Summaries of Wildlife Research Findings
2006

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

SUMMARIES OF WILDLIFE RESEARCH FINDINGS 2006

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LOCAL RING-NECKED DUCK POST-FLEDGING MOVEMENT, SURVIVAL, AND REFUGE USE: A PILOT STUDY

David P. Rave and John R. Fieberg

SUMMARY OF FINDINGS

Breeding ring-necked duck (*Aythya collaris*) populations have been increasing continentally, but appear to be declining in Minnesota. We initiated a pilot study in August 2006 to investigate post-fledging movement, survival, and the use of established refuges by locally produced ring-necked ducks. Between August 14 and 19, 2006, we captured and implanted radio transmitters subcutaneously in 25 locally produced, hatch year (HY) ring-necked ducks. We followed birds from the ground for 2 weeks, and then from the air until we lost contact with the last birds on October 19. We also set up 4 remote receiving stations on established waterfowl refuges. All birds survived the first 2 weeks following surgery. Retention of radios was a problem in the pilot study with at least 10 of 25 birds shedding transmitters prior to the end of the study. A different transmitter attachment strategy will be required in future years. The remote receiving stations worked well and we will set up receiving stations at 14 refuges in future years of the study.

INTRODUCTION

Minnesota's 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 appear to be declining in Minnesota. Further, hunter harvest of ring-necked ducks has declined markedly in the last 20 years even as numbers of these birds staging in fall on most traditional ring-neck refuges have increased (Wetland Wildlife Populations and Research Group, unpublished data). Factors influencing resident populations are poorly understood, and efforts to better understand their status began in 2003 with the development of a Minnesota ring-necked duck breeding-pair survey. Minnesota's Fall Use Plan also 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 as is the influence that the distribution of the resident population might have on that of migrant ring-necks arriving in the fall.

In response to these information needs, a pilot study of post-fledging resident ring-neck ducks was initiated during the 2006 summer and fall field season. This study was used to develop and test methods of capturing and monitoring birds and to gain a preliminary understanding of the post-fledging movements and fall distribution of local ring-necked ducks. The ability to assess the influence of refuges on survival will largely depend our ability to mark and follow an adequate sample of ducks. Therefore, information from this first year of data was used to plan a more expanded study to be completed over the next 3 years.

The objective of this research is to gain an understanding of the influence of north central Minnesota refuges on the distribution and welfare of resident ring-necked ducks. Specifically, we will employ radio telemetry to: 1) characterize the post-fledging movements of local ring-necked ducks prior to their fall departure, particularly as a function of distance from natal marshes and distance from waterfowl refuges; 2) quantify use of refuges and relate refuge use to refuge level characteristics (size, number of birds on refuge, vegetation characteristics) as well as individual level covariates (gender, proximity of natal marsh to refuge); and 3) estimate survival of locally raised birds during this period, and relate the survival of locally raised birds to their relative use of established refuges.

Study Area

The proposed study area (Figures 1 and 2) encompasses a significant portion of the core of the ring-necked duck breeding range in Minnesota (Zicus et al. 2005) and includes all important ring-necked duck refuges in this part of Minnesota (Figure 3). Presently, banding locations for resident ring-necked ducks are concentrated in the NW portion of the area.

METHODS

We decided to use 2006 as a pilot year to test equipment and methodology. We elected to utilize subcutaneous radio transmitters for our pilot study because this type of radio had been used successfully on hatch year mallards (*Anas platyrhynchos*), canvasbacks (*Aythya valisineria*) (Korschgen et al. 1996a, 1996b), and common loons (*Gavia immer*) (Kenow et al. 2003). Subcutaneous transmitters require a surgical technique that is less invasive to the birds than transmitters implanted in the body cavity and can be done without the need to hire veterinary assistance (R. Gatti Wisconsin DNR, K. Kenow, US Geological Survey, Upper Midwest Environmental Sciences Center, and J. Berdeen, Minnesota DNR Wetland Wildlife Populations and Research Group, personal communication). Surgical techniques followed those of Korschgen et al. (1996a). Transmitters were equipped with mortality switches.

We captured hatch year ring-necked ducks using night-lighting techniques (Lindmeier and Jessen 1961). The following morning, birds were weighed (g), tarsus and culmen lengths measured (mm), and surgery performed in our lab. Birds were then held in a darkened room throughout the day and released in the evening at the lake from which they had been captured. Radio-marked birds were relocated from the ground for the first 2 weeks post marking. We then attempted to locate birds weekly using aerial surveys.

Survival was estimated using the generalized Kaplan Meier estimator (Kaplan and Meier 1958). Birds that were located during one search, but were not located in any further searches, were censored the day after the last location. Birds that died or dehisced their transmitter between 2 searches (i.e. transmitter went into mortality mode) were censored on the day closest to half the length of the period between the last location and the location when the transmitter had gone into mortality mode. Birds that were not located for >14 days, but were located again, were censored for the period between the two location events and were treated as a new bird in the population following the period of absence. Birds that were killed on a known day were assigned that day as the end of their survival period.

We erected 4 remote receiving stations on refuges within the study area. Stations were located on Drumbeater Lake, Fiske Blue Rocks Lakes, Gimmer Lake, and Preston Lakes Refuges (Figure 4). Stations consisted of a 6-meter mast with 1-3 yagi antennas, depending on the size and shape of the refuge. At Preston Lakes and Gimmer Lake Refuges, we used Advanced Telemetry Systems (ATS) R4000 scanning receivers that we had on hand, coupled with ATS DCC (Data Collection Computer) standard data loggers. These receivers continuously scanned through all radio frequencies we used and stored any frequencies detected on the refuge to the data logger. These stations were visited weekly throughout the study to download data from the data logger to a portable computer. At Drumbeater Lake and Fiske Blue Rocks Lakes, we used ATS R4500S receivers that had integrated data loggers and were equipped with DSP (Digital Signal Processor) technology. These new receivers were equipped with a cell phone download unit to test remote downloading from the data loggers. Twice weekly, we called these stations using a modem and downloaded data directly to an office computer without the need to visit the station. All receiver-data logging systems were powered with 12-volt marine batteries recharged daily with solar panels to minimize the need to periodically change batteries. Reference radio-transmitters were stationed permanently on each refuge to assure that receivers and data loggers were functioning properly.

RESULTS

In 2006 we focused our capture efforts on lakes that have traditionally been used to leg band ring-necked duck ducklings. We marked 25 class II and class III ducklings (Gollop, J. B. and W. H. Marshall. 1954. A guide to aging duck broods in the field. Unpublished report. Mississippi Flyway Council). Marking occurred from 14–19 August, and 1–2 ducklings were marked from each banding location. Each surgery took about 15 minutes. Mean mass of radio-marked birds was 574.8 g (range 515 – 660 g). Mean tarsus and culmen lengths were 42.9 and 44.3 mm respectively (see Appendix 1).

Radio-marked birds were relocated from the ground for the first 2 weeks post surgery, then were relocated weekly from the air starting on 9 September. All birds survived at least 10 days post surgery. Two birds were still on the study area and alive as of 19 October, but no birds were relocated after that date. Most smaller wetlands and refuges were frozen by 24 October and we did not fly after that date.

We used a 50-day survival period to look at survival in 2006. The survival rate for ring-necked ducks during the pilot year was 0.750 (95% CI 0.505 – 0.995) between 17 August and 5 October. Between 14 August and 19 October, 5 radio-marked birds are known to have died, however, 2 birds had left the study area and had been right censored prior to their deaths. Hunters shot all 5 birds. Two birds were also reported shot by hunters after they left Minnesota, 1 in Texas and 1 in Illinois.

The remote receiving stations operated well. We were able to download data from each, either by visiting the station or via cell phone technology. The receivers worked well and continuously recorded the presence of reference radio-transmitters. One radio-marked ring-neck used a refuge for several days. This was verified both by the remote receiving station and by aerial flights over the refuge.

We had problems during the pilot year that will need to be resolved in future years. Radio transmitters were incorrectly assembled by ATS, leading to very poor signal strength. We were unable to receive transmitter signals at distances >1 mile, even from the air, during the pilot study. This led to difficulty finding birds after they began to disperse from their natal marshes. Further, we are unsure whether birds may have used portions of refuges that were beyond the range of transmitters. ATS has assured us that this problem will be resolved, however, other options will be explored. We also had problems with transmitter retention. Over the course of the study, 10 of 25 birds shed their transmitters and were right censored. Mass of birds that lost transmitters averaged slightly less than birds that retained them (Fig. 5). Transmitter retention will be a major focus in future years of the study.

DISCUSSION

Treating the first year of our radio-telemetry study as a pilot year proved invaluable. We used subcutaneous transmitters because they had been used successfully on mallards, canvasbacks, and loons. However, the subcutaneous transmitters we used did not work well on HY ring-necked ducks, as retention rates were poor. This is likely because body size of HY ring-necks is small, and there was little room under the skin in these birds for the transmitter. We will try a different attachment technique for the transmitters in 2007. Subcutaneous transmitters with minnow seine material glued to the back have worked to greatly increase retention rates in eiders and shorebirds (D. Mulcahey, U.S. Geological Survey Alaska Science Center, personal communication). However, it is possible that we will still have retention problems this year, and may be forced to try yet another attachment technique such as abdominally implanted transmitters in future years of the project.

The direct recovery rate of radioed birds in 2006 (28 %) was higher than recent recovery rates of ring-necks banded in the same area (J. Berdeen, unpublished data). This high recovery rate may be the actual recovery rate of locally produced ring-necks, an anomaly, or a transmitter effect. We plan to put $100 reward bands on all radio-marked birds and on a sample of normally banded birds to determine whether the recovery rate is different between these groups.

During 2006, we deployed 4 remote receiving stations. Two of the stations used ATS R4000 receivers as well as ATS data loggers. We visited these stations weekly to download data. At the other 2 stations, we used ATS R4500 receivers, with built in data loggers and an attached cellular phone so data could be downloaded via modem directly to a computer. This system seemed to work flawlessly and data could be downloaded without the need to travel to the remote station locations. In 2007 we will erect remote stations at 14 waterfowl refuges (Table 1). We will use 4 R4000 receivers with data loggers, and 8 R4500 receivers with cell technology on state designated refuges within the study area. Further, in 2007–2009, remote receiving stations will be located at Rice Lake and Tamarac National Wildlife Refuges, as these refuges have agreed to be cooperators in this project.

In future years of the study, we will need to radio-mark birds from additional locations to better represent the birds residing within the study area. We will locate additional banding lakes throughout the study area in early-mid August 2006. We may be able to use the same ring-necked duck habitat models that we currently use for the Ring-necked Duck Breeding Pair Survey to help locate lakes throughout the study area for ring-neck capture. Finding and capturing birds from lakes throughout the study area will be imperative to meet study assumptions. Wetland conditions may also be a determinant as to whether we can capture an adequate sample of birds in 2007. Low wetland conditions during summer 2006 made capturing ring-necks on many wetlands difficult, and if drought conditions persist into summer 2007, capturing ring-necked ducklings may be even more difficult.

