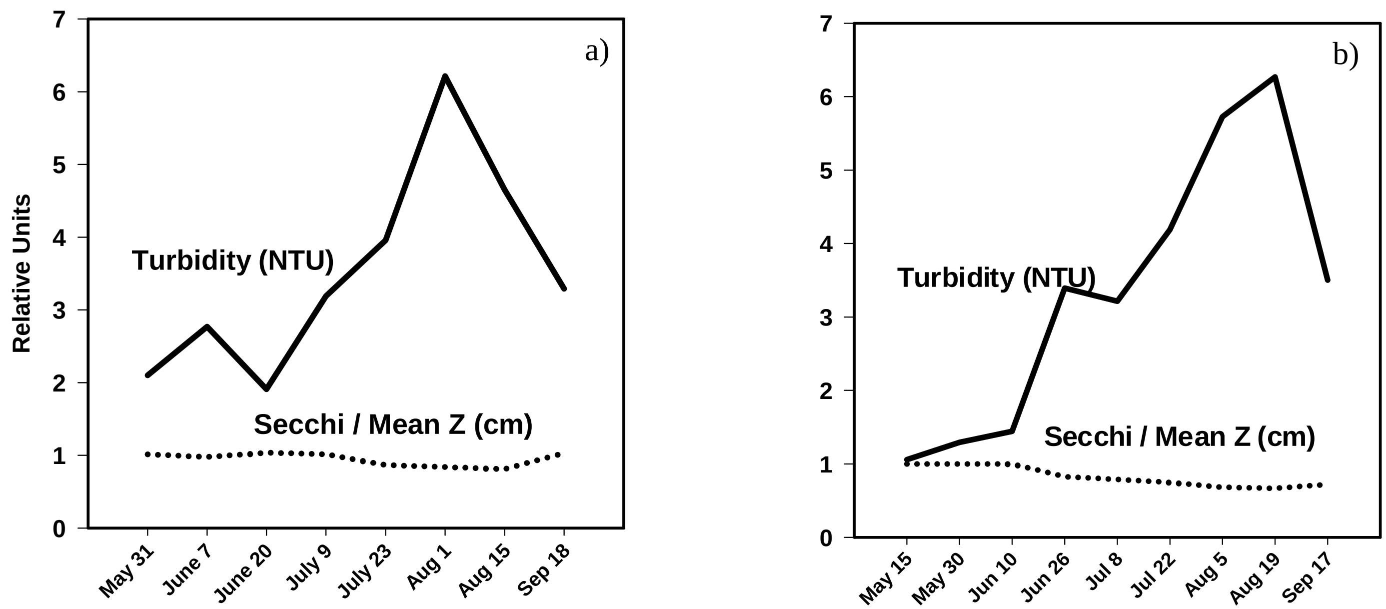


Figure 8. Water transparency patterns at Kenogama Lake during 2007 (panel a) and 2008 (panel b) depicted by ratio of Secchi disk transparency: mean lake depth, and turbidity measured using a nephelometer.

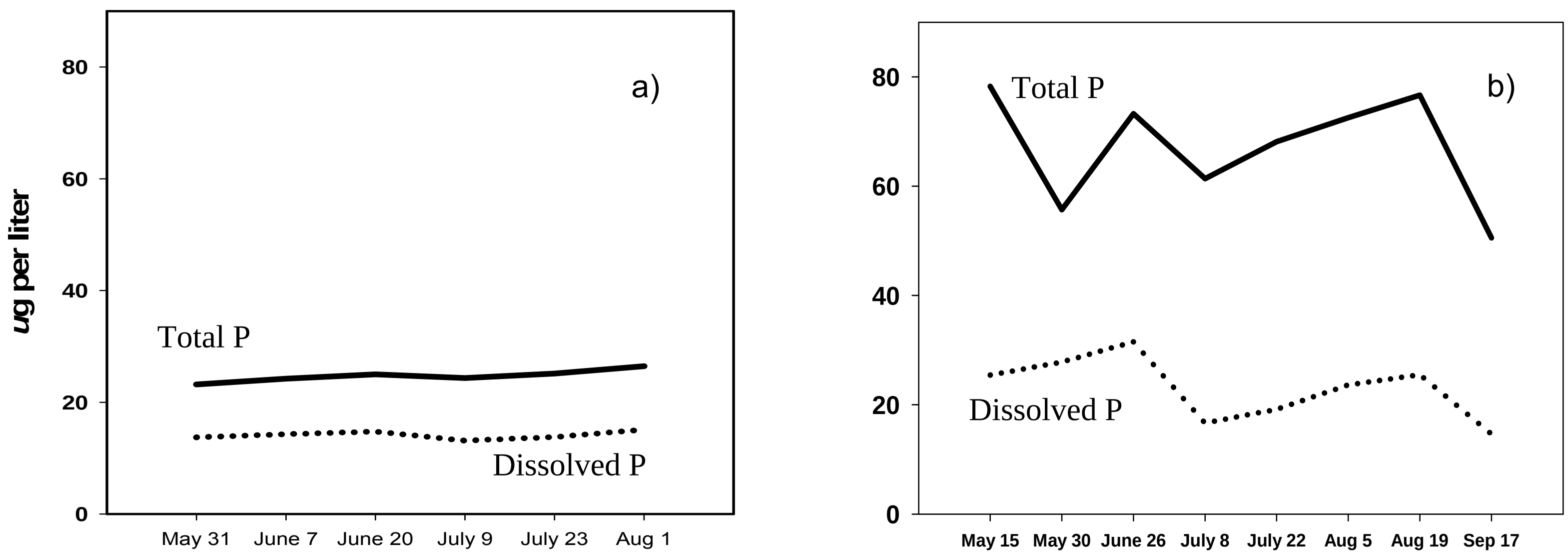


Figure 9. Patterns in mean phosphorus concentrations (total and dissolved forms, solid and dashed lines, respectively) at Kenogama Lake during 2007 (panel a) and 2008 (panel b).

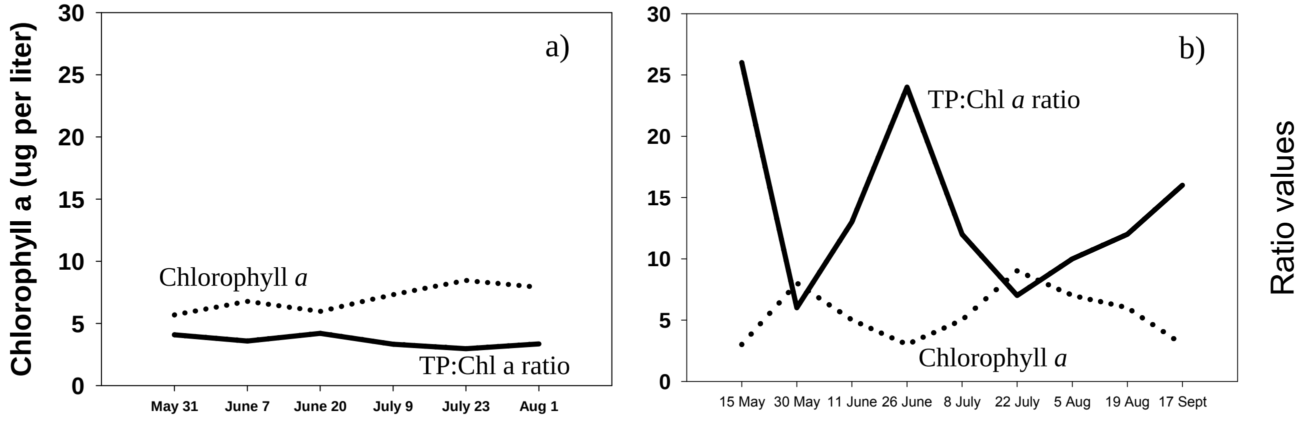


Figure 10. Phytoplankton abundance as indicated by water-column Cholorphyll a concentrations (dashed line) at Kenogama Lake during 2007 (panel a) and 2008 (panel b). Solid line depicts ratios of total phosphorus:chlorophyll *a* (same scale); ratio values above approximately 3 (as are all shown here) are often characteristic of lakes in a turbid-water regime (Dokulil and Teubner 2003).

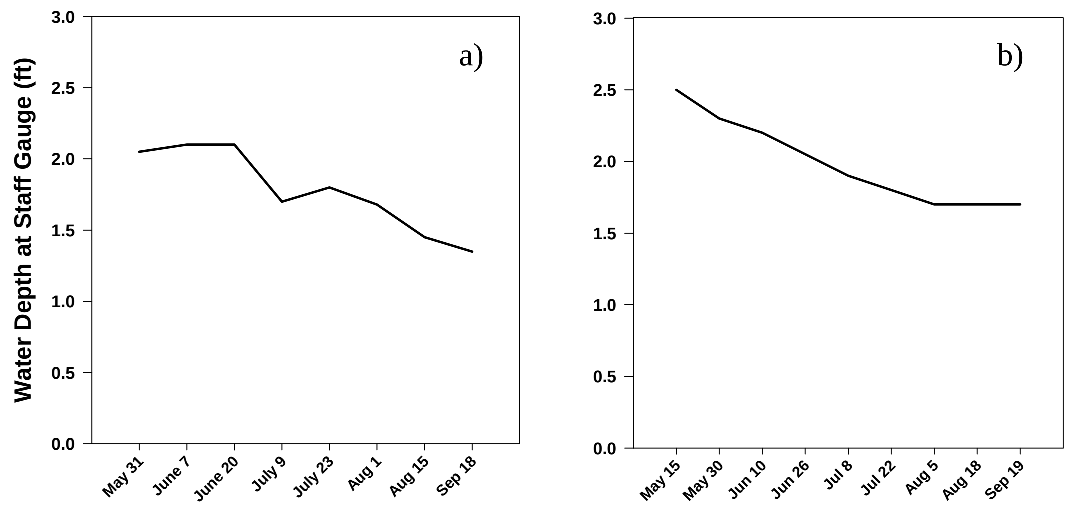
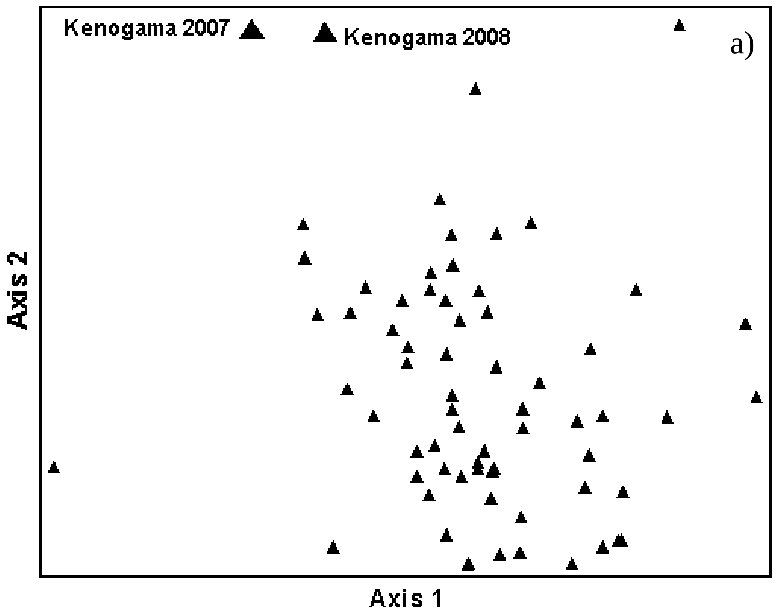


Figure 11. Water levels at the Kenogama Lake staff gauge during 2007 (panel a) and 2008 (panel b).