Management Implications

Post-fledging ecology of most waterfowl has received relatively little study, and refuge management has been identified as an important element for duck management in the fall (Minnesota Department of Natural Resources. 2001. Restoring Minnesota's Wetland and Waterfowl Hunting Heritage, Minnesota Department of Natural Resources, St. Paul, Minnesota, USA). This study will attempt to relate the distribution and welfare of a local population of ducks to the pattern of refuges existing in north central Minnesota. The study also will provide information for a resident waterfowl species that has received little attention and which appears to be declining. Understanding factors influencing the distribution of locally raised ring-necked ducks in the fall also might be a key to understanding the distribution of migrant ring-necks in the fall. This understanding may provide valuable insights regarding the distribution of refuges required to meet management objectives for local ring-necked ducks.

ACKNOWLEDGMENTS

M. Zicus was instrumental in all phases of the planning, development, and initiation of this project. J. Berdeen helped with planning, coordinating and conducting surgeries, coordinating nightlighting, hiring interns, and with data analysis. S. Cordts and J. Lawrence helped with surgeries, and Jeff helped collect equipment after the field season. A. Killian helped set up remote towers. J. Heineman flew telemetry flights, and A. Buchert flew us to Drumbeater Lake and helped set up the remote tower. Dr. A. Pillar and M. Kelly from the Bemidji Veterinary Hospital ordered surgical equipment and gave technical advice for the surgeries. S. Bischof, J. Holmes, B. Korman, A. Miller, A. Reed, and N. Walker, helped set up remote towers, capture, and radio track birds. John Finn, Leech Lake DNR, guided us into Drumbeater Lake. R. Lego allowed us access to the Leech River, and D. Barrett sterilized the transmitters for us.

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Table 1. Refuges to be included in the study and number of recording telemetry stations needed on each refuge.

Refuge	Location	~Peak numbers	Receivers
Donkey Lake	6 mi. SW Longville	350	1
Drumbeater Lake	2 mi. N of Federal Dam	160,000	1
Fiske and Blue Rock Lakes	8 mi. SE Northhome	40,000	1
Gimmer Lake	10 mi. SE Blackduck	200	1
Hatties and Jim Lakes	13 mi. SE Blackduck	0	1
Hole-in-the-Bog Lake	2 mi. SW Bena	4,000	1
Mud and Goose Lakes	4mi. SSW of Ballclub	2,100	1
Lower Pigeon Refuge	4 mi. S Squaw Lake	700	1
Pigeon River	6 mi. S Squaw Lake	700	1
Preston Lakes	22 mi. ENE of Bemidji	535	1
Rice Lake Waterfowl Refuge	8 mi. N Deer River	7,000	1
Rice Pond	9 mi. E of Turtle River	15	1
Tamarac National Wildlife Refuge	16 mi. NE Detroit Lakes	10,000	4
Rice Lake National Refuge	5 mi SSW of McGregor	120,000	4

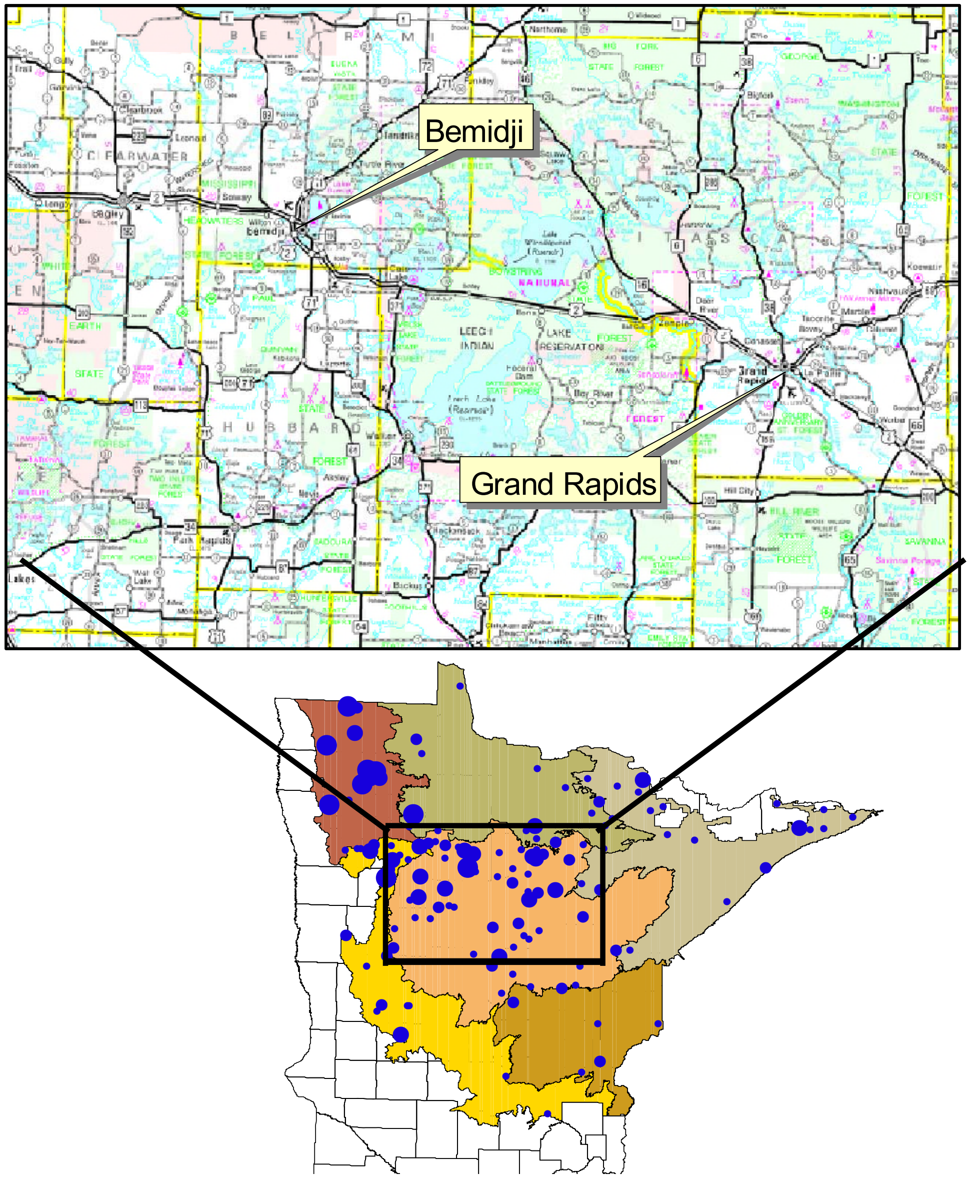


Figure 1. Proposed study area, 2006 – 2009. State map reflects results from 2004 – 2006 helicopter survey (see Figure 2 for details).

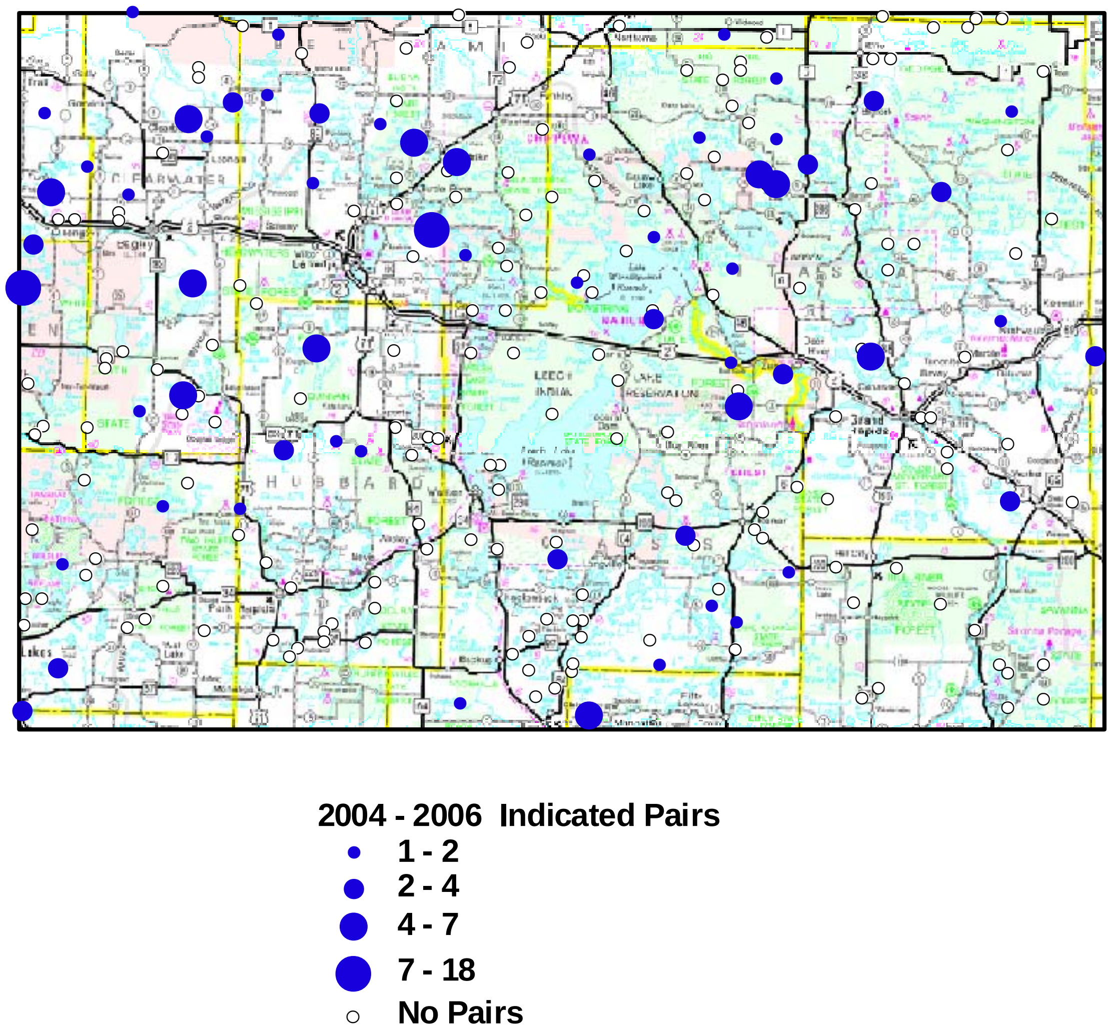
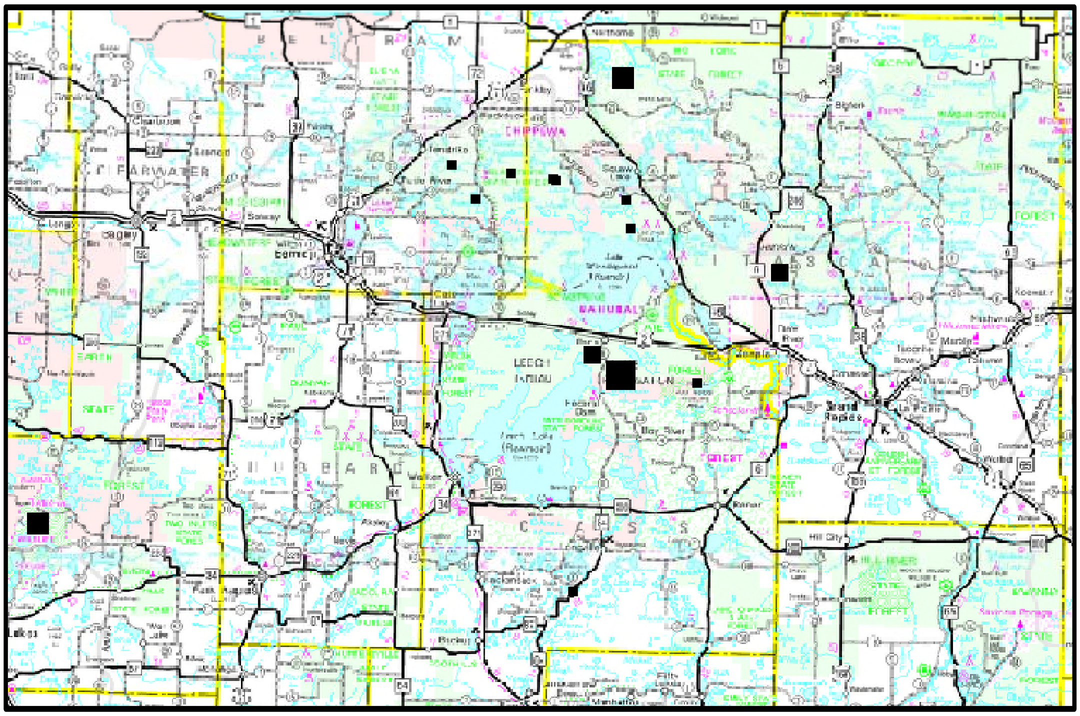