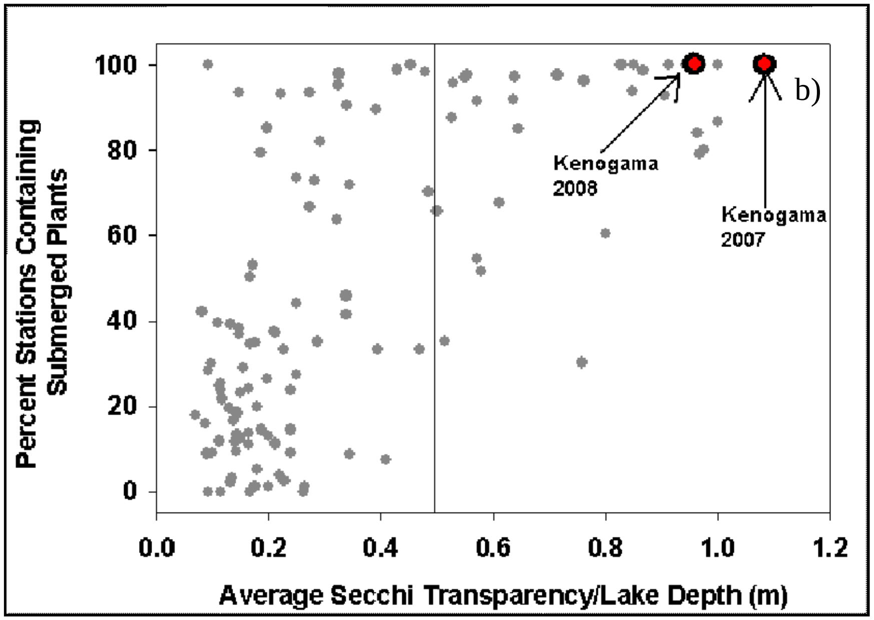


Figure 12. Plant community characteristics depicted by Non-metric Multidimensional Scaling (panel a) based on scores from a combined species matrix containing 71 shallow lakes in western and central MN, 2005 and 2006, and Lake Kenogama, 2007 and 2008. Panel b compares water clarity and macrophyte relationships of Kenogama and other shallow lakes surveyed by MNDNR Shallow Lakes Program (data provided by Nicole Hansel-Welch et al.). Vertical line (Secchi/depth value = 0.5) indicates approximate threshold depth where light penetration is sufficient to support rooted plants at mean lake depth. Separation in NMS space and water clarity/plant relationships indicate extent of similarity in abundance and species composition of Kenogama and water transparency relative to other shallow lakes recently studied in MN.

MANAGING BOVINE TUBERCULOSIS IN WHITE-TAILED DEER IN NORTHWESTERN MINNESOTA: A 2008 PROGRESS REPORT

Michelle Carstensen[1], Erika Butler, Michael DonCarlos, and Lou Cornicelli

SUMMARY OF FINDINGS

Bovine tuberculosis (TB), first discovered in 2005, has now been found in 12 cattle operations in northwestern Minnesota. To date, all of the infected cattle herds have been depopulated and the Board of Animal Health (BAH) has continued to test cattle herds in the area. The strain has been identified as one that is consistent with Bovine TB found in cattle in the southwestern United States and Mexico. In response to the disease being detected in cattle, the Minnesota Department of Natural Resources (MNDNR) began surveillance efforts in free-ranging white-tailed deer (*Odocoileus virginianus*) within a 15-mile radius of the infected farms in fall 2005. To date, 25 deer have been found infected with Bovine TB. All infected deer were sampled within a 164mi^2 area, called the Bovine TB Core, which is centered in Skime, Minnesota, and encompasses 8 of the previously infected cattle farms. In fall 2008, Minnesota was granted a Split-State Status for Bovine TB by the United States Department of Agriculture (USDA) that resulted in a lessening of testing requirements for cattle in the majority of the state (status level = "Modified Accredited advanced"), with a small area in northwestern Minnesota remaining more restrictive (status level = "Modified Accredited"). Also in 2008, the Minnesota State Legislature passed an initiative that allocated funds to buy-out cattle herds located in the Bovine TB Management Zone, spending $3 million to remove 6,200 cattle from 46 farms by January 2009; resulting in the discovery of the 12th infected cattle herd. The remaining cattle farms in the Bovine TB Management Zone (*n* = 27) were required to erect deer-exclusion fencing to protect stored forage and winter feeding areas, costing an additional $690,000 in state funds. In November 2008, the MNDNR conducted Bovine TB surveillance of hunter-harvested white-tailed deer within the newly created Modified Accredited Zone, and results indicated that none of the 1,246 deer tested were positive for the disease. This marked the first large scale surveillance effort that failed to detect the disease in hunter-harvested deer since sampling efforts began in 2005. MNDNR also conducted targeted removal operations in the Bovine TB Core Area, using both aerial and ground sharpshooting, during winters 2007, 2008 and 2009. These intensive winter deer removal operations removed a combined total of 2,163 deer and detected 13 (52%) of the TB-positive deer discovered to date. Further, a recreational feeding ban, covering 4,000mi^2 in northwestern MN, was instituted in November 2006 to help reduce the risk of deer to deer transmission of the disease and enforcement officers have been working to stop illegal feeding activities. The MNDNR will continue to conduct hunter-harvested surveillance for the next 5 years to monitor infection in the local deer population, and consider the continuation of aggressive management actions (e.g., sharpshooting deer in key locations) to address concerns of deer becoming a potential disease reservoir.

INTRODUCTION

Bovine tuberculosis is an infectious disease that is caused by the bacterium *Mycobacterium bovis (M. bovis)*. Bovine TB primarily affects cattle; however, other mammals may become infected. Bovine TB was first discovered in 5 cattle operations in northwestern Minnesota in 2005. Since that time, 2 additional herds were found infected in 2006, 4 more in 2007, and 1 in 2008; resulting in further reduction of the state's Bovine TB accreditation to Modified Accredited in early 2008. By fall 2008, Minnesota was granted a split-state status for TB accreditation that maintained only a small area (2,670mi^2) in northwestern Minnesota as "Modified Accredited," allowing the remainder of the state to advance to "Modified Accredited

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advanced." To date, 25 wild deer have been found infected with the disease in northwestern MN. Although Bovine TB was once relatively common in U.S cattle, it has historically been a very rare disease in wild deer. Prior to 1994, only 8 wild white-tailed and mule deer (*O. hemionus)* had been reported with Bovine TB in North America. In 1995, Bovine TB was detected in wild deer in Michigan. Though deer in Michigan do serve as a reservoir of Bovine TB, conditions in northwestern Minnesota are different. Minnesota has no history of tuberculosis infection in deer or other wildlife, and the *M. bovis* strain isolated from the infected Minnesota herd does not match that found in Michigan. Also, there are much lower deer densities in the area of the infected herds than in the affected areas of Michigan. Further, unlike Michigan, Minnesota does not allow baiting (hunting deer over a food source), which artificially congregates deer and increases the likelihood of disease transmission.

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

METHODS

In 2008, a fall surveillance strategy was developed to meet the sampling goals established in a recent Memorandum of Understanding (MOU) with USDA, signed by both MNDNR and BAH, that required 1,500 deer to be tested for Bovine TB within the newly created Modified Accredited Zone (MAZ), and 300 deer to the immediate south and west of the MAZ boundaries (Figure 1).

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

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

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

Prior to the start of the each winter sharpshooting effort, MNDNR conducted aerial surveys of the Bovine TB Core Area to assess deer numbers and distribution (Figure 2). This information was used to guide sharpshooting activities and estimate the percentage of deer removed from the area.

RESULTS AND DISCUSSION

In fall 2008, we collected 1,246 samples from hunter-harvested deer; 805 samples from within the Modified Accredited Zone and 441 samples outsize the zone (Figure 3). We did not identify any of the deer as “suspects,” meaning they did not have obvious lesions on the lungs or inside the chest cavity that were consistent with clinical signs of Bovine TB. This marks the first large-scale surveillance effort since fall 2005, in which no suspects were identified. Testing of lymph node samples at NVSL has confirmed that there were no positive cases detected during the fall 2008 surveillance. However, the fall sampling effort fell 30% short of its collection goal of 1,800 samples; thus additional deer removal efforts in winter 2009 increased the sampling total to 1,984 deer, or 10% higher than targeted. Apparent prevalence of Bovine TB in the local deer population, sampled throughout a 1,730 to 2,670mi^2 surveillance zone, indicates a significant decreasing trend from 2006–2008 (Table 1, Figure 4).

To supplement the number of samples collected through fall hunter-harvested surveillance and to further reduce deer density in the area where TB-positive deer had been confirmed, MNDNR used both ground and aerial sharpshooting in the Bovine TB Core Area during winters 2007–2009. In total, these operations removed 2,163 deer from the TB Core Area, included 13 TB-positive individuals. The most recent case was a 5.5 year old male found positive during the winter 2009 sharpshooting effort, which removed a total of 738 deer (Figure 5). Disease prevalence in the TB Core Area has decreased dramatically from 2007 to 2009 (Table 1, Figure 4). Although disease prevalence estimates in the TB Core Area are biased due to the limited geographic distribution of TB-positive deer and the increased probability of detecting a positive individual, the decreasing trend is consistent with the large-scale surveillance of the local deer populations in the fall.

Aerial survey results from February 2009 estimated that the deer population in the Bovine TB Core Area was a minimum of 664 ± 87 deer (Figure 2). This is not significantly different than the January 2008 population estimate of 806 ± 133, but less than the February 2007 estimate of 935 ± 150. It is apparent that aggressive deer removal in the TB Core Area through liberalized hunting, disease management permits, landowner shooting permits, and targeted sharpshooting has not been able to dramatically reduce the deer population in this 164mi^2 area. It is likely that that the TB Core Area is home to both migratory and resident deer, some of which may move out of the zone to spring-summer-fall or winter ranges during the year. It is further likely that deer from the surrounding area are immigrating into the TB Core Area as deer numbers are reduced and habitat availability increases. The lack of severe winter weather condition in recent years has also allowed for good overwinter survival, increased reproduction, and recruitment into the local deer population.

The proximity of the TB-infected deer to infected cattle herds, the strain type, and the fact that disease prevalence (<0.2%) 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 6,206 deer in the northwest since 2005, and a total of 25 confirmed culture-positive deer (Figure 6). Further, all deer found infected to date would have been alive in 2005, when the initial detection of Bovine TB in cattle occurred. The lack of infected yearlings or fawns and limited geographic distribution of infected adults further supports that this disease is not being spread efficiently in the local deer population.

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

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

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

ACKNOWLEDGMENTS

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

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Table 1. Number of deer sampled for bovine TB and testing results listed by sampling strategy, fall 2005 to spring 2009, northwestern Minnesota.

Sampling Strategy	**2005**	**2006**	**2007**	**2008**	**2009**	**Totals**
Hunter-harvested (Oct-Jan)	474	942	1,166	1,246	n/a	3,828
# TB-positive	1	5	5	0		
Apparent Prevalence	0.21%	0.53%	0.43%	0.0%		
Sharpshooting (Feb-May)	0	0	488	937	738	2,163
# TB-positive			6	6	1	
Apparent Prevalence			1.23%	0.64%	*0.14%	
Landowner/Tenant	0	90	0	125	0	215
# TB-positive		1		0		
Total Deer Tested	474	1,032	1,654	2,308	738	6,206
Total # TB-positive	1	6	11	6	1	25

*Final culture results from winter 2009 sampling are still pending at NVSL.