Figure 2. Distribution of indicated breeding pairs of ring-necked ducks based on survey plots in the 2004 – 2006 helicopter survey.



Peak numbers

- 0 - 2100
- 2101 - 7000
- 7001 - 40000
- 40001 - 160000

Figure 3. Approximate peak numbers of ring-necked ducks in fall on designated refuges, 2005.

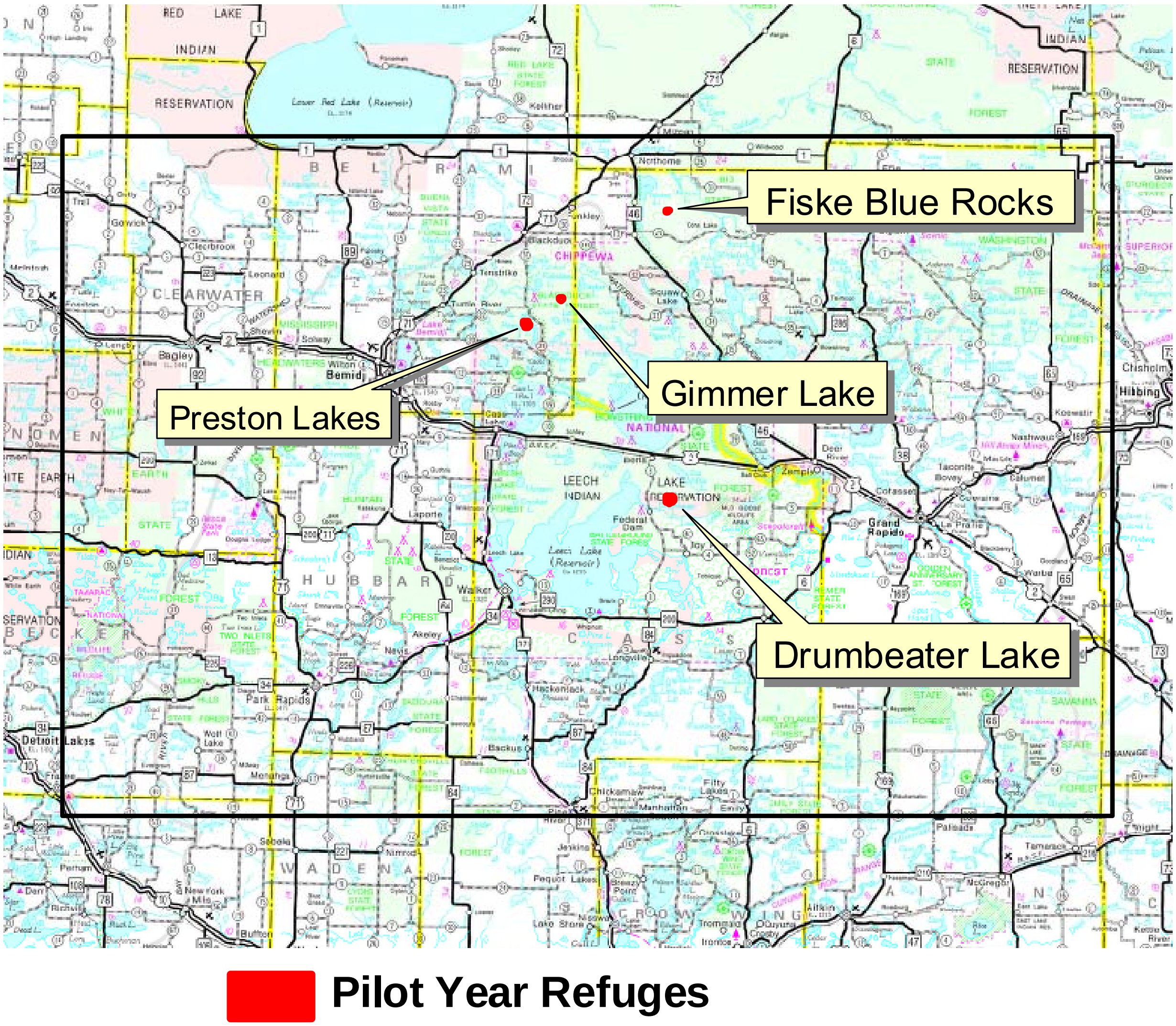


Figure 4. Refuges where remote receiving stations were located during the 2006 pilot year of the ring-necked duck telemetry study.

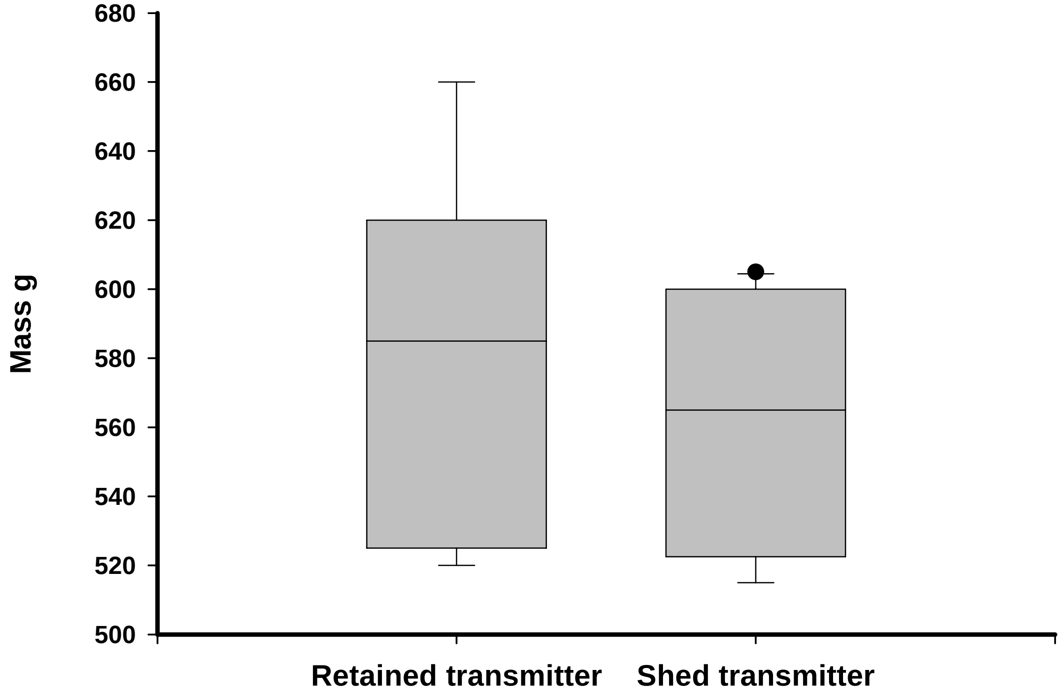


Figure 5. Box and whisker plot of the mass of ring-necked ducks at capture for those that retained their transmitters and those that shed them, August – October 2006.

Appendix 1. Data for hatch year ring-necked ducks captured between 14 and 19 August, 2006.

Year	Capt. Date	FWS Band Number	Mass (g)	Sex	Tarsus (mm)	Culmen (mm)	Capture Lake	Frequency	Term date	Term status
2006	8142006	104680754	600	F	44.4	46.8	East Four-legged	151.013	10022006	Dehisced
2006	8162006	108678227	650	M	43.2	44.1	White Oak	151.023	9302006	Killed
2006	8162006	104680790	565	M	43.4	40.2	Little Pine WMA	151.045	10092006	Not Found
2006	8182006	108679503	550	F	42.3	45	Little Moose	151.053	10092006	Dehisced
2006	8142006	104680744	565	M	38.4	43.1	W. Four-legged	151.065	9122006	Dehisced
2006	8192006	108679535	615	M	45.7	46	Little Puposky	151.075	9072006	Not Found
2006	8142006	104680751	585	M	41.2	42.2	E. Four-legged	151.084	9212006	Not Found
2006	8162006	104680788	620	M	40	45.6	Big Pine	151.104	10022006	Not Found
2006	8152006	108678220	520	F	42.6	42.2	Upper Rice	151.205	10092006	Not Found
2006	8182006	108679516	565	F	45.4	47.1	Rabideau	151.223	10022006	Dehisced
2006	8192006	108678234	605	M	45	46.8	Whitefish	151.245	10192006	Alive
2006	8142006	108678208	520	F	42.4	43.5	Muskrat	151.265	10192006	Alive
2006	8192006	108679533	660	M	44.9	46	Little Puposky	151.284	10092006	Not Found
2006	8162006	108678226	515	F	42.2	42.6	White Oak	151.324	10092006	Dehisced
2006	8162006	104680792	525	F	42.3	44.1	Little Pine WMA	151.344	10012006	Killed
2006	8182006	108679517	540	F	42.5	44.8	Rabideau	151.363	9212006	Dehisced
2006	8142006	104680743	585	M	41.3	45.9	W. Four-legged	151.383	9212006	Dehisced
2006	8152006	108678217	525	F	41.7	40.5	Upper Rice	151.402	9252006	Dehisced
2006	8162006	104680787	520	F	41.3	43.1	Big Pine	151.425	10022006	Not Found
2006	8152006	104680774	610	M	46.4	45.8	Little Pine	151.444	10192006	Not Found
2006	8152006	104680676	600	M	45.3	47.4	Dutchman	151.565	10182006	Dehisced
2006	8142006	108678210	660	F	43	45.6	Muskrat	151.584	10182006	Not Found
2006	8182006	108679504	550	F	41.9	44.6	Little Moose	151.603	9302006	Killed
2006	8162006	104680677	515	F	41.3	43.2	Damon	151.663	9262006	Dehisced
2006	8152006	104680775	605	F	43.2	41.5	Little Pine	151.685	9262006	Dehisced

MINNESOTA'S RING-NECKED DUCKS: A PILOT BREEDING PAIR SURVEY

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

SUMMARY OF FINDINGS

Little is known about the distribution and relative abundance of Minnesota's ring-necked duck (*Aythya collaris*) breeding population. We conducted the third year of a pilot survey to better understand the issues involved in monitoring these important but poorly studied ducks. The helicopter-based counts (06–16 June 2006) entailed 10 flight days and included a portion of Minnesota that is considered primary breeding range. Minnesota Department of Natural Resources' MN-GAP land cover data again were used to quantify presumed ring-necked duck nesting cover in Public Land Survey (PLS) section-sized survey plots, and 4 habitat classes were defined based on the amount of nesting cover in each plot. Similarly to 2005, we combined results from 2 separate surveys to estimate population size. We apportioned 200 plots among 12 strata (i.e., 6 Minnesota Department of Natural Resources' Ecological Classification System sections x 2 habitat classes) using a stratified random sampling design to estimate population size in the best habitat. We used a simple random sample of 50 plots to estimate population size in the remaining habitat. The combined population was estimated to be ~15,600 indicated breeding pairs (~31,000 birds). Numbers of ducks counted from the air and the ground on 14 lakes differed less in 2006 than in 2005, and the difference was likely due to less time elapsed between the air and ground surveys. The stratification we used continued to account for geographical- and habitat-based differences in ring-necked duck abundance, whereas, we would have needed approximately 1.2 times as many plots to achieve the same precision under a simple random sampling design.

INTRODUCTION

Staff in the Minnesota Department of Natural Resources (DNR) Wetland Wildlife Populations and Research Group have been developing a forest wetlands and waterfowl initiative. The status of ring-necked ducks has been among the topics considered because the species has been identified as an indicator species for the Forest Province (Minnesota Department of Natural Resources. 2003. A Vision for Wildlife and its Use – Goals and Outcomes 2003 – 2013 (draft). Minnesota Department of Natural Resources, unpublished report, St. Paul), but little is known about the current distribution and abundance of breeding ring-necked ducks in Minnesota.

In 2004, a pilot survey was conducted in a portion of Minnesota that is considered primary breeding range (Zicus et al. 2005). Minnesota Department of Natural Resources' MN-GAP land cover data were used to quantify presumed ring-necked duck nesting cover in PLS section-sized survey plots, and 4 habitat classes were defined based on the amount of nesting cover in each plot. Plots in 2 habitat classes were not sampled because few ring-neck pairs were believed to occupy these plots. The resulting population estimate (~9,000 indicated pairs) was almost certainly biased low because >69% of the survey area was not sampled, and some survey plots in the habitat classes, that were not surveyed, were misclassified.

Our objectives were to: 1) conduct the third year of a pilot study to determine the most appropriate sampling design and allocation for an operational breeding-pair survey of ring-necked ducks in Minnesota; and 2) make recommendations for future operational surveys.

METHODS

Two separate surveys were again conducted in 2006 to reduce the bias associated with the 2004 estimate. We apportioned 200 plots among 12 strata (i.e., 6 Minnesota Department of

Natural Resources' Ecological Classification System (ECS) sections x 2 habitat classes) using a stratified random sampling design to estimate population size in the best habitat. We used a simple random sample of 50 plots to estimate population size in the remaining habitat. We continued to use a stratified random sampling design with 2 stratification variables: ECS sections and presumed nesting-cover availability (i.e., a surrogate for predicted breeding ring-necked duck density) to estimate population size in the best ring-necked duck habitat. We used a 2-stage simple random sampling design to estimate population size in the remainder of the survey area. 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. We considered pairs, lone males, and males in flocks of 2–5 to indicate breeding pairs (IBP; J. Lawrence, Minnesota Department of Natural Resources, personal communication). The total breeding population in the survey area was considered to be twice the IBP plus the number of birds in mixed sex groups and lone or flocked females.

Statistical Population, Sampling Frame, and Sample Allocation

The surveys were restricted to an area believed to be primary breeding range of ring-necked ducks for logistical efficiency (Zicus et al. 2005). However, we again used habitat class definitions modified from those used for stratification in 2004 (Table 1). Based on 2004 results, we added MN-GAP Level 4 cover class 10 (lowlands deciduous shrub) as presumed nesting cover. Furthermore, we reduced the maximum distance that we believed ring-necked ducks were likely to be from a shoreline from 250 to 100 m. We also corrected a GIS processing error that we made in 2004. Habitat class 1 and 2 plots were presumed to represent the best habitat, whereas, habitat class 3 and 4 plots represented the remainder of the survey area. As in 2004 and 2005, 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. Finally, we determined from the 2004 and 2005 survey that breeding ring-necked ducks did not use large fish lakes, therefore, for the 2006 survey we removed all "nesting cover" associated with lakes having a General or Recreational Development shoreline classification.

A stratified sampling design was used to estimate breeding ducks in habitat class 1 and 2 plots, and the sampling frame consisted of 12 strata (i.e., 6 ECS sections x 2 habitat classes). We proportionally allocated 250 plots to the 12 strata using the same approach as in 2004 (Zicus et al. 2005). We used a 2-phase sampling process to sample plots in habitat classes 3 and 4. The phase-1 sample consisted of 1,000 habitat class 3 and 4 plots, disregarding ECS sections. These plots were visually inspected using 2003 Farm Services Agency (FSA) true color aerial photography and classified as to their ring-necked duck potential (i.e., possible breeding pairs vs. no pairs). PLS sections containing open water except for small streams were considered potential ring-necked duck plots. The proportion of plots classified as potentially having pairs was used as an estimate of the proportion of all class 3 and 4 plots that had potential for breeding pairs. We then randomly selected 50 plots (phase-2 sample) from those having the potential for ring-necked duck pairs in order to estimate the mean number of breeding pairs in these plots.

Data Analyses

Estimated Population Size. – We used SAS PROC SURVEYMEANS (SAS 1999) to estimate population totals for habitat class 1 and 2 plots in each ECS section and the entire survey area. In this analysis, PLS sections were the primary sampling unit in a stratified random sampling design. For the second survey, we estimated population size (τ) for habitat class 3 and 4 plots in the entire survey area as follows:

$$\hat{\tau} = \hat{P} * \bar{x} * N \,,$$

where $\hat{P}$ = proportion of phase-1 plots classified as habitat-class 3,
$\bar{x}$ = mean breeding ducks detected on phase-2 sample plots, and
N = total habitat-class 3 and 4 plots in sampling frame.

The variance of $\hat{\tau}$ was estimated using the delta method as:

$$\mathrm{var}(\hat{\tau}) = N^2 ((\hat{P}^2 * \mathrm{var}[\bar{x}]) + (\bar{x}^2 * \mathrm{var}(\hat{P})).$$

Estimates from the 2 surveys were combined to produce an overall population estimate for the survey area.

Aerial Visibility

An implicit assumption in aerial waterfowl surveys is that the proportion of the population of interest that is observed from the air is known or can be estimated (Smith 1995). Surveys using helicopters usually rely on the assumption that virtually all individuals are seen (Ross 1985, Cordts 2002). In fact, counts of ring-necked duck pairs in boreal wetlands that were made from helicopters were similar to those made when walking around wetlands or by traversing wetlands in a canoe (Ross 1985). We again examined this assumption by comparing aerial counts of indicated ring-necked duck pairs on the 14 lakes included in the Bemidji Area Ring-Necked Duck Pair Survey (Zicus et al. 2004) with pair counts from these lakes that were made from boats. Only 13 lakes were compared in 2006 because of weather and timing constraints.

Stratification Evaluation

We estimated the relative efficiency (RE) of the stratified sampling design by dividing the estimated variance for a simple random sample [var(SRS)] by the variance of the stratified random sample [var(StRS)] (Schaefer et al. 1996, Cochran 1997) where:

var(SRS) = estimated variance of $\bar{x}$ if we treated the observations as having been drawn using a simple random sample (i.e., based on a weighted sum of sample variances in each stratum), and

var(StRS) = estimated variance of the stratified mean.

If stratification performed well, it would account for differences in indicated ring-necked duck pairs seen on plots among the strata in the survey. As a result, the population variance would be smaller than that obtained by a comparable simple random sample (Cochran 1997). If each estimator is unbiased, then RE will describe the relative gain in precision by using ECS and habitat classes as stratification variables. We also evaluated the stratification by comparing the mean number of indicated pairs seen among ECS sections, habitat classes, and the interaction between ECS sections and habitat classes using SAS Proc GLM (SAS 1999).

Data Acquisition

The 2006 survey utilized an ArcView 3.x extension (DNRSurvey) in conjunction with a Global Positioning System receiver and DNR Garmin program (real time survey technique) to collect the survey data. This approach allowed us to display the aircraft's flight path over a background of aerial photography and the survey plots. The flight path and ring-necked duck observations were recorded directly to ArcView shapefiles, all in real time (R. Wright, Minnesota Department of Natural Resources, personal communication).

RESULTS

More PLS sections in the northeast were classified as habitat classes 1 and 2 in 2005 and 2006 versus 2004 because we included MN-GAP cover class 10 as potential nesting cover. As a result, survey plots were distributed somewhat more to the northeastern portion of the survey area than they were in 2004 (Figure 1). Most plots (77) were located in the Northern Minnesota Drift and Lake Plains section. The fewest plots (8) were located in the Lake Agassiz, Aspen Parklands section this year, similar to 2005, rather than the Northern Superior Uplands section as in 2004 (Table 2). The highest and lowest sampling rate again occurred in the Lake Agassiz, Aspen Parklands section and Northern Superior Uplands section, respectively. The survey was conducted 06–16 June and entailed 10 survey-crew days. Observed pairs represented 44% of the indicated pairs tallied during the survey compared to 36% in 2005 and 57% in 2004 (Table 3).