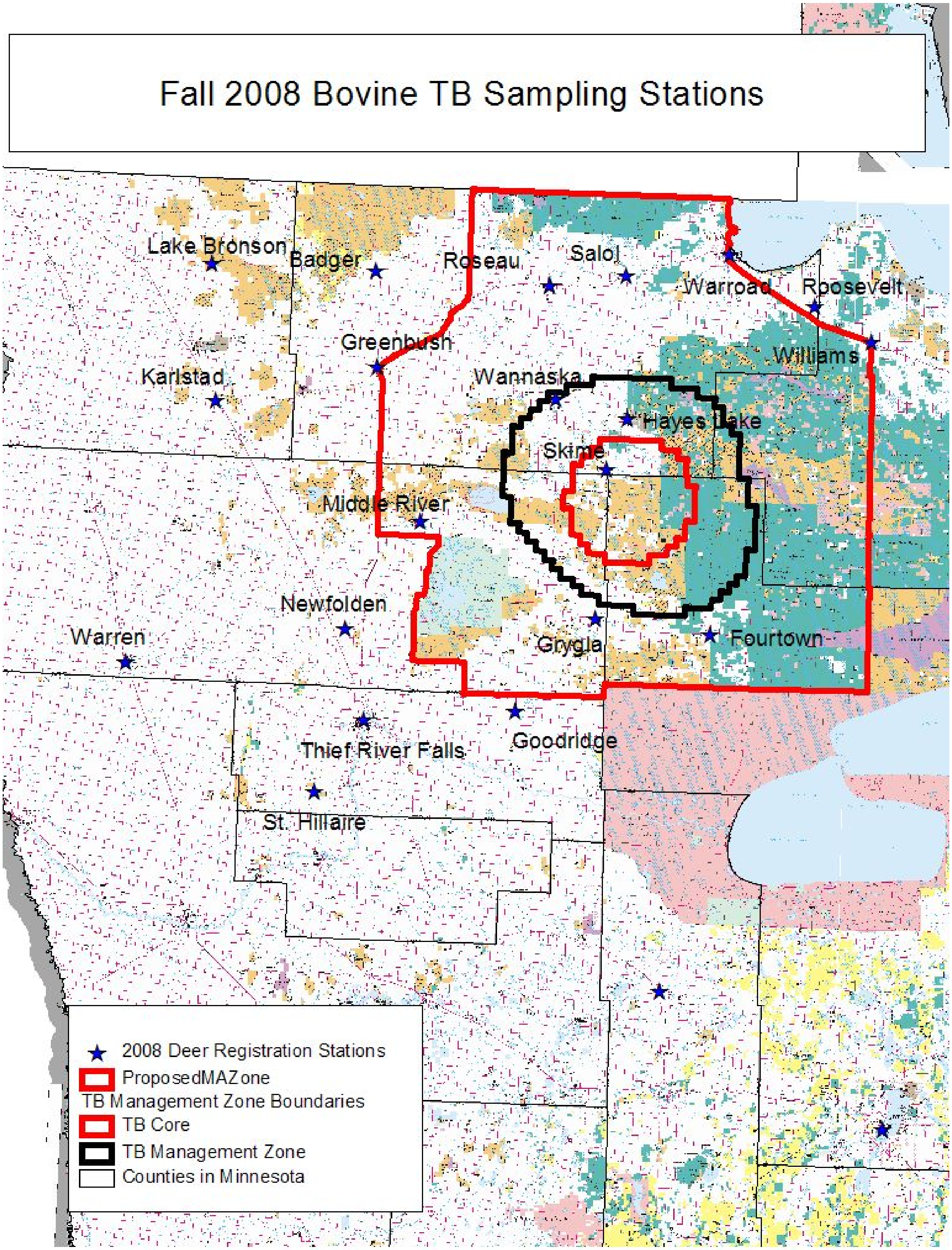


Figure 1. Locations of deer registration stations for sampling hunter-harvested deer for bovine tuberculosis during fall 2008, northwestern Minnesota.

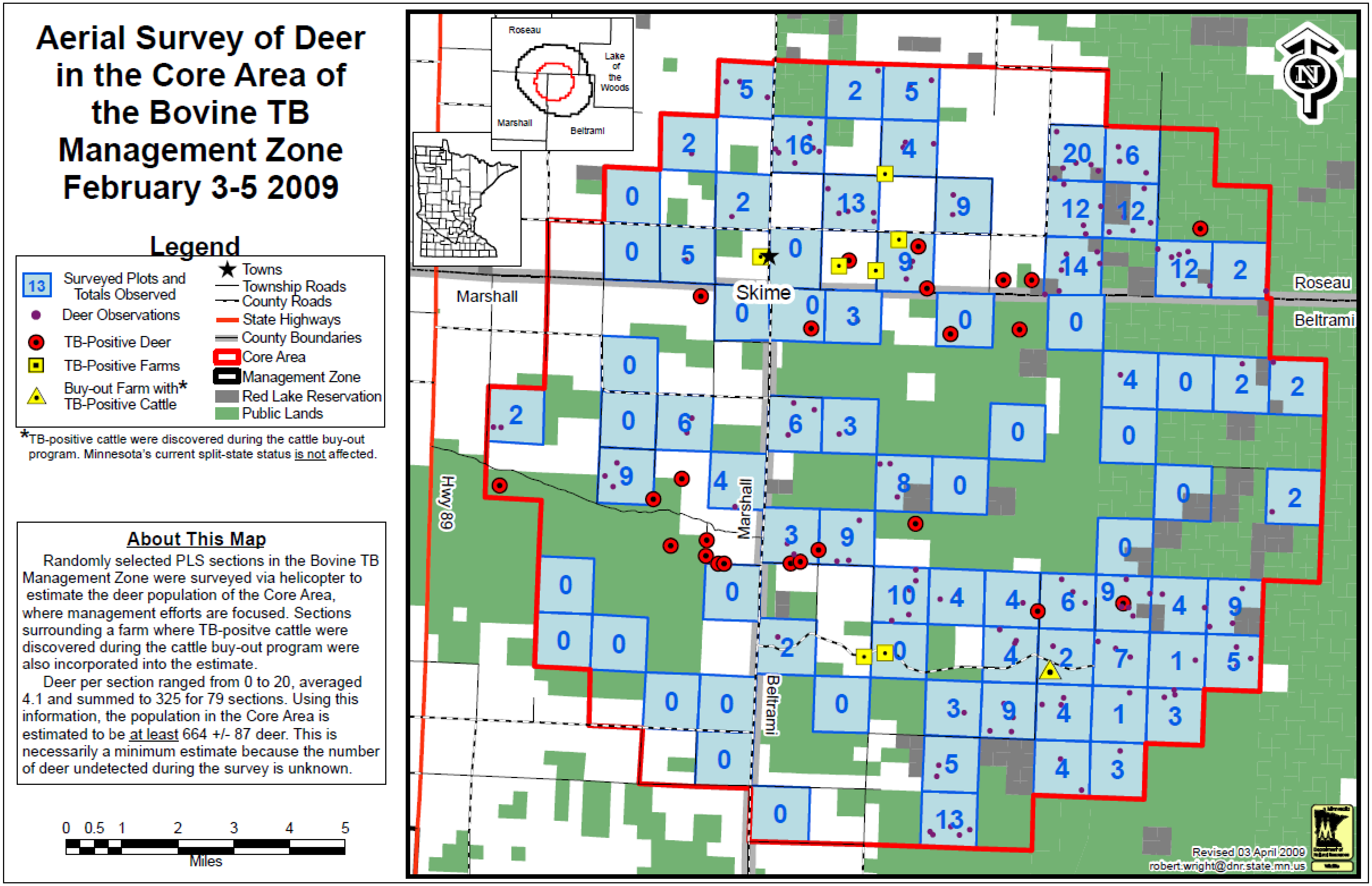


Figure 2. Results of aerial white-tailed deer survey of the Bovine TB Core Area in February 2009, northwestern Minnesota.

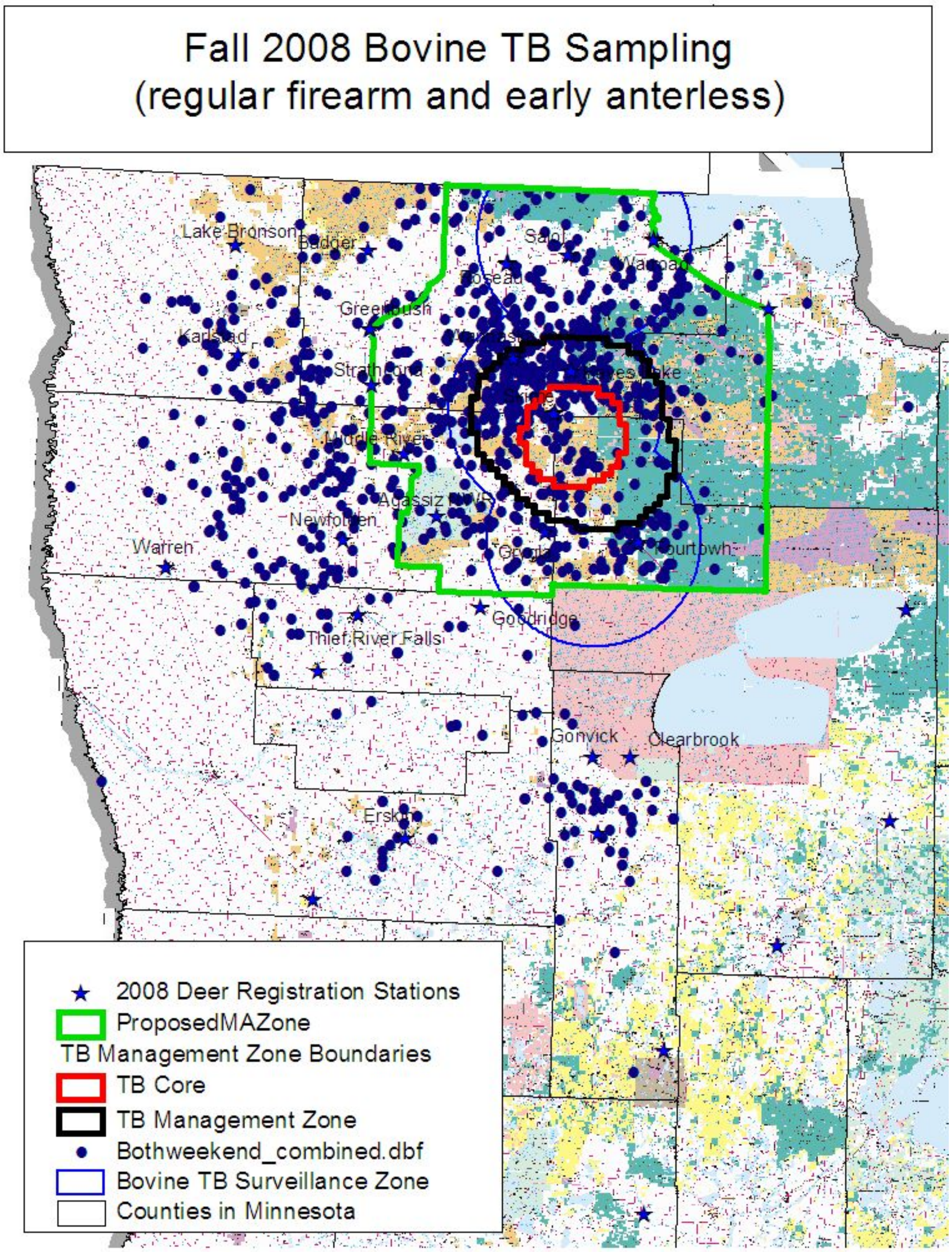


Figure 3. Locations of hunter-harvested deer (*n*=1,246) sampled for Bovine tuberculosis (TB) during fall 2008 in northwestern Minnesota.