Estimated Pair Density

Mean pair density on habitat class 1 and 2 plots ranged from a high of 4.16 pairs/plot in the Lake Agassiz, Aspen Parklands section to a low of 0.30 pairs/plot in the Western and Southern Superior Uplands section (Table 4). Mean pair densities were higher in all of the 6 ECS sections compared to 2005. Considering both years, pair densities were greatest in the Lake Agassiz, Aspen Parklands section with lowest pair densities in the Western and Southern Superior Uplands and the Northern Minnesota and Ontario Peatlands sections.

Estimated Population Size

Estimated indicated breeding pairs on habitat class 1 and 2 plots ranged from a high of 6,334 in the Northern Minnesota Drift and Lake Plains section to a low of 669 in the Western and Southern Superior Uplands section (Table 5). More breeding pairs were estimated in 2006 in all 6 ECS sections than in 2005. Pair numbers were greatest in the Northern Minnesota Drift and Lake Plains section and fewest in the Western and Southern Superior Uplands section.

The estimated population of ring-necked ducks on habitat class 1 and 2 plots ranged from a high of 14,816 in the Northern Minnesota Drift and Lake Plains section to a low of 1,338 in the Western and Southern Superior Uplands section (Table 6). As with indicated breeding pairs, more ducks were estimated in 2006 in all 6 ECS sections than in 2005. Considering both years, the most birds occurred in the Northern Minnesota Drift and Lake Plains section and the fewest in the Western and Southern Superior Uplands section.

In 2006, we estimated indicated breeding pairs and total birds for the entire survey area (Table 7). The estimated number of indicated breeding pairs for the survey area was 15,631 (90% confidence interval = 11,221 – 20,042), and the estimated ring-necked duck population was 34,342 (90% confidence interval = 24,766 – 43,918).

Observed Distribution

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. Indicated pair observations in 2005 and 2006 shifted somewhat to the east compared to 2004 (Figure 1). Estimates from 2004–2006 suggest that some ECS subsections or portions of a section might have substantial numbers of breeding ring-necked ducks even though few birds were observed in the ECS section (Figure 2). For example, pairs/plot and total estimated pairs were relatively high in the Northern Superior Uplands, yet few plots in the section had indicated breeding pairs (Table 5 and 6).

Aerial Visibility

Counts from boats generally agreed with aerial counts of IBPs on the individual lakes included in the 14-lake survey (Figure 3). Boat counts in 2004 were conducted 14–18 June in 2004 with the aerial survey of the 14 lakes done on 17 June. In contrast, boat counts were conducted 15–21 June with the aerial survey done on 24 June in 2005. In 2006, boat counts were conducted 8–13 June with the aerial survey flown on 12 June. Poorer agreement between the 2 surveys in 2005 than in either 2004 or 2006 was likely due to the greater time that elapsed between the boat counts and aerial surveys.

Stratification Evaluation

Analysis of variance indicated that the strata identified using the MN-GAP models were reasonable. For the most part, IBPs were related significantly to ECS sections and to habitat classes within the ECS sections (Table 8). Results from 2004 might have been an exception as IBPs were related to an interaction between ECS section and habitat class. Stratification by ECS section resulted in a thorough distribution of sample plots throughout the survey area (Fig 1). However, lack of IBP observations in some strata suggested we might have over-stratified relative to the number of plots we surveyed. Estimated relative efficiency suggested that a modest increase in plots would have been needed to achieve the same precision under a simple random design as we did using a stratified design. However, estimated relative efficiency should be interpreted cautiously because we lacked variance estimates for 1 and 3 strata in 2004 and 2005, respectively.

Data Acquisition

Generally less time was required to survey a plot in 2006 than in 2005 or 2004 (Table 9). Survey time ranged from 1–13 minutes (mean=4.5) compared to 1–22 minutes (mean=5.2) in 2005 and 1–29 minutes (mean=7.2) in 2004 (Figure 4). Use of the real time survey technique accounted for the reduction in plot survey time in 2005 (Fieberg et al. 2006), and it reduced the total airtime required to survey the plots by >8 hours.

DISCUSSION

We further improved our understanding of the issues involved in designing and conducting a survey to estimate the abundance and describe the distribution of breeding ring-necked ducks in Minnesota. Survey dates in 2004–2006 appeared appropriate because 36–57% of the indicated pairs were counted as paired birds, and survey timing is considered optimal when most birds are counted as pairs and not in flocks (Smith 1995). The stratified random sampling design that we employed was adequate for plots in habitat classes 1 and 2, while the second survey based on a simple random sample of plots in habitat classes 3 and 4 again provided an estimate for the survey area that was unbiased (i.e., included all potential breeding habitat). Detection rates appeared to be relatively high in all habitats, suggesting that any bias probably would be minor.

MN-GAP land cover data provided a convenient way to stratify the survey area, but they have shortcomings as well as strong points. They provided a consistent statewide source of land use/cover data that was available in an easy to use, raster format. However, the data are derived from 1991 and 1992 satellite imagery, which makes them dated. Further, the data exist at 4 levels of resolution, and classification accuracy of cover types is diminished at the level that we used. Nearly 50% (487 of 1,000) of habitat class 3 and 4 plots were incorrectly classified when compared to conditions that existed in 2003 (based on FSA photography).

Misclassifications resulted from MN-GAP data missing small wetland areas capable of supporting ring-necked duck pairs or from wetland conditions that had changed between 1991 and 2003. We improved the stratification in 2006 by eliminating emergent shoreline-vegetation associated with larger lakes containing fish from our definition of potential ring-necked duck nesting cover. Ring-necked ducks do not occupy these types of lakes during the breeding season.

The stratification approach that we used worked relatively well and assured a reasonable geographical distribution of survey plots throughout the survey area. However, failure to observe birds in 3 strata in 2005 indicated that we might have over-stratified given the sample size of 230 habitat class 1 and 2 plots. As a result, our variance estimates were biased low because the estimated sample variance in some strata was 0 and these strata contributed nothing to the overall variance. Likewise, the design effect (i.e., RE) becomes difficult to estimate when some strata have no observations; therefore, our estimate of relative efficiency should be viewed cautiously.

Survey costs are an important consideration with any wildlife survey, and survey efficiency is the product of optimal plot size as well as appropriate stratification and efficient data acquisition. A complete examination of plot size efficiency will require consideration of the time required to fly to and among plots in the sample as well as the number of refueling stops required. We began modeling to evaluate various plot sizes after the 2006 field season.

Recommendations

- Conduct the 2007 survey using the same proportional allocation of 200 habitat class 1 and 2 plots among the 6 ECS sections. Conduct a second survey choosing a simple random sample of 50 habitat class 3 and 4 plots. <u>Rationale</u>: An operational survey might need to focus on a core area within the primary ring-necked duck breeding range to reduce costs and improve the precision of the estimate. The 2005 and 2006 data contained a better geographical distribution of plots than 2004, and have helped define a core area for indicated breeding pairs. Another year with a similar sample distribution will continue to define the core area for breeding ring-necked ducks in Minnesota.
- Begin the survey as soon after 5 June as possible. <u>Rationale</u>: A set starting date will assure the needed flight time can be scheduled. Although phenology will vary from year to year, this date should result in the survey being done while most ring-necked ducks are still paired.
- Pending further discussions within the DNR Wetland Group and the Waterfowl Committee, conduct future operational surveys in enough of the primary breeding range to provide the desired population information in the most cost-effective manner. <u>Rationale</u>: Obtaining population estimates for the entire primary breeding range would be ideal. However, the information gained by surveying some areas that are logistically difficult to reach or that have few ring-necked ducks might not be worth the added cost.
- Continue using PLS sections as sampling units unless future modeling indicates some other unit is more efficient. <u>Rationale</u>: Preliminary modeling in 2004 suggested that quarter-sections might be a more efficient plot size. However, this modeling did not account for the time required to fly to and among plots in the sample as well as the number of refueling stops required. Consequently, we have no basis for recommending a different size plot at this time.

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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 – 2006.

	Definition[a]		%[b]		
Habitat class	2004	2005 and 2006[c]	2004	2005	2006
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
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
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
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

[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]Percent of the survey area.
[c]Habitat class definitions in 2005 and 2006 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.

Table 2. Sampling rates by Ecological Classification System section for Minnesota's ring-necked duck breeding- pair survey, June 2004 – 2006

		~Area[a]			Sampling rate (%)		
Ecological Classification System sections	Habitat Classes	2004	2005	2006	2004	2005	2006
W & S Superior Uplands[b]	1	1,638	2,461	2,218	1.1	0.9	0.9
Northern Superior Uplands	2	1,810	4,648	4,209	0.7	0.8	0.8
N Minnesota & Ontario Peatlands	3	1,817	2,737	2,389	1.4	1.3	1.3
N Minnesota Drift & Lake Plains	4	5,048	8,383	7,145	1.5	1.1	1.1
Minnesota & NE Iowa Morainal	5	3,510	4,033	3,561	1.4	0.9	0.9
Lake Agassiz, Aspen Parklands	6	316	363	340	4.7	2.2	2.4

[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. Social status of the indicated pairs observed in the Minnesota ring-necked duck breeding pair survey area, June 2004-2006.

				Indicated pairs			
Year	Habitat class	No. of plots	Total ducks	n	% pairs	% Lone males	% Flocked males
2004[a]	1,2	200	278	160	57.5	18.1	24.4
2005[b]	1,2	230	147	92	35.9	28.2	35.9
2005	3,4	21	11	7	57.1	0.0	42.9
2006[c]	1,2	200	279	167	43.7	27.6	28.7
2006	3,4	50	4	3	33.3	66.7	0.00

[a]Survey conducted 6 – 17 June.
[b]Survey conducted 12 – 24 June.
[c]Survey conducted 6 – 16 June.

Table 4. Estimated indicated breeding pairs per plot in the habitat class 1 and 2 strata in the Minnesota ring-necked duck breeding pair survey area, June 2005-2006.

	2005			2006		
Ecological Classification System sections	Plots	Mean pairs/plot	SE	Plots	Mean pairs/plot	SE
W & S Superior Uplands[a]	22	0.181	0.179[b]	20	0.302	0.178
Northern Superior Uplands	36	0.252	0.118	33	0.636	0.215
N Minnesota & Ontario Peatlands	35	0.087	0.045[b]	30	0.658	0.228
N Minnesota Drift & Lake Plains	94	0.416	0.138	77	0.887	0.279
Minnesota & NE Iowa Morainal	35	0.228	0.010	32	0.590	0.318
Lake Agassiz, Aspen Parklands	8	3.403	1.365[b]	8	4.160	1.463

[a]Western and Southern Superior Uplands sections combined due to the small area of the Southern Superior Uplands occurring in the survey area.
[b]Standard error estimate is biased low because no birds were observed in one of the Ecological Classification System section's strata.

Table 5. Estimated indicated breeding pairs in the habitat class 1 and 2 strata in the Minnesota ring-necked duck breeding pair survey area, June 2005-2006.

	2005				2006			
Ecological Classification System section	Pairs	LCL[a]	UCL[a]	CV(%)	Pairs	LCL	UCL	CV(%)
W & S Superior Uplands[b]	444	0	1,207	99.5[c]	669	0	1,355	59.1
Northern Superior Uplands	1,169	244	2,095	46.8	2,679	1,148	4,210	33.7
N Minnesota & Ontario Peatlands	239	20	457	54.1[c]	1,572	644	2,499	34.7
N Minnesota Drift & Lake Plains	3,490	1,577	5,404	33.0	6,334	3,011	9,657	31.5
Minnesota & NE Iowa Morainal	918	241	1,595	43.6	2,102	178	4,026	53.9
Lake Agassiz, Aspen Parklands	1,235	273	2,198	40.1[c]	1,414	448	2,381	35.2

[a]Estimates were based on a stratified random sample of Public Land Survey (PLS) sections in habitat classes 1 and 2 and 6 ECS sections. LCL = lower 90% confidence level. UCL = upper 90% confidence level.
[b]Western and Southern Superior Uplands sections combined due to the small area of the Southern Superior Uplands occurring in the survey area.
[c]Variance estimate for the Ecological Classification System section is biased low because no birds were observed in one of the section's strata. As a result, the confidence interval is too narrow and the CV is optimistic.

Table 6. Estimated ring-necked ducks in the habitat class 1 and 2 strata in the Minnesota ring-necked duck breeding pair survey area, June 2005-2006.

	2005				2006			
Ecological Classification System section	Birds	LCL[a]	UCL[a]	CV(%)	Birds	LCL	UCL	CV(%)
W & S Superior Uplands[b]	889	0	2,415	99.5[c]	1,338	0	2,710	59.1
Northern Superior Uplands	2,339	488	4,190	46.8	5,357	2,295	8,419	33.7
N Minnesota & Ontario Peatlands	477	40	915	54.1[c]	4,076	1,141	7,012	42.3
N Minnesota Drift & Lake Plains	6,981	3,154	10,808	33.0	14,816	7,504	22,127	29.6
Minnesota & NE Iowa Morainal	4,122	187	8,057	56.4	4,204	375	8,052	53.9
Lake Agassiz, Aspen Parklands	2,471	545	4,396	40.1[c]	2,829	896	4,762	35.2

[a]Estimates were based on a stratified random sample of Public Land Survey (PLS) sections in habitat classes 1 and 2 and 6 ECS sections. LCL = lower 90% confidence level. UCL = upper 90% confidence level.
[b]Western and Southern Superior Uplands sections combined due to the small area of the Southern Superior Uplands occurring in the survey area.
[c]Variance estimate for the ECS section is biased low because no birds were observed in one of the ECS section's strata. As a result, the confidence interval is too narrow and the CV is optimistic.

Table 7. Estimated indicated breeding pairs and breeding population size in the Minnesota ring-necked duck breeding pair survey area, 2004-2006.

		Indicated breeding pairs				Breeding population			
Year	Habitat classes	Pairs	LCL[a]	UCL[a]	CV(%)	Birds	LCL[a]	UCL[a]	CV(%)
2004	1,2[b]	9,443	6,667	12,220	17.8[d]	20,321	14,248	26,395	18.1[d]
2005	1,2[b]	7,496	5,022	9,971	20.0[d]	17,279	11,156	23,402	21.5[d]
2005	3,4[c]	3,832	0	9,269	86.3	7,664	0	18,539	86.3
2005	All	11,328	5,359	17,298	32.0[d]	24,943	12,476	37,411	30.4[d]
2006	1,2[b]	14,770	10,465	19,075	17.6[d]	32,621	23,231	42,010	17.4[d]
2006	3,4[c]	861	0	1,908	74.0	1,721	0	3,816	74.0
2006	All	15,631	11,221	20,041	17.2[d]	34,342	24,766	43,918	17.0[d]

[a]LCL = lower 90% confidence level. UCL = upper 90% confidence level.
[b]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 ECS sections).
[c]Population estimates were based on a simple random sample of Public Land Survey (PLS) sections in habitat classes 3 and 4.
[d]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 8. General linear model evaluation of the Minnesota ring-necked duck breeding pair survey stratification developed using 2004 – 2006 MNGAP habitat models and the estimated relative efficiency of the resulting stratified random design.

MNGAP model	Stratification variable	df	F	P	RE[a]
2004	ECS section	5, 188	2.17	0.059	1.02
	Habitat class	1, 188	9.08	0.003	
	ECS section* habitat class	5, 188	0.93	0.462	
2005	ECS section	5, 218	7.17	<0.001	1.17
	Habitat class	1, 218	28.70	<0.001	
	ECS section* habitat class	5, 218	7.94	<0.001	
2006	ECS section	5, 188	3.51	0.005	1.06
	Habitat class	1, 188	7.25	0.008	
	ECS section* habitat class	5, 188	1.03	0.403	

[a]Relative efficiency of stratified random design compared to a simple random sample.

Table 9. Time required to complete the Minnesota ring-necked duck breeding pair survey, June 2004-2006.

			Time (min)[a]			
Year	No. of plots	Flight days	Operation[b]	Survey[c]	Min/plot	% Survey time
2004	200	13	4,686	1,441	7.2	30.8
2005	251	10	4,868	1,307	5.2	26.8
2006	250	10	4,399	1,126	4.5	25.6

[a]Includes all observers.
[b]Time between the initial start of the helicopter each morning and final shutdown of the helicopter each afternoon.
[c]Air time spent surveying the individual plots.

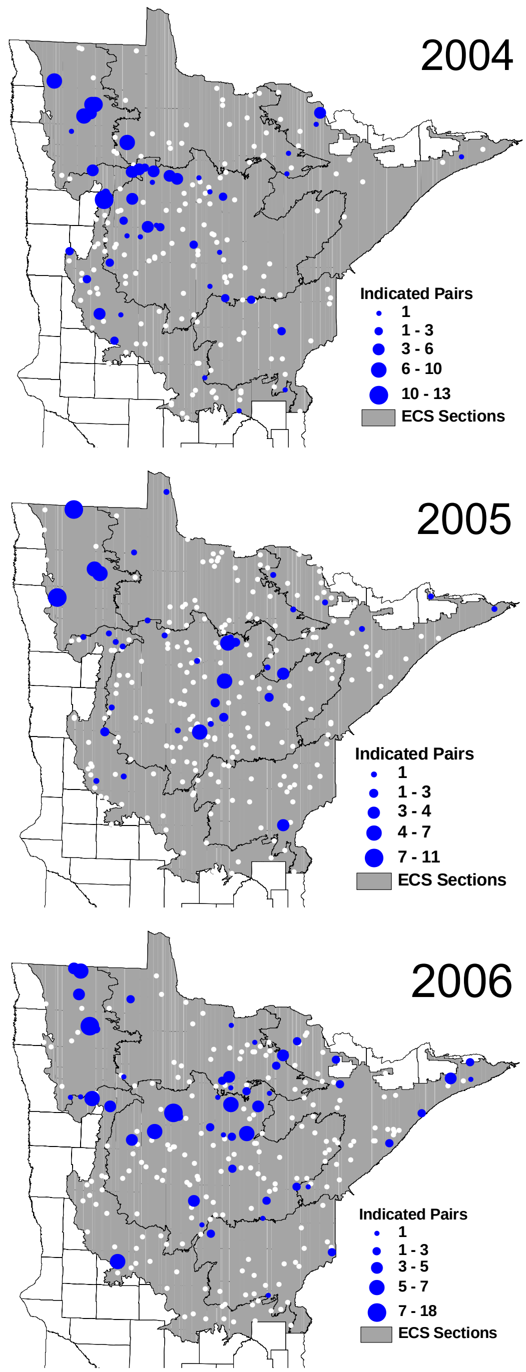

Figure 1. Plot locations and numbers of indicated breeding pairs of ring-necked ducks observed on survey plots in the Minnesota survey area in June 2004 (top), 2005 (middle), and 2006 (bottom). White circles indicate plots where no indicated pairs were seen.

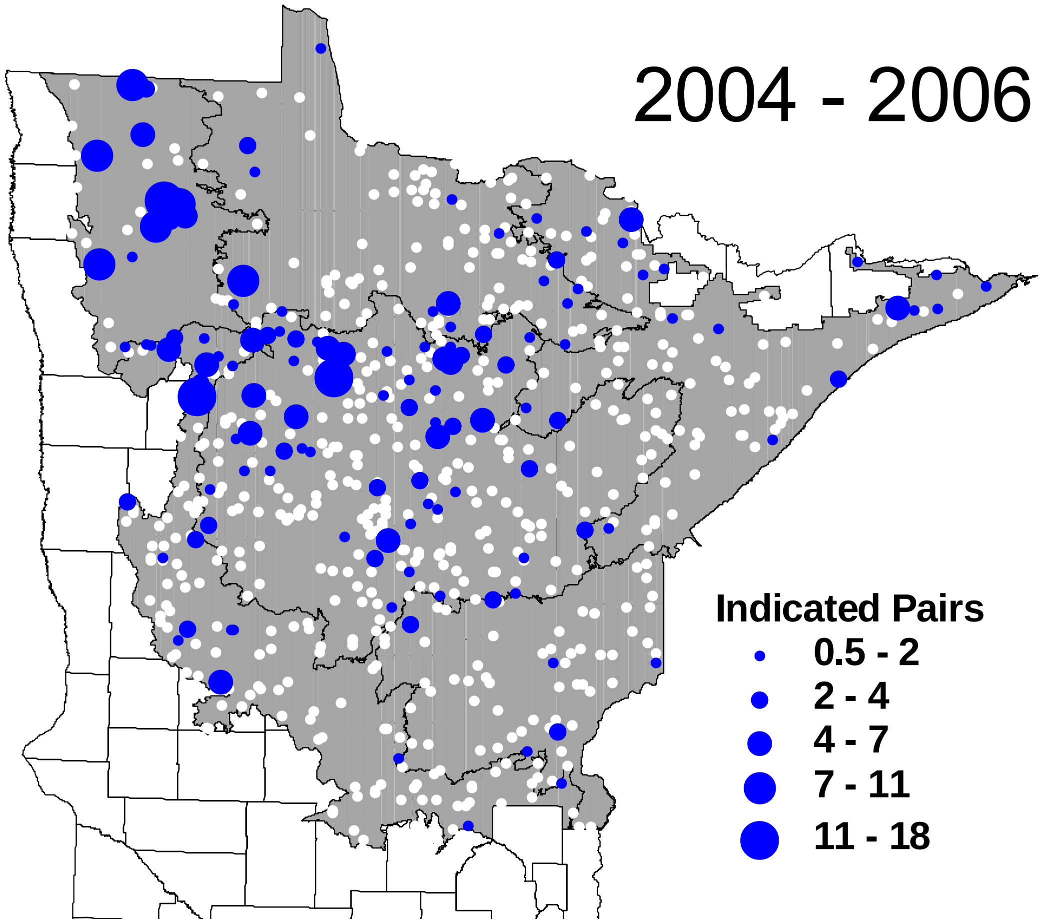

Figure 2. Plot locations and numbers of indicated breeding pairs of ring-necked ducks observed on survey plots in the Minnesota survey area, June 2004-2006. White dot indicates a plot where no birds were seen.