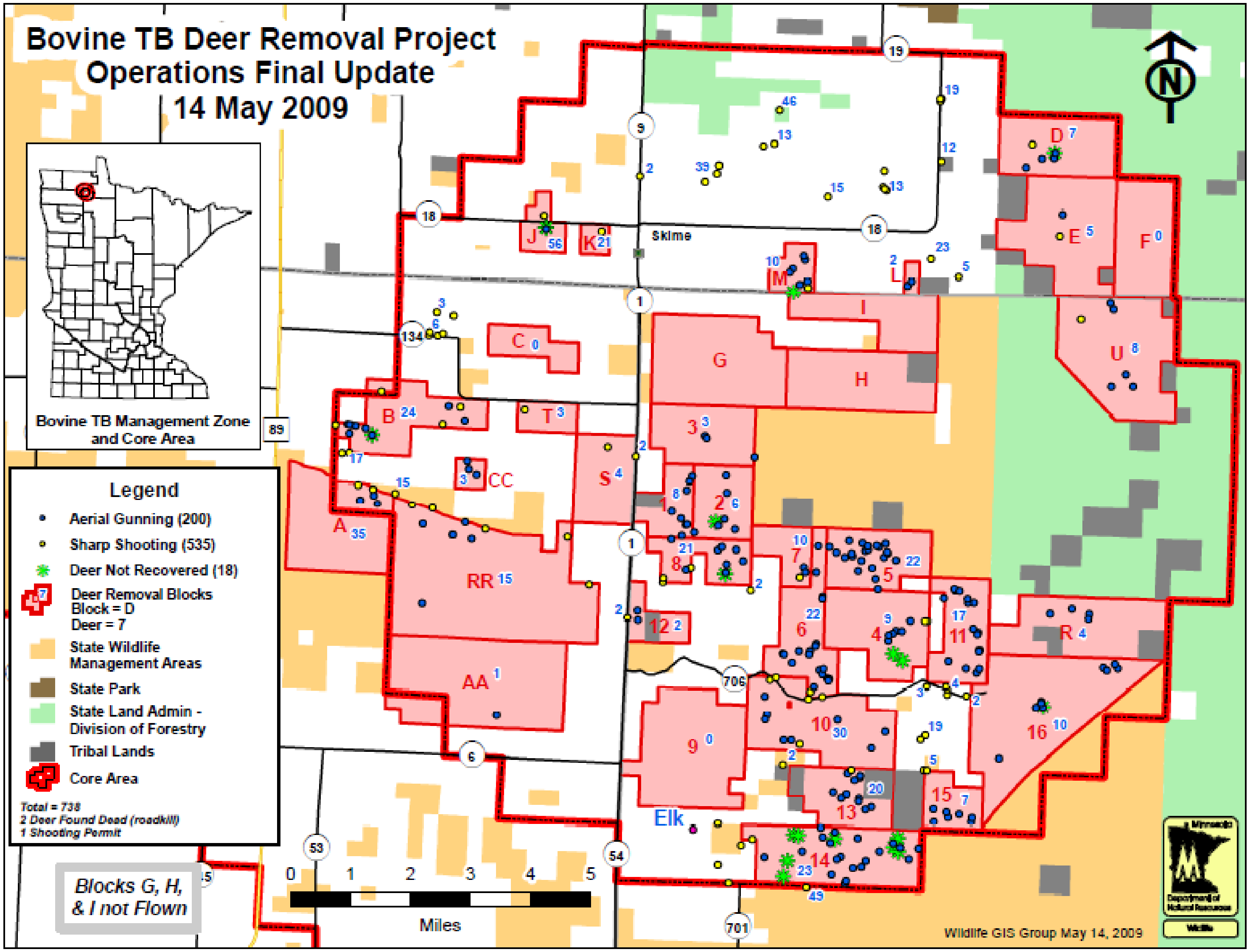
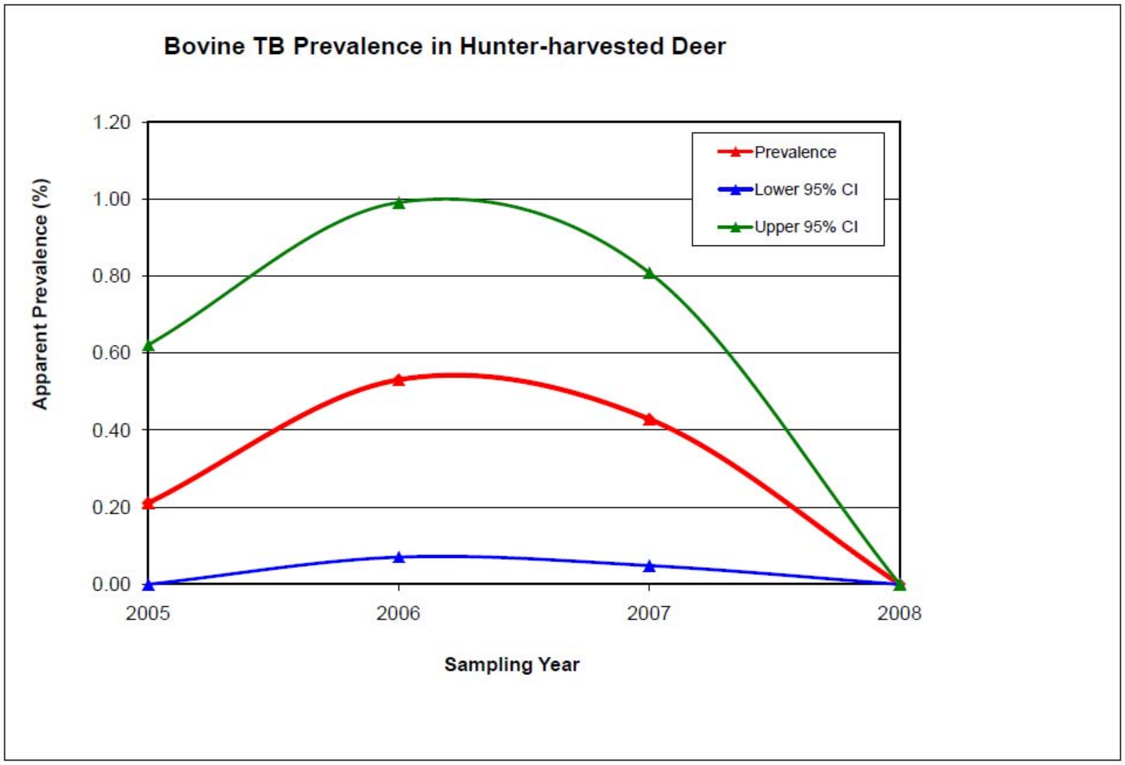


Figure 4. Locations of deer removed (n=738) by USDA ground sharpshooters and aerial gunning during February-April, 2009 within the Bovine TB Core Area, a 164mi^2 area within Bovine tuberculosis Management Zone in northwestern Minnesota.



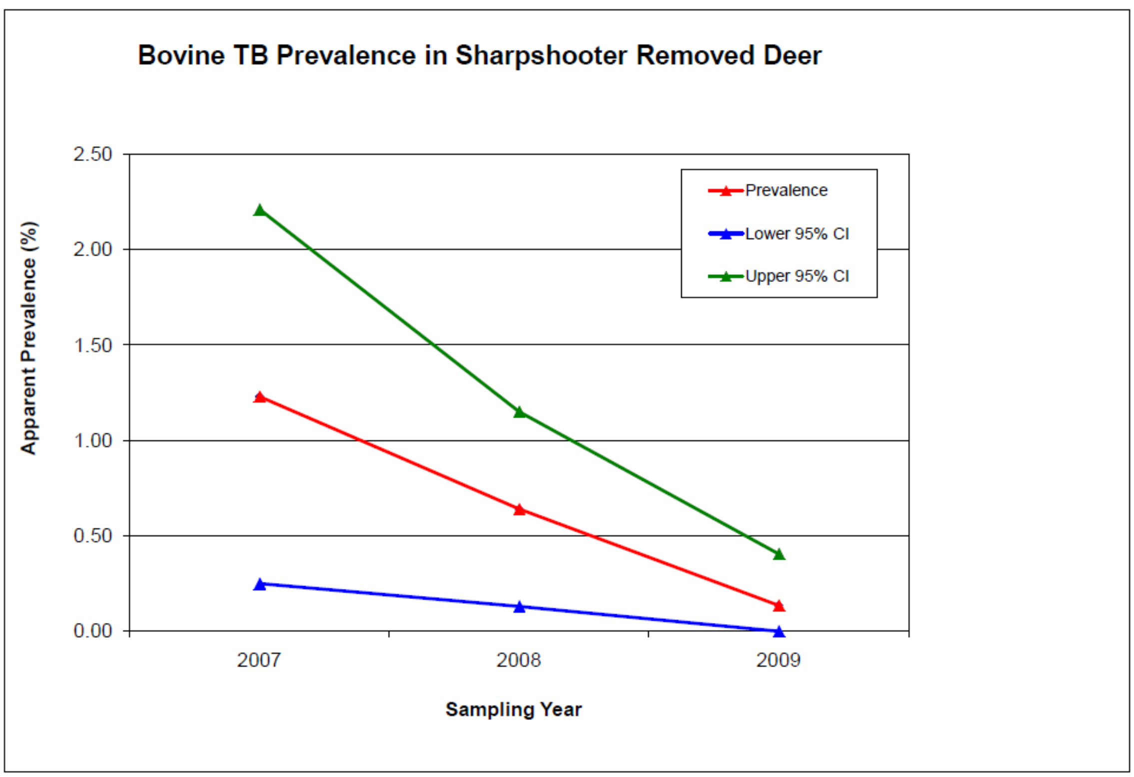


Figure 5. Prevalence of Bovine TB in hunter-harvested deer from 2005–2008 in the Bovine TB Surveillance Zone and disease prevalence from sharpshooter removed deer from 2007–2009 in the Bovine TB Core Area, northwestern Minnesota.

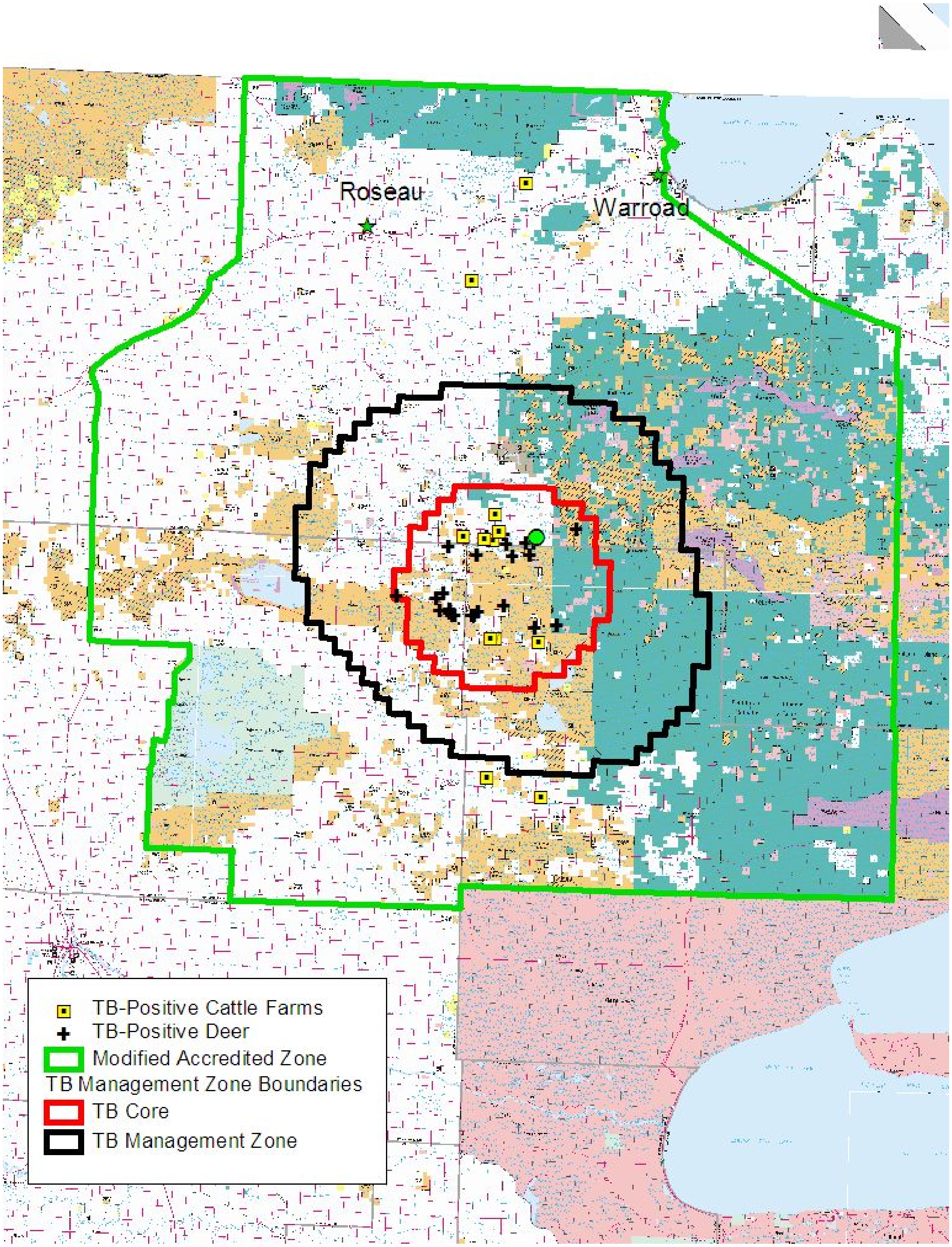


Figure 6. Locations of white-tailed deer found infected (*n*=25) with Bovine TB since fall 2005 in northwestern Minnesota, with the most recent case detected in March 2009 indicated in green. The 12 previously-infected cattle operations are also included.