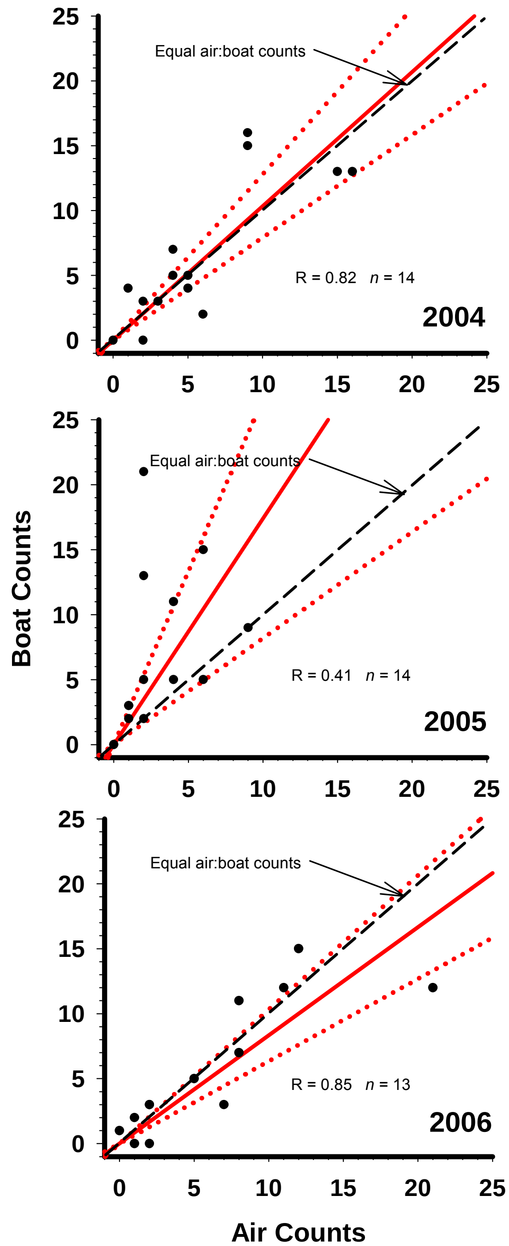

Figure 3. Regression lines and 95% confidence intervals comparing the numbers of indicated ring-necked duck breeding pairs counted from a boat and from the air on 14 lakes comprising the *Bemidji Area Ring-necked Duck Survey*, June 2004 (top), 2005 (middle), and 2006 (bottom).

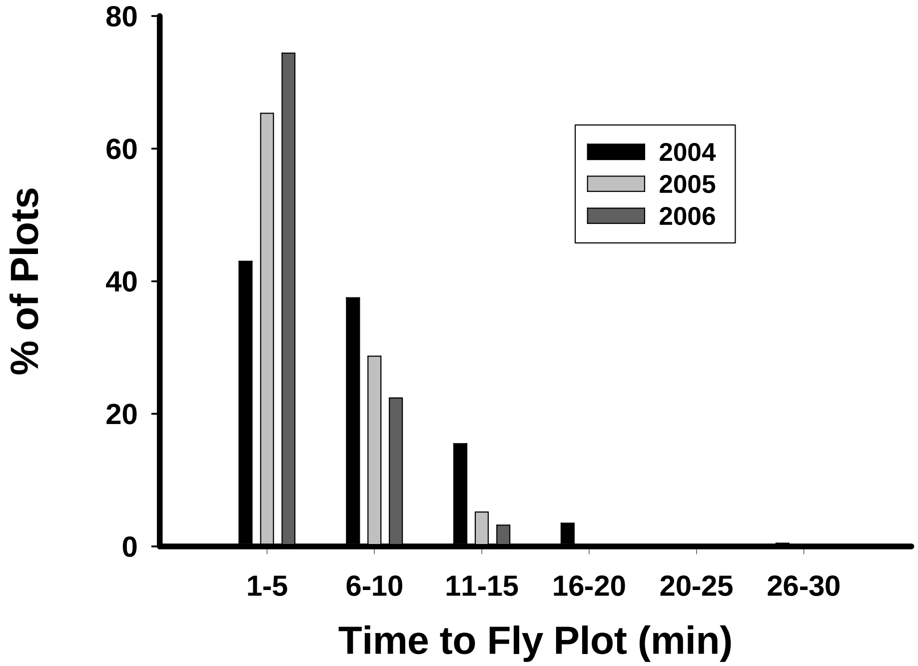

Figure 4. Time required (all observers) to survey individual ring-necked duck breeding pair plots in the Minnesota survey, June 2004 – 2006.

EVALUATING FUNCTIONAL LINKAGES AMONG LANDSCAPES AND WETLAND ATTRIBUTES: ASSESSING THE ROLES OF GEOMORPHIC SETTING, LAND USE, AND FISH ON WETLAND COMMUNITY CHARACTERISTICS

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SUMMARY

During 2005-06, we assessed fish community patterns and influences of site- and landscape-level variables on fish communities and ecological features of prairie wetlands in two areas in western Minnesota (generally Polk and Grant County areas). Fish populations were found to occur in nearly all wetlands. Diverse, multi-species fish communities were common, and often contained combinations of planktivorous, benthivorous, and piscivorous species. Preliminary analyses indicated that landscape-scale variables were poor predictors of fish populations in study wetlands. However, fish communities did reflect wetland size and site-level influences of piscivores. Here, we summarize procedures used in development and analyses of spatial (landscape) and site-level wetland data, and discuss preliminary trends in major variables including fish communities, aquatic invertebrates, limnological characteristics, submerged macrophytes, waterfowl use (breeding pairs and broods), amphibians, and periphyton. Future publications will more thoroughly describe relationships among these variables and landscape characteristics at spatially-explicit scales, and will clarify site-level influences of fish on wetland invertebrates, submerged macrophytes, and characteristics of clear – vs. turbid – water states in shallow Minnesota lakes.

INTRODUCTION

Installation of drainage tile and ditches, consolidation of wetlands, and other anthropogenic activities (e.g., agricultural land uses, road construction, nonnative invasive flora and fauna, intentional fish stocking, water control structures) are widespread in prairie regions of Minnesota. It is plausible that these landscape modifications have increased ecological influences of wetland fishes (reviewed by Bouffard and Hanson 1997), favoring preponderance of turbid, phytoplankton-dominated wetlands with low abundances of invertebrates and submerged aquatic vegetation. Furthermore, a prolonged period of above-average precipitation in Minnesota has increased depth of many prairie wetlands, increased surface connectivity among wetlands, and favored lower frequency of winter anoxia. These interacting influences contribute to development of permanent populations of fathead minnows (*Pimephales promelas)* and other fish species in a large proportion of wetlands remaining in Minnesota's prairie region (Hanson et al. 2005).

In shallow lakes and wetlands, reductions in herbivorous zooplankton due to predation by planktivorous fish are thought to reduce water transparency, favoring shifts towards increased turbidity and loss of submerged vegetation (Scheffer et al. 1993; Scheffer 1998). Across western and southern Minnesota, landscape modifications, along with resulting changes in fish distribution and population persistence, may have favored shifts toward a large proportion of degraded prairie marshes. Presently, many such sites are characterized by high turbidity, sparse communities of submerged aquatic plants and invertebrates, and limited suitability for waterfowl.

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Aquatic food web characteristics reflect density and community structure of associated fish populations. Fish-mediated influences on invertebrate community structure and water transparency are often pronounced (Bendell and McNicol 1987; Wellborn et al. 1996; Zimmer et al. 2000, 2001). Scheffer proposed that shallow-water ecosystems exist in 1 of 2 alternative conditions, either a clear-water, macrophyte-dominated state, or a turbid-water, phytoplankton-dominated state (Scheffer et al. 1993). Recent studies in Minnesota's Prairie Pothole Region (PPR) documented the strong negative influences of fathead minnows on invertebrate populations (Zimmer et al. 2000, 2001, 2002). Reductions in herbivorous zooplankton resulting from fish predation have been shown to increase phytoplankton biomass and turbidity consistent with predictions of models by Scheffer et al. (1993) and Scheffer (1998). Minnesota PPR wetlands largely conform to a binomial distribution (clear or turbid), rather than a normal distribution of features along a theoretical continuum (Zimmer et al. 2001; Herwig et al. 2004).

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

Fish community composition in lakes reflects interplay of isolation and extinction, but Magnuson et al. (1998) suggested that extinction is a more important influence. Extinction factors generally include environmental features of lakes such as surface area, habitat heterogeneity, depth and depth-related factors, winter oxygen concentrations, pH, presence of piscivores, watershed position, and local water chemistry (Tonn and Magnuson 1982; Rahel 1984; Marshall and Ryan 1987; Robinson and Tonn 1989; Keller and Crisman 1990, Magnuson et al. 1998). Isolation features are also important in structuring fish assemblages in lakes, and reflect differences in geomorphic setting (Magnuson et al. 1998; Hershey et al. 1999). Landscape features important in structuring fish assemblages in arctic lakes are primarily attributes of the lake/stream drainage network (Hershey et al. 1999). Alternatively, in small northern lakes, a combination of factors including presence of connecting streams, barriers, and characteristics of nearby species source pools have been identified as important predictors of fish community characteristics (Magnuson et al. 1998).