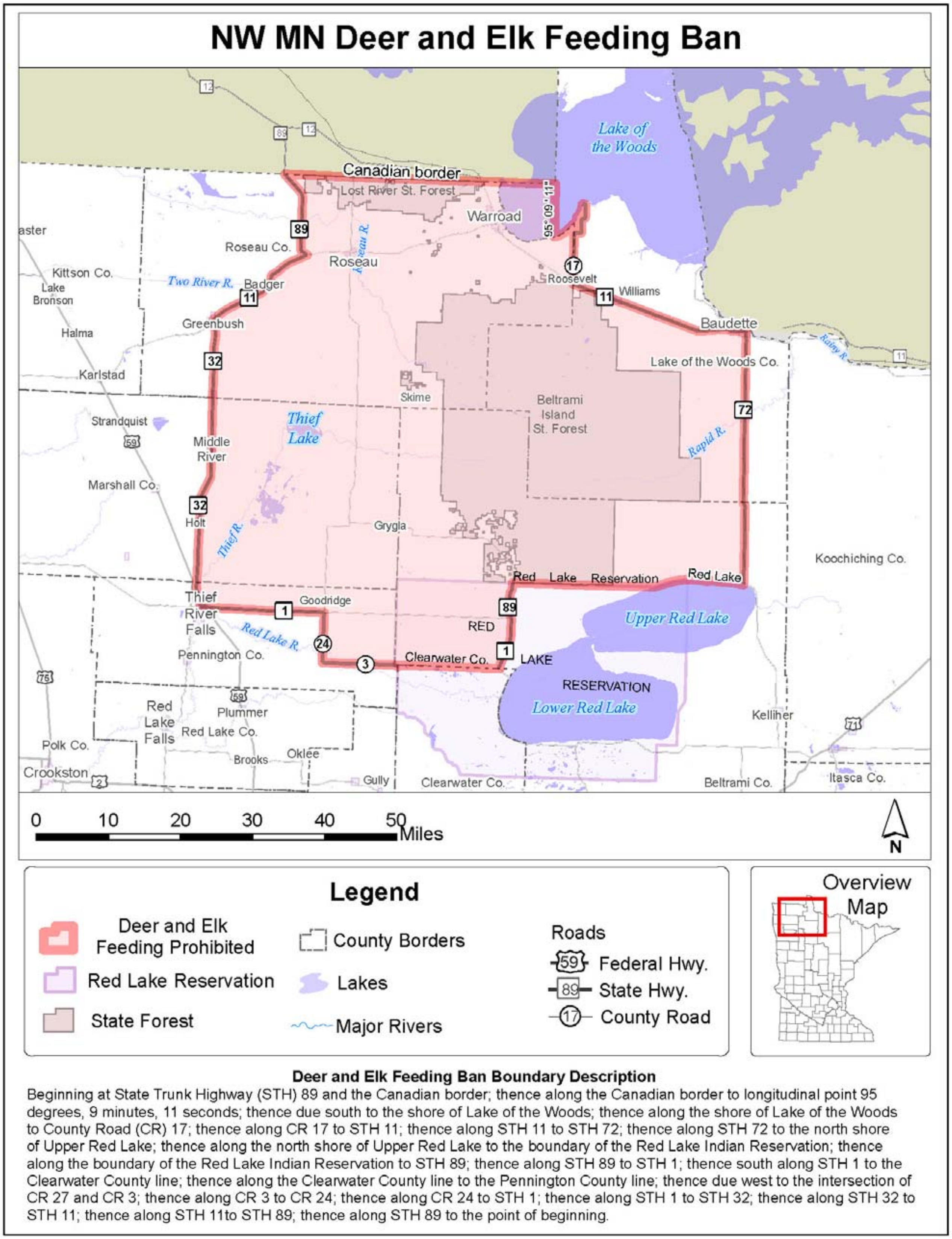


Figure 7. Area in northwestern Minnesota where recreational feeding of deer and elk was banned in November 2006, as a preventative measure to reduce risk of disease transmission.

PRELIMINARY RESULTS FROM THE 2007-2008 MOOSE HERD HEALTH ASSESSMENT PROJECT

Erika Butler[1], Michelle Carstensen, Mike Schrage, David Pauly, Mark Lenarz, and Lou Cornicelli

SUMMARY OF FINDINGS

The purpose of this project is to screen 2007-2009 hunter-harvested (and presumably healthy) moose (*Alces alces*) for a variety of disease agents. The results are intended to indicate which diseases the northeast Minnesota (NE MN) moose population is being exposed to as well allowing for comparisons between similar testing completed on non-hunting moose mortalities from the same population. Positive results confirm moose were exposed to, though not necessarily ill from, eastern equine encephalitis, West Nile Virus, malignant catarrhal fever, *Neospora*, anaplasmosis, bovine herpes virus 1, bovine viral diarrhea virus 1 and 2, *Leptospira sp,* and parainfluenza virus 3. A variety of fecal parasites were identified on fecal examination and multiple organisms were cultured from lung and liver samples. Histological examination was performed on all submitted tissues. All results were negative for *Mycobacterium paratuberculosis*, brucellosis, blue tongue virus, epizootic hemorrhagic disease, chronic wasting disease, and bovine tuberculosis.

INTRODUCTION

Several lines of evidence suggest that the moose population in northeastern Minnesota is declining. Since 2002, annual survival and reproductive rates were substantially lower than documented elsewhere in North America (Lenarz et al. 2007) and population modeling based on these vital rates indicated that the population has declined since at least 2002 (Lenarz unpublished). Recruitment rates and the percent twins observed during aerial surveys have steadily declined since 2002 (Lenarz 2009). In addition, hunter success rates have steadily declined over the past 8 years (Lenarz 2009). Finally, anecdotal reports from local residents have reported a noticeable decline in moose numbers.. Parasites have been documented, including *Parelaphostrongylus tenuis, Echinococcus granulosus, Elaeophora schneideri, Sarcocystis spp*., *Fascloides magna,* and *Dermacenter albipictus*. Copper deficiency has been documented in some moose. Many causes of mortality remain unknown with numerous prime-age animals dying, often during low stress periods of the year. Poor antler development has also been noted in some bull mortalities.

The purpose of this project is to screen 2007-2009 hunter-harvested (and presumably healthy) moose for a variety of disease agents. The results are intended to indicate which diseases the NE MN moose population is being exposed to as well as allowing for comparisons between similar testing completed on non-hunting moose mortalities from the same population. Positive results only indicate that the animal was exposed to the disease agent and are not diagnostic of clinical disease. While some of the test results may be all negative, this does not necessarily mean that the disease is not present or impacting the population. Some diseases cause death so quickly, or without an immune response, that finding a positive in a seemingly healthy animal would be extremely rare.

METHODS

In order to conduct this herd health assessment, hunters (both tribal and state) were asked to collect samples of lung, liver, blood, feces, hair, ticks, and an incisor for aging. We

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provided a presentation and instructions relative to the moose health survey at the mandatory Minnesota Department of Natural Resources (MNDNR) Moose Hunt Orientation Sessions and tribal natural resource offices. Hunters were given a sampling kit with instructions at the orientation sessions. Post-harvest, these samples were dropped off at official registration stations by the hunters when they registered their moose. At the time of registration, hunters were asked to locate their kill site on appropriate maps.

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

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

RESULTS AND DISCUSSION

In 2007, moose hunters at MNDNR registration stations and tribal natural resource offices throughout moose range turned in a total of 135 sampling kits in northeastern Minnesota (Figure 1). Of the kits submitted, 118 were complete, with the reminder being partial submissions.

In 2008, moose hunters turned in 123 sampling kits (Figure 1). Of the kits submitted, 111 were complete. The quality of the samples collected both years was quite good, with very few errors in tissue identification or insufficient quantities. The following is a brief overview of the major findings:

Eastern Equine Encephalitis (EEE)

A combined total of 228 serum samples were submitted to NVSL for Virus Neutralization (VN) testing in 2007 and 2008. Positive results were reported for 14 of the moose (14/228 = 6.1%). Multiple animals had titers ≥ 100. See Table 1 for a breakdown of results by year and titer levels.

The positive results indicate that these animals were exposed to the EEE virus as the VN test prevents cross-reactivity with other viruses. 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.

EEE is spread by mosquitoes and causes neurologic signs and often death. It poses a greater mortality threat for most species than West Nile Virus does (though the effects of EEE infection have not been studied in moose).

West Nile Virus (WNV)

A combined total of 229 samples were submitted to NVSL for VN testing in 2007 and 2008. Positive results were reported for 87 of the moose (87/229 = 37.9%). Multiple animals had titers ≥ 100. See Table 1 for a breakdown of results by year and titer levels.

The positive results indicate that these animals were exposed to the WNV virus as the VN test prevents cross-reactivity with other viruses. 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.

Little is known about the effects of WNV in moose. In white-tailed deer (*Odocoileus virginianus*) it has been found that they often have a low titer and no clinical signs. However, the USDA has found that reindeer (*Rangifer tarandus*) infected with WNV have high mortality rates and high titers, indicating that the virus is more serious for some species than others.

Malignant Catarrhal Fever (MCF)

A combined total of 229 samples were submitted to NVSL for peroxidase-linked assay (PLA) testing in 2007 and 2008. If the PLA test came back positive, the samples were screened with a VN test. A total of 90 samples tested positive on the PLA test (90/229 = 39.3%). All the VN testing results were negative.

The PLA test is more sensitive than the virus isolation, meaning it is much better at identifying true positives. VN is more specific, which means it is better at identifying true negatives. There are a couple of issues with this testing. The PLA reacts with multiple Gammaherpes Viruses (such as the wildebeest strain, the sheep strain, the deer strain, etc). A PLA positive does not indicate what strain has been found, only that one has. The higher the positive value with the PLA test, the stronger the positive in the sample. The VN test only screens for the wildebeest strain (which is exotic to the U.S.) and would be negative if other strains are present. This means a sample that was positive on PLA and negative on VN was likely exposed to a gammaherpes virus, but not the wildebeest strain.

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

Fecal Examination for Parasites

A combined total of 225 fecal samples were screened for evidence of parasites in 2007 and 2008. Parasites were identified in 28 samples (28/225 = 12.4%). In 2007, evidence of parasitism was found in 18 of the samples (18/123 = 14.6%). Five of the samples contained *Nematodirus*, 5 contained *Moniezia*, 6 contained Strongyle type ova, 1 contained *Nematodirus/Moniezia*, and 1 contained *Dictyocaulus.* In 2008, evidence of parasitism was found in 10 of the samples (10/102 = 9.8%). Three of the samples contained *Nematodirus*, 5 contained *Moniezia*, and 2 contained Strongyle type ova.

Negative results do not necessarily mean the animal was parasite free, only that it was not actively shedding at the time the feces were collected.

Fecal Sedimentation

In 2007, a total of 12 fecal samples underwent fecal sedimentation. Sedimentation is used to identify patent liver fluke infection. None of the samples were positive for liver fluke ova. This screening was not repeated in 2008.

Moose are considered dead-end hosts for liver fluke, though reports of moose passing fluke ova in their feces exist. Negative results do not mean that the animals weren't infected with liver flukes, only that they were not actively shedding ova in their feces.

Liver and Lung Culture

In 2007, a total of 121 livers were cultured for bacteria. No significant growth was found in 119 samples, *E. coli* was isolated from 1 sample, and *Pantoea sp*. was isolated from 1 sample. A total of 125 lung samples were submitted for bacterial culture. No significant growth was found in 124 of the samples and *E. coli* was isolated from 1 sample. The *E. coli* isolations are likely due to cross-contamination from contents of the intestinal tract.

The decision was made in 2008 to only culture liver and lung if the histology results warranted it. These results have yet to be reported.

Culture-Other

In 2007, one abscess was submitted and cultured. *Arcanobacterium pyogenes* was isolated. *Arcanobacterium pyogenes* is a bacterium commonly found in infected wounds and abscesses of ruminants and other animals. There were also samples from 2 spleens submitted for culture in 2007. No significant growth was documented in 1, and *Pantoea sp*. was isolated from the other.

Pulmonary *Mycoplasma* Culture

In 2007 a total of 119 lung samples were submitted for *Mycoplasma* culture. None was isolated. This was discontinued in 2008.

***Mycobacterium paratuberculosis* (Johne's)**

A combined total of 192 fecal samples were submitted for *M. paratuberculosis* culture in 2007 and 2008. All culture results were negative. In 2007, PCR was run on 118 fecal samples, with all results negative, and Biocor (serology) was run on 121 samples, with all of the results negative. PCR and Biocor testing were not continued in 2008

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

All species of ruminants are believed to be susceptible to Johne's and it is frequently diagnosed in cattle and sheep (Manning and Collins, 2006). Clinical signs in wild ruminants are similar to those seen in sheep, and 1 moose with diarrhea, which resulted in death, was diagnosed with Johne's (Soltys et al., 1967). Serologic evidence of exposure to Johne's in moose has been documented, with 9/426 (2.1%) seropositive moose in Norway (Tryland et al., 2004).

Anaplasmosis

A combined total of 219 samples were screened for Anaplasmosis (*Anaplasma phagocytopila*, formerly *Ehrlichia phagocytophila*) with the card test in 2008 and 2009. One of these samples was positive (1/219 = 0.5%). Positive test results indicates that exposure to this bacterium is occurring. See Table 1 for a breakdown of results by year and titer levels.

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

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

Borreliosis (Lymes Disease)

A combined total of 221 samples were screened for lymes disease with an immunofluorescence assay (IFA) in 2007 and 2008. Positive results were reported for 41 of the samples (41/221 = 18.6%). See Table 1 for a breakdown of results by year and titer levels.

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

Brucellosis

A combined total of 205 samples were submitted in 2007 and 2008 for *Brucella* screening with the card test. All of the results were negative. These negative results indicate that these animals were not likely exposed to the bacterium. See Table 1 for a breakdown of results by year.

While naturally occurring fatal *Brucella* infections have been documented in free ranging moose (Honour and Hickling, 1993) and serologic evidence suggests that moose are being exposed to *Brucella sp*. (Zarnke, 1983), evidence suggests that the prevalence is low (Honour and Hickling, 1993).

Bovine Viral Diarrhea Virus (BVD) 1 & 2

A combined total of 230 samples were submitted for serum neutralization (SN) testing for BVD 1 & 2 in 2007 and 2008. Positive results were reported for 3 of the samples (3/230 = 1.3%). These results indicate that the moose population is being exposed to BVD. See Table 1 for a breakdown of results by year and titer levels.

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

Bovine Herpes Virus 1 (BHV)

A combined total of 230 samples were screened for BHV using a SN test in 2007 and 2008. One results was reported as positive (1/230 = 0.4%). See Table 1 for a breakdown of results by year and titer levels.

BHV is a disease of the respiratory tract. It is believed to infect all ruminant species and has been isolated from a large number of wild species. It is most commonly isolated in feedlot cattle.

Blue Tongue Virus (BTV)

A combined total of 231 samples were screened using a Competitive Enzyme-Linked Immunoabsorbent Assay (cELISA) for BTV in 2007 and 2008. All results were negative. See Table 1 for a breakdown of results by year.

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

Epizootic Hemorrhagic Disease (EHD)

A combined total of 231 samples were screened for EHD using an Agar Gel Immuno Diffusion (AGID) test in 2007 and 2008. All results were negative. See Table 1 for a breakdown of results by year.

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

Leptospira sp.

A combined total of 231 samples were screened for 6 species of *Leptospira* using a microscopic agglutination test (MAT) in 2007 and 2008. Positive results per species are reported below. See Table 1 for a breakdown of results by year and titer levels.

- *L. bratislava*:
  - 4/231 (1.7%)
- *L. canicola*:
  - 2/231 (0.9%)
- *L. grippothyphosa*:
  - 4/231 (1.7%)
- *L. hardjo*:
  - 0/231
- *L. interrogans* serovar *icterohaemorrhagicae*:
  - 16/231 (6.9%)
- *L. pomona*:
  - 13/231 (5.6%)

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

Neospora sp.

A combined total of 232 samples were screened for *Neospora* with an ELISA test in 2007 and 2008. Positive results were reported for 9 samples (9/232 = 3.9%). See Table 1 for a breakdown of results by year and titer levels.

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

Antibodies to Neospora have been found in numerous species of wildlife, including 8/61 moose from NE MN (Gondim et al., 2004).

Parainfluenza Virus 3 (PI)

A combined total of 232 samples were screened for PI using a hemagglutination inhibition (HI) test in 2007 and 2008. There was 1 positive result (1/232 = 0.4%). See Table 1 for a breakdown of results by year and titer levels.

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

Chronic Wasting Disease (CWD)

In 2007, a total of 14 obex samples and 23 retropharyngeal lymph nodes were screened for CWD using immunohistochemistry (IHC). In 2008, 32 obex samples and 33 lymph node samples were screened for CWD. All results were negative.

CWD is a transmissible spongiform encephalopathy that causes neurological disease in cervids. CWD is known to occur in moose, but has never been documented in wild cervids in MN.

Bovine Tuberculosis

In 2007, 23 sets of head lymph nodes (parotid, retropharyngeal, and submandibular) were collected and cultured for *Mycobacterium bovis*. In 2008, 33 sets of head lymph nodes were submitted. All results were negative.

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

Brain Histopathology

A total of 23 brains were collected and submitted for histopathology. These results are pending at this time. This examination is meant to help identify if there are chronic migration tracts (presumably due to *P. tenuis*) present in the brains of apparently healthy animals.

Liver Histopathology

In 2007, a total of 114 liver samples underwent histological examination. There were no significant findings with 57 of the samples. Thirty-nine of these samples had a diffuse, hepatocellular lipidosis, of which 27 were classified as mild and 12 were classified as moderate.

Fourteen of the samples exhibited varying types and degrees of hepatitis. Perihepatitis was described in 3 samples. Four of the samples exhibited evidence of fluke infection, either currently or previously. Three samples exhibited fibrosis. There were single cases of lymphoid hyperplasia, hydatid cysts, and possible capsulitis/peritonitis. The results from samples collected in 2008 have yet to be evaluated.

Lung Histopathology

In 2007, a total of 126 lung samples underwent histological examination. There were no significant findings in 93 of the samples examined. Pulmonary hemorrhage, likely related to the gunshot, was documented in 10 of the samples. Hydatid cysts, likely *Echinococcus*, were found in 5 samples. Lymphoid hyperplasia was observed in 6 samples. Four samples had chronic pleuritis. Varying types and degrees of pneumonia were found in 4 samples. Single cases of bronchitis, emphysema, an eosinophilic granulama, and intrabronchial foreign material (likely agonal aspiration) were reported. The results from samples collected in 2008 have yet to be evaluated.

Other Histology

In 2007, a total of 24 brainstem samples underwent histologic examination. Twenty-three had no significant findings and 1 had mild hemorrhaging, which was likely related to the gunshot. Twenty-one lymph nodes were examined. Twenty exhibited no significant findings and 1 of them had blood resorption, which was likely related to the gunshot. Fifteen spleens were examined. None of them exhibited any significant findings. One sample of cerebellum, kidney, heart, and brain were examined, with no significant findings. One sample of the colon and small intestine were examined and found to have enteritis. The results from samples collected in 2008 have yet to be evaluated.

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Table 1. Serology results from the moose herd health assessment broken down by year.

Year	Disease	n	Positive	Comments
2007	EEE	116	5 (4.3%)	Titers: 2 @ 100, 3 @ ≥100
2008	EEE	112	9 (8%)	Titers: 4 @ 10, 1 @ 100, 4 @ ≥100
2007	WNV	117	45 (38.5%)	Titers: 32 @ 10, 6 @ 100, 7 @ ≥ 100
2008	WNV	112	42 (37.5%)	Titers: 34 @ 10, 5 @ 100, 3 @ ≥ 100
2007	MCF	117	8 (6.8 %)	Titers: 4 @ 20, 4 @ 100
2008	MCF	112	82 (73.2%)	Titers: 71 @ 20, 11 @ 100
2007	Anaplasmosis	117	1 (0.9%)	
2008	Anaplasmosis	102	0	
2007	Borreliosis	111	38 (34.2%)	Titers: 7 @ 80, 7 @ 160, 12 @ 320, 3 @ 640, 9 @ 1280
2008	Borreliosis	110	3 (2.7%)	Titers: 1 @ 160, 1 @ 320, 1 @ 640
2007	Brucellosis	112	0	
2008	Brucellosis	93	0	
2007	BVD 1 and 2	120	2 (1.7%)	Titers: 1 @ 1024/4096, 1 @ 128/256
2008	BVD 1 and 2	110	1 (1%)	Titers: 1 @8/8
2007	BHV	120	0	
2008	BHV	110	1 (1%)	Titers: 1 @ 8
2007	BTV	121	0	
2008	BTV	110	0	
2007	EHD	121	0	
2008	EHD	110	0	
2007	L. bratislava	121	4 (3.3%)	Titers: 2 @ 100, 2 @ 200
2008	*L. bratislava*	110	0	
2007	*L. canicola*	121	2 (1.7%)	Titers: 1 @ 100, 1 @ 200
2008	*L. canicola*	110	0	
2007	*L. grippothyphosa*	121	3 (2.5%)	Titers: 2 @ 100, 1 @ 200
2008	*L. grippothyphosa*	110	1 (1%)	Titers: 1 @ 100
2007	*L. hardjo*	121	0	
2008	*L. hardjo*	110	0	
2007	*L. interrogans* serovar *icterohaemorrhagicae*	121	2 (1.7%)	Titers: 1 @ 100, 1 @ 200
2008	*L. interrogans* serovar *icterohaemorrhagicae*	110	14 (12.7%)	Titers: 11 @ 100, 3 @ 200
2007	L. pomona	121	10 (8.3%)	Titers: 4 @ 100, 1 @ 200, 5 @ 400
2008	*L. pomona*	110	3 (2.7%)	Titers: 1 @ 200, 2 @ 400
2007	Neospora	122	0	
2008	Neospora	110	9 (8.2%)	
2007	PI	122	1 (0.8%)	Titers: 1 @ 10
2008	PI	110	0	

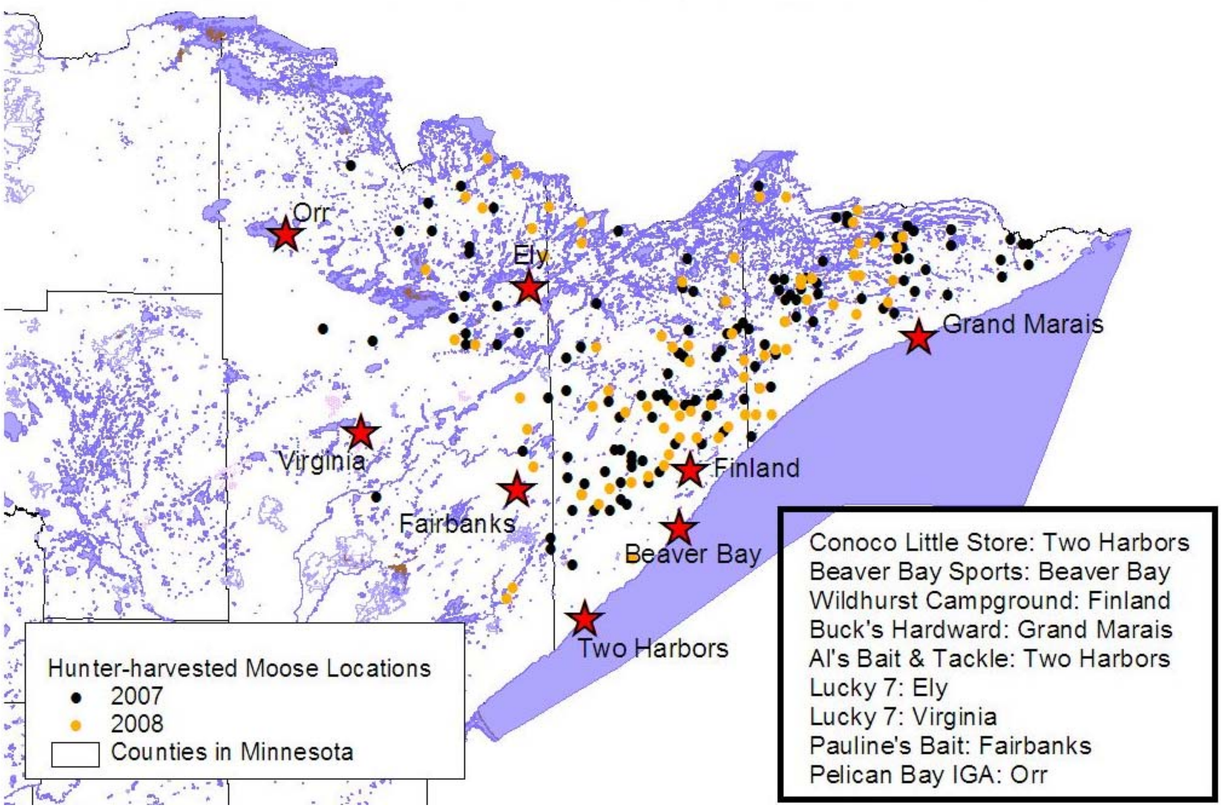

Figure 1. Locations of 2007 and 2008 hunter-harvested moose included in health assessment project in Minnesota.

SURVEILLANCE FOR HIGHLY PATHOGENIC AVIAN INFLUENZA IN MINNESOTA'S WATERFOWL

Michelle Carstensen[1] and Michael DonCarlos

SUMMARY OF FINDINGS

As part of a national strategy for early detection of highly pathogenic avian influenza (HPAI) in North America, Minnesota Department of Natural Resources (MNDNR) and the United States Department of Agriculture (USDA) conducted surveillance for the virus in waterfowl in the state. A combined total of 1,547 birds were sampled for HPAI in Minnesota during 2008. Testing did not result in any positive cases of HPAI, especially the Asian strain of subtype H5N1, however numerous ducks (*n*=43) did test positive for a low pathogenic strain of avian influenza with the subtype H5. Approximately 65,000 wild birds were sampled throughout the United States in 2008, and no positive cases of HPAI were detected. It is likely that Minnesota will continue surveillance for the virus in the state's waterfowl next year, in cooperation with the Mississippi Flyway, Council of the U.S. Fish and Wildlife Service, and the USDA.

INTRODUCTION

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

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

In July 2008, the MNDNR entered into a $90,000 cooperative agreement with the United States Department of Agriculture's Wildlife Services (USDA-WS) to sample 800 wild birds (either live-caught or hunter-harvested) in Minnesota for HPAI-H5N1 during 2008. In addition to the 800 samples to be collected by MNDNR, USDA-WS was also planning to collect a similar number of samples in the state during the same period. Bird species that were targeted include those listed as priority species in the National Strategic Plan or approved for sampling in Minnesota by the Mississippi Flyway Council.

Avian influenza is a viral infection that occurs naturally in wild birds, especially waterfowl, gulls, and shorebirds. It is caused by type A influenza viruses that have 2 important surface antigens, hemagglutinin (H) and nuraminidase (N), that give rise to 144 possible virus subtypes. Influenza viruses vary widely in pathogenicity and ability to spread among birds. The emergence of an Asian strain HP-H5N1 virus in 1996 and subsequent spread of the virus in Asia, Africa, and Europe has killed thousands of wild birds and millions of domestic poultry. In 1997, HP-H5N1 became zoonotic in Hong Kong and to-date has infected at least 423 humans in Eurasia and Africa, resulting in over 258 deaths.

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METHODS

The MNDNR planned to sample 100 common goldeneye (*Bucephala clangula*), 100 ring-necked ducks (*Aythya collaris*), 50 mallards (*Anas platyrhynchos*), and 50 Canada geese (*Branta Canadensis*) during the summer months, primarily in conjunction with planned banding activities. In the fall, through hunter-harvested surveillance, sampling targets were as follows: 100 Northern pintails (*Anas acuta*), 100 mallards, 100 American green-winged teal (*Anas crecca*), 100 American blue-winged teal (*Anas discors*), 50 Northern shovelers (*Anas clypeata*), and 50 American wigeon (*Anas americana*). USDA-WS planned to sample a similar number of either the duck species mentioned above or others from their functional group (e.g., dabblers, divers, shorebirds) as well as 50 Canada geese. If sampling goals per species could not be met, other targeted waterfowl species within the same functional group could be sampled and counted toward the state's total. Sampling strategies were coordinated between the MNDNR and USDA-WS to maximize access to targeted birds species through existing banding operations and fall hunter-harvested surveillance.

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

RESULTS AND DISCUSSION

From July 1, 2008 through March 31, 2009 MNDNR and USDA collected a total of 1,547 samples from wild-caught live birds (*n*=519), hunter-harvested birds (*n*=961), agency (USDA-WS) harvested (*n*=29), and mortality/morbidity events (*n*=38). USDA also collected 716 fecal samples. Thus, a combined total of 2,263 bird samples were screened for HPAI-H5N1 in Minnesota in 2008 (Figure 1, Table 1).

Testing did not result in any positive cases of HPAI-H5N1; however 10 different duck species tested positive for a low pathogenic strain of avian influenza with the subtype H5, and only 1 tested positive for a N1 subtype (Figure 2, Table 2). The testing protocol was limited to the screening for H5, H7, and N1 subtypes only; however in some cases other subtypes were identified and reported.

According to the latest numbers on the United States Geologic Survey's website (http://wildlifedisease.nbii.gov/ai/), approximately 65,000 birds have been sampled for HPAI-H5N1 in the U.S. in 2008. No positive cases of HPAI-H5N1 have been found anywhere in North American to date. Since the majority of H5 positives (low pathogenic forms only) detected by USDA-WS in the United States since 2006 have been found in dabbling ducks, the primary focus of future sampling will be on these species (Genus *Anas*, *Aix*, *Cairina*, and *Dendrocygna).*

Surveillance for HPAI-H5N1 will likely continue in Minnesota, and other parts of the U.S. next year. The USDA has banked all samples taken from 2006 to 2008, and is currently accepting proposals from state agencies and universities for further avian influenza research. Minnesota remains prepared to assist with future surveillance objectives if needed. In addition, the MNDNR has developed a surveillance and response plan for HPAI in wild birds, which includes increased vigilance of mortality and morbidity events within the state.

ACKNOWLEDGEMENTS

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

goals. Lastly, much of the hunter-harvested sampling was accomplished through assistance from Pat Reddig, University of Minnesota, and numerous students from both the Natural Resources program and the veterinary college.

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Table 1. Bird species sampled for highly pathogenic avian influenza H5N1 by Minnesota Department of Natural Resources and United States Department of Agriculture-Wildlife Services in 2008. Table includes live-bird, hunter-harvested, agency harvested, and mortality/morbidity. Fecal samples are excluded as they cannot be attributed to an individual species.

Species sampled	*n*
Ducks	
American Coot	11
American Green-Winged Teal	138
American Wigeon	78
American Blue-Winged Teal	153
Bufflehead	13
Canvasback	27
Common Goldeneye	112
Common Merganser	16
Gadwall	41
Greater Scaup	4
Hooded Merganser	4
Lesser Scaup	33
Mallard	305
Northern Pintail	58
Northern Shoveler	66
Red-Breasted Merganser	1
Redhead	21
Ring-Necked Duck	179
Wood Duck	74
Canada Geese	171
Other	
American White Pelican	16
American Woodcock	1
Double-Crested Cormorant	14
Lesser Snow Goose	1
Pied-Billed Grebe	1
Ruddy Duck	1
Ring-Billed Gull	6
Surf Scooter	1
Unidentified Duck	1
Total	1,547

Table 2. Results of avian influenza testing by the National Veterinary Services Laboratories (NVSL)[1] from samples submitted by Minnesota in 2008.

Species	Collection strategy	Test type[1]	Test result	Total
American Green-Winged Teal	Hunter-harvested	AI NVSL-AIV H5 RRT-PCR	H5	4
American Green-Winged Teal	Hunter-harvested	AI NVSL-Subtyping	H3N2	1
American Wigeon	Hunter-harvested	AI NVSL-AIV H5 RRT-PCR	H5	2
American Wigeon	Live Wild Bird	AI NVSL-AIV H5 RRT-PCR	H5	2
American Wigeon	Hunter-harvested	AI NVSL-Subtyping	H5N2	1
American Wigeon	Live Wild Bird	AI NVSL-Subtyping	H10N7	1
American Wigeon	Live Wild Bird	AI NVSL-Subtyping	H10N8	1
American Wigeon	Live Wild Bird	AI NVSL-Subtyping	H4N8	1
Bufflehead	Hunter-harvested	AI NVSL-AIV H5 RRT-PCR	H5	1
Blue-winged Teal	Hunter-harvested	AI NVSL-AIV H5 RRT-PCR	H5	4
Blue-winged Teal	Hunter-harvested	AI NVSL-Subtyping	H4N4, N8	1
Gadwall	Hunter-harvested	AI NVSL-AIV H5 RRT-PCR	H5	2
Lesser Scaup	Hunter-harvested	AI NVSL-AIV H5 RRT-PCR	H5	1
Mallard	Hunter-harvested	AI NVSL-AIV H5 RRT-PCR	H5	24
Mallard	Hunter-harvested	AI NVSL-Subtyping	H3N4, N8	1
Mallard	Hunter-harvested	AI NVSL-Subtyping	H4N8	1
Mallard	Hunter-harvested	AI NVSL-Subtyping	H4N9	1
Mallard	Hunter-harvested	AI NVSL-Subtyping	H5N2	4
Mallard	Hunter-harvested	AI NVSL-Subtyping	H6N1	1
Mallard	Live Wild Bird	AI NVSL-Subtyping	H10N7	1
Northern Pintail	Hunter-harvested	AI NVSL-AIV H5 RRT-PCR	H5	2
Northern Shoveler	Hunter-harvested	AI NVSL-AIV H5 RRT-PCR	H5	1
Northern Shoveler	Live Wild Bird	AI NVSL-Subtyping	H3N8	1

[1]Test results include AI NVSL Subtyping = identifies other strains of avian influenza that are not H5N1;
AI NVSL-AIV H5 RRT-PCR = test for the H5 avian influenza subtype only.

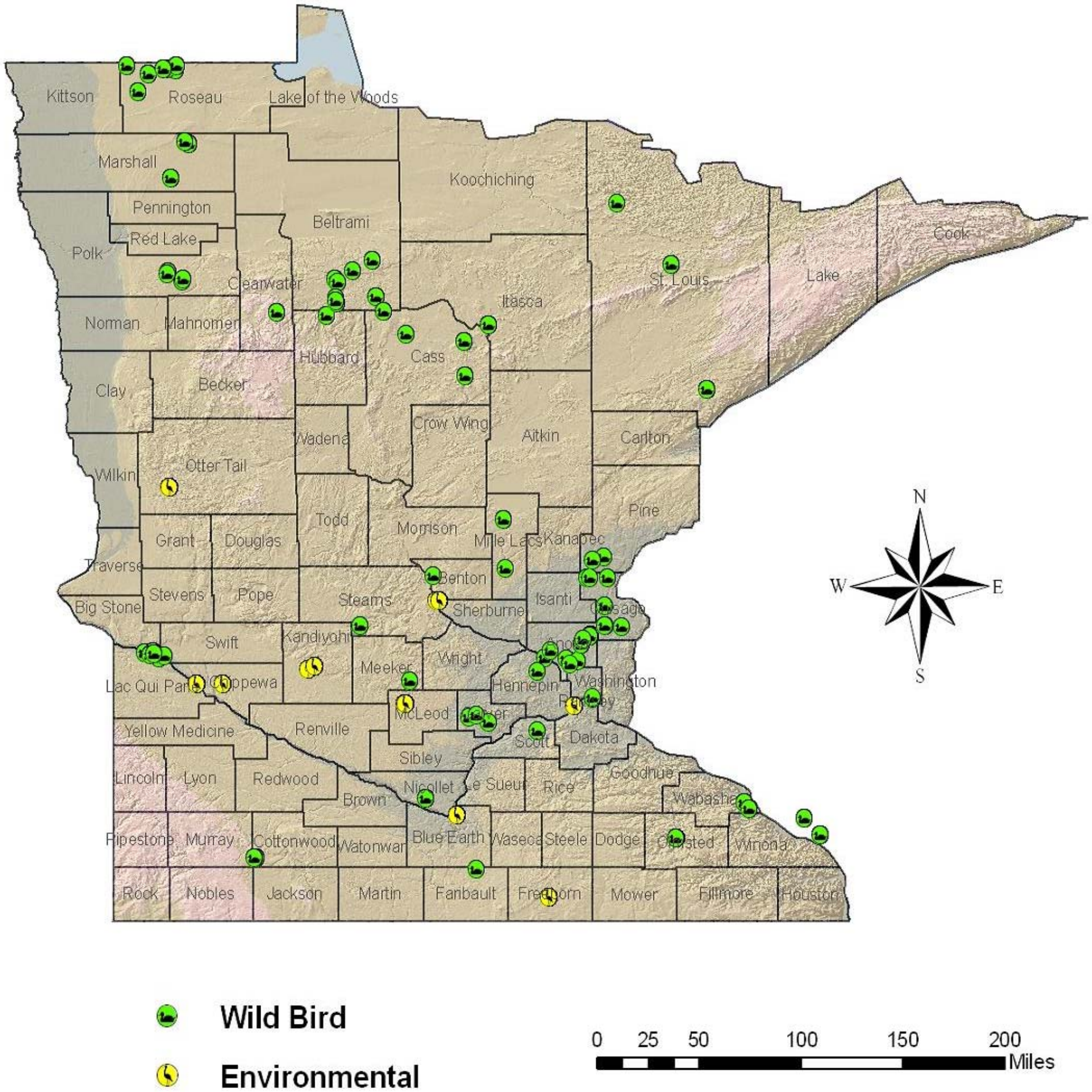

Figure 1. Collection sites from which live bird and environmental (fecal) samples (n=2,263) were tested for highly pathogenic avian influenza in Minnesota during 2008.

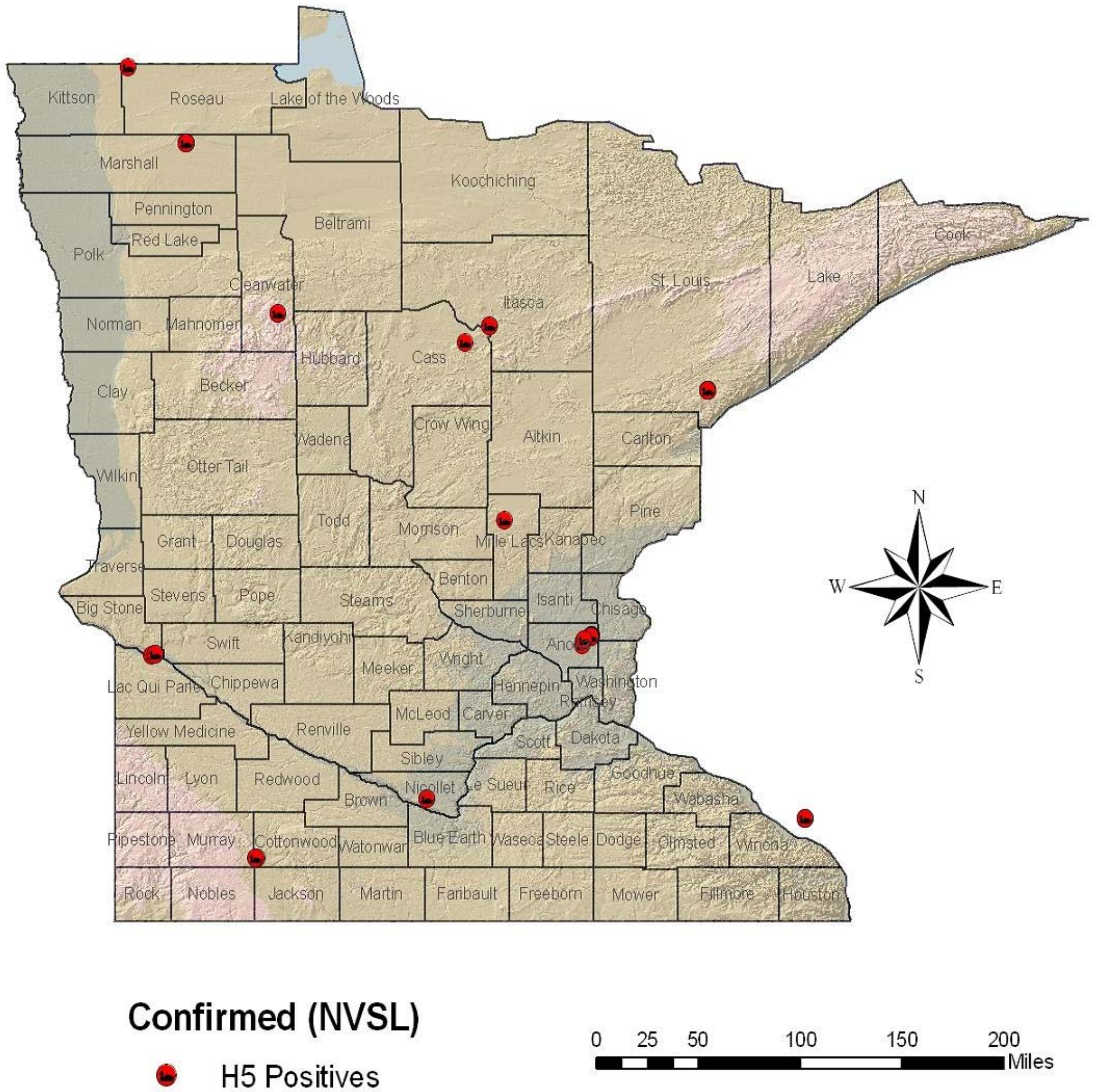

Figure 2. Collection sites where a low pathogenic H5 strain was detected (red dots) among the waterfowl (*n*=43) sampled in Minnesota during 2008.

MINNESOTA DEPARTMENT OF NATURAL RESOURCES CWD SURVEILLANCE PROGRAM 2008

Michelle Carstensen[1], Erika Butler, Michael DonCarlos, and Lou Cornicelli

SUMMARY OF FINDINGS

In 2008 and early 2009, the Minnesota Department of Natural Resources (MNDNR) sampled 1,440 hunter-harvested white-tailed deer (*Odocoileus virginianus*) for chronic wasting disease (CWD). The majority of these samples (66%) were collected in northwestern Minnesota, in conjunction with surveillance efforts for bovine tuberculosis; the remainder (34%) of samples were collected along the MN-WI border. All of the samples were negative for CWD. In addition, MNDNR submitted samples from 56 deer through targeted surveillance, which included sick animals, escaped captive cervids, and roadkills; these samples were also negative for the disease. MNDNR plans to conduct hunter-harvested surveillance in southeastern MN in fall 2009, in response to a recently detected CWD-positive captive elk facility in Olmsted county and the continued risk of disease spread from CWD-infected wild deer from Wisconsin.

INTRODUCTION

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

To date, CWD has been diagnosed in 3 captive elk herds and 1 captive white-tailed deer herd within the state of Minnesota. Two of the elk herds (Stearns and Aitkin counties) were discovered in 2002 and depopulated; no additional CWD positive animals were found. In spring 2006, a captive white-tailed deer was found infected with CWD from a mixed deer/elk herd in Lac Qui Parle county. That herd was also depopulated without additional infection being detected. However, over 40 additional premises within the state have been impacted as a result of trace-forwards and trace-backs conducted by the Minnesota Board of Animal Health (BAH). Currently, nearly all of these herds have undergone surveillance protocols. In all of these cases, the original source of the CWD has not been identified. In early 2009, a third captive elk herd (Olmsted county) was found infected with CWD. This most recent herd is expected to be depopulated once an indemnity agreement can be reached between the herd owner and the United States Department of Agriculture (USDA). MNDNR and BAH are working cooperatively to address the impact of CWD in these captive facilities, as well as management options to control its spread.

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Over the past 7 years, MNDNR has tested in excess of 30,000 deer across the state for CWD, all of which have been negative. Consequently, in recent years, sampling has been scaled back to address 3 main components:

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

METHODS

MNDNR continues to sample deer exhibiting clinical symptoms consistent with CWD (targeted surveillance). 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.

Hunter-harvested surveillance occurs at deer registration stations during the firearm hunting seasons. At the stations, hunters were asked to voluntarily submit retropharyngeal lymph node samples for CWD testing. Samples were submitted to the Veterinary Diagnostic Laboratory at the University of Minnesota for disease screening. Any presumptive positive samples were submitted to the National Veterinary Services Laboratories (Ames, IA) for official confirmation of the disease. Hunter information was recorded, including the hunter’s name, address, telephone number, MNDNR number, and location of kill. Maps were provided to assist the hunters in identifying the location (Township, Range, and Section) of the kill. Cooperating hunters were given a cooperator’s patch.

During fall 2008, MNDNR also collected approximately 1,250 lymph node samples from hunter-harvested deer in the northwest as part of a surveillance program for bovine tuberculosis. MNDNR had planned to screen approximately 1,000 of these samples for CWD as well. The registration stations that were selected to screen for both diseases include those in the northwest part of the state. Registration stations were also selected along the MN-WI border, as the disease exists in wild populations in WI, and screened for CWD only. The sampling goal along the WI-MN border was 500 samples.

RESULTS AND DISCUSSION

From January 2008 to May 2009, MNDNR collected a total of 56 samples from targeted surveillance efforts. This includes samples from 13 escaped captive cervids, 29 free-ranging sick deer, and 14 car-killed deer (collected within 10 miles of recent CWD-positive elk facility in Olmsted county). All samples were negative for CWD.

MNDNR collected a total of 1,440 samples from hunter-harvested deer for CWD screening during fall 2008. The vast majority of these samples (n=951) were collected in conjunction with bovine tuberculosis surveillance in the northwest, and the remaining samples (n=489) were collected along the MN-WI border (Figure 1). All samples were negative for CWD.

Since the agency has now collected in excess of 30,000 negative samples in statewide surveillance efforts, we feel that future resources for CWD surveillance, in addition to targeted surveillance, are better spent addressing changing risk factors. Specifically, it is important to monitor the CWD surveillance activities occurring in our bordering states, and conduct periodic surveillance in Minnesota in response to CWD status changes in these states. Additionally, periodic surveillance in the vicinity of previous cases of CWD in captive cervids in Minnesota may be prudent. Given the most recent case of a CWD-infected cervid farm in Olmsted county, MNDNR plans to conduct extensive surveillance during the fall 2009 firearm hunting season in the southeast portion of the state. Targeted surveillance of suspect deer is expected to continue throughout the state.

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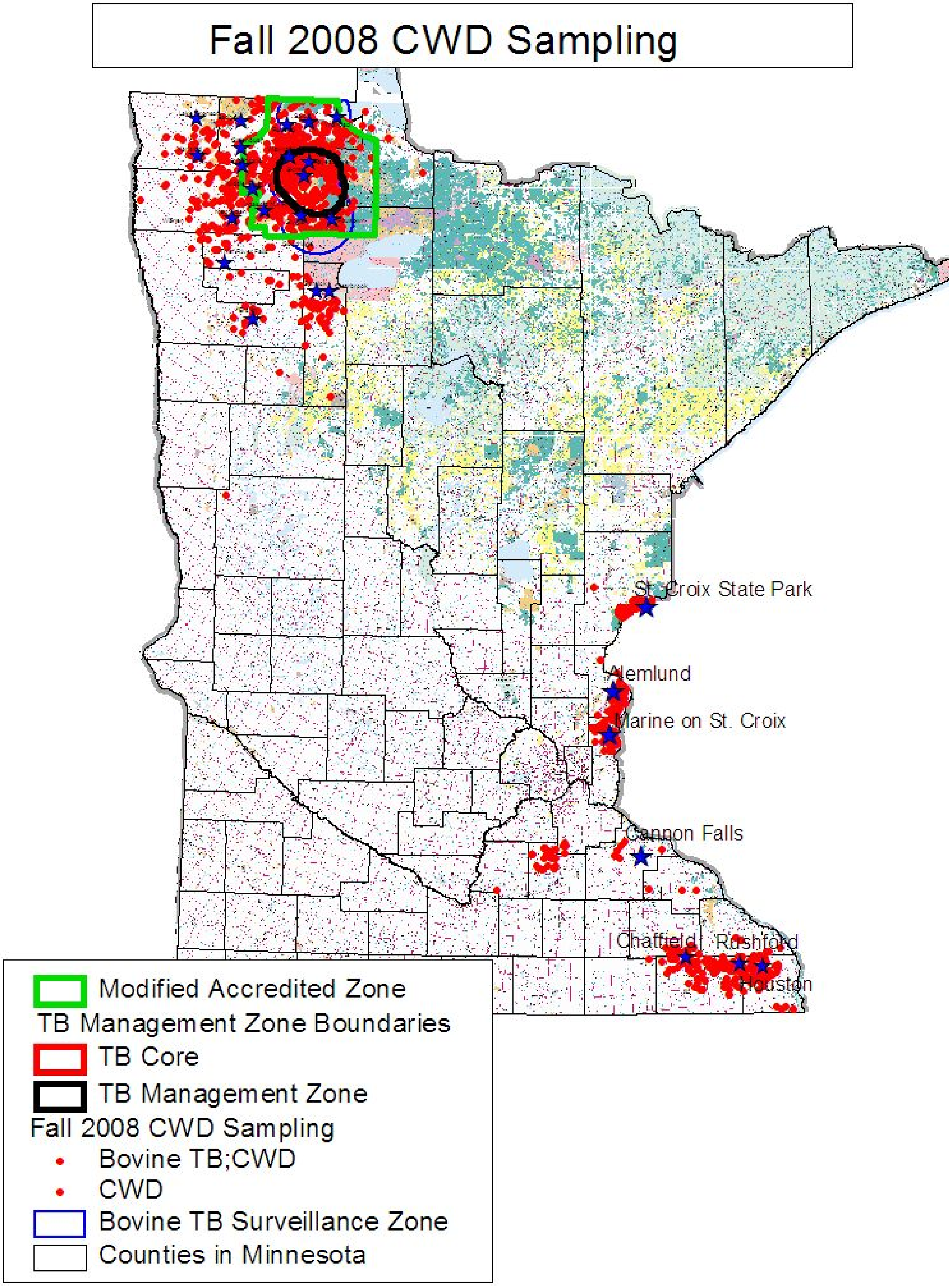

Figure 1. Sampling distribution for hunter-harvested deer tested for chronic wasting disease in Minnesota, fall 2008.

<u>Farmland Research Group Publications</u>

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