Fish community composition has been successfully predicted from only a few landscape and environmental variables, likely indicating that structuring mechanisms are robust (Tonn and Magnuson 1982, Rahel 1984, Robinson and Tonn 1989; Magnuson et al. 1998; Hershey et al. 1999). Fish assemblages in lakes also reflect regional and geographic patterns of fish distributions (i.e., reflecting local species pools) when larger spatial scales are considered (lakes - Jackson and Harvey 1989; wetlands - Snodgrass et al. 1996). In isolated wetlands in the southeastern US, disturbance frequency (drying) and connectivity determined the presence or absence of fish (Snodgrass et al. 1996). In contrast to prairie wetlands where low winter oxygen concentrations and sometimes drought influence fish distributions (Peterka 1989), drying and colonization rates were more important in determining the distribution of fish in coastal-plain wetlands (Snodgrass et al. 1996). Along the eastern part of the PPR (e.g., Minnesota), there is a propensity for intermittent surface water connections, and frequent fish invasions, as a result of natural east-west gradients in precipitation and topography (Leibowitz and Vining 2003; Hanson et al. 2005). In Minnesota's PPR wetlands, isolation (or connectivity) and extinction (environmental) characteristics are likely both important in structuring fish assemblages, but relative magnitude of influences are unknown.

Hershey et al. (1999) suggested a "geomorphic trophic" model to illustrate how stream drainage networks influenced the dispersal and subsequent distribution of native fishes in arctic lakes. There, fish controlled lake trophic structure (invertebrates, prey fish, etc.), but influences of fish also reflected extinction and isolation of fish populations due to constraints of landscape features. Thus, landscape configuration indirectly controlled trophic structure and expression of specific biological attributes within these lakes (Hershey et al. 1999). This model forms the basis of our overall working hypothesis of landscape control of PPR wetland food webs, where:

Geomorphic setting → Fish distributions → Ecosystem characteristics

We hypothesize that landscape and environmental features constrain fish communities, and interactively regulate the distribution of wetland fish throughout PPR regions of Minnesota. Fish, in turn, influence ecological characteristics of semi permanent and permanent wetlands in Minnesota's prairie landscape. Hence, by extension, landscape setting indirectly influences wetland ecosystem characteristics.

Landscape setting, including site-level wetland characteristics, may also directly influence water body features. For example, watershed position of a lake or wetland determines a variety of physical, chemical, and biological attributes of lakes (Kratz et al. 1997; Riera et al. 2000) and wetlands (Euliss et al. 2004). These properties include potential responses to drought, predominant groundwater interactions, and concentrations of dissolved constituents including organic carbon in lakes (Kratz et al. 1997). Other landscape features that have been found to influence water quality in lakes include percentage wetland extent in the watershed (Detenbeck et al. 1993; Prepas et al. 2001) and land use, where agricultural land was associated with a higher trophic state index (Detenbeck et al. 1993). Site-level wetland characteristics and processes that may also influence community characteristics include nutrient status (Scheffer et al. 1993; Bayley and Prather 2003; Jackson 2003), lake surface area (Hobæk et al. 2002; Wellborn et al. 1996), wetland depth (Scheffer et al. 1993), and macrophytes (Scheffer et. al. 1993; Paukert and Willis 2003; Zimmer et al. 2003).

The goal of our study was to develop conceptual and empirical models linking landscape features, site-level environmental influences, and wetland fish assemblages, and to assess the influences of these factors on characteristics of semi-permanent and permanent prairie wetlands. Our overall working hypothesis was that landscape setting indirectly influences wetland characteristics through structuring influences on fish communities.

METHODS

Study Area, Site Selection, Development of Landscape Predictor Variables

Our proposed study areas ("study landscapes") were selected to reflect a range of human-induced modifications. This gradient of anthropogenic influence results largely from a north to southwest transition toward increasing agricultural land use within Minnesota's PPR. Thus, our study focuses on 2 landscapes, 1 high-impact (HI) and 1 low-impact (LO) landscape. The HI landscape is located primarily in the southern portions of Grant County, and extends into northern Stevens County and western Douglas County, and includes 1 site in Ottertail County (Figure 1). The LO landscape is located primarily in eastern half of Polk County, with 1 study site located in northern Mahnomen County (Figure 1). In addition to differences in extent of human influence between our study landscapes, the HI and LO landscapes also fall into different ECS classifications due to variation in geomorphic features, climate, and vegetation patterns (Almendinger et al. 2000). Our study landscapes are also positioned in different major river drainages. The HI landscape lies between the Red River and Minnesota River drainages, while the LO landscape is located entirely within the Red River drainage. The LO and HI landscapes encompass approximately 1,292 km^2 and 1,435 km^2 respectively.

Within each study landscape, wetlands were selected for measurement of fish assemblages, wetland characteristics, and surrounding landscape attributes. For each landscape study area, we identified all candidate Type IV and V wetlands using the Minnesota Department of Natural Resource's (MN DNR) National Wetlands Inventory (NWI) Quick Theme layer. From this layer, we selected all "MnWet 4s and 5s", which best match the Circular 39 Types IV and V (Shaw and Fredine 1956; Stewart and Kantrud 1971). We imposed additional requirements such that all resulting candidate wetlands were between 2-40.5 ha, and were not licensed for aquaculture activities. Determining aquaculture status was accomplished by referencing the population of candidate study sites against the Division of Waters (DOW) numbers corresponding to basins licensed for aquaculture (either white sucker or walleye [*Sander vitreus]*) (Roy Johannes, MN DNR Fisheries, Aquaculture Program Coordinator). The remaining population of study sites within each study area was then stratified among 27 different bins based on the following criteria: 1) wetland size; 2) distance to nearest permanent stream, wetland, or lake; and 3) proportion agriculture within a 500 m buffer surrounding the wetland. We then randomly selected 1 study site from each of the resulting categories (for a total of 27 sites), and 9 (LO landscape) or 10 (HI landscape) additional sites across the 27 categories, with a maximum of 2 study sites per category imposed. If we were unable to obtain permission for a wetland in private ownership, or if some other conflict was identified (e.g., inaccessible), we then randomly selected a new site within that category, and repeated this process until a suitable study site was identified. In 2006, we selected 2 additional sites that were known to be fishless from previous studies within the HI landscape to facilitate statistical comparisons between fish and fishless sites.

A total of 36 study sites were selected for study in the LO landscape (35 sites in Polk County, 1 site in Mahnomen County) and a total of 39 study sites were selected for study in the HI landscape (31 sites in Grant County, 4 sites in Stevens County, 3 sites in Douglas County, 1 site in Ottertail County). In the LO landscape, 22 of the 36 study sites are either partially (6 sites), or completely (16 sites) within public ownership (i.e., Waterfowl Production Area (WPA), Wildlife Management Area (WMA), or National Wildlife Refuge (NWR)), with the remaining 14 study sites completely within private ownership. In the HI landscape, 21 of the 39 study sites are either partially or completely within public ownership (i.e., WPA, WMA, or county-owned), with the remaining 18 study sites completely within private ownership.

Existing GIS layers will be used to derive metrics that characterize features of the landscape surrounding each study wetland. Data layers not currently available will be developed as needed. Landscape attribute summaries might include, but are not limited to, the following: 1) distance to permanent and ephemeral water bodies; 2) distance to roads and driveways; 3) distance to (or presence of) drainage ditches and culverts; 4) latitude, elevation, and position of the study sites within the watershed; 5) surrounding land use assessed at multiple spatial scales; and 6) watershed ratios: direct contributing area (DCA) and total watershed areas (TWA) to the wetland surface area will be calculated. Watersheds for the DCAs and TWAs for each wetland will be manually delineated in Arc View using standard heads-up digitizing techniques. The on screen digitizing environment will incorporate hydrologically-corrected digital elevation models (DEM), digital raster graphics (DRGs), and digital orthoquads (DOQs), digitized hydrological connections and directional flow captured from DOQs, DEMs and/or corroborated from field inspections, as well as evidence from several other data sources (e.g., NWI, DNR streams and rivers coverage with proper connectivity and directionality, DOW Protected Wetlands Inventory (PWI) lake coverage, Department of Transportation (DOT) culvert point coverage, and existing digital major and minor watershed coverages). From this information, we plan to extract watershed areas at several spatially-explicit scales, summarize land cover types within watersheds, develop variables that capture influence of hydrological connectivity and geomorphic setting, and calculate average watershed slope. Within the DCA of each wetland, surrounding land cover types and connectivity features (streams, ditches) will be captured and categorized as outlined in Table 1. Our primary references for delineating land cover features were 2003 FSA Color DOQs, and 1 square mile land use maps obtained from county Farm Service Agency (FSA) offices. We will apply existing GAP data layers (or 2000 land cover data layers furnished by the USFWS, HAPET Office; Fergus Falls, MN), and existing flow network layers (MN DNR-Division of Waters) to characterize cover types and hydrological features within the watershed areas extending beyond the DCAs. We will use ArcView to summarize land cover types at various distances from the study basin, up to and including the DCA, and TWA scales. Existing aerial photographs (2003 FSA Color DOQs) and Global Positioning System (GPS) mapping were used to develop updated estimates of wetland size in 2005. Maximum depth of the study wetlands was also determined during the 2005 field season by measuring depths along parallel transects throughout the open water zone of each wetland.

Fish Community Assessments

Fish species composition was determined from July surveys using a combination of gear deployed overnight. Three mini-fyke nets (6.5 mm bar mesh with 4 hoops, 1 throat, 7.62 m lead, and a 0.69 m X 0.99 m rectangular frame opening into the trap) were set overnight in the littoral zone of each wetland. One experimental gill net (76.2 m multifilament net with 19, 25, 32, 38, and 51-mm bar meshes) was set along the deepest depth contour available in wetlands less than 2 m deep or along a 2 m contour in wetlands with sufficient depth. Preliminary results from the LO landscape indicated that results from minnow traps were redundant with the other types of gear, so fish sampling was restricted to 3 mini-fyke nets and 1 gill net per wetland in the HI landscape.

The protocol outlined above has been shown to be effective in sampling fish assemblages in small lakes from other regions (Tonn and Magnuson 1982; Rahel 1984; Jackson and Harvey 1989; Robinson and Tonn 1989), and enabled us to capture fish of different sizes, species, and from all major trophic guilds (e.g., planktivores, benthivores, piscivores) in the study wetlands. Number of individuals and total biomass of each species collected were determined for each type of gear in each site.

Aquatic Invertebrates

Zooplankton were sampled twice each year, once in late May or early June, and again in late July or early August by collecting 2 replicate vertical column samples (Swanson 1978a) at 6 locations in each wetland. Resulting data were used to estimate density of major invertebrate groups and taxon richness of these communities. Relative abundance of free-swimming macroinvertebrates was estimated using submerged activity traps (ATs) (Swanson 1978b; Murkin et al. 1983; Ross and Murkin 1989) placed in each wetland for 24-hours. Six ATs were deployed at the interface of open water and emergent macrophytes because this area often concentrates organisms. Estimates of relative abundance and taxon richness were developed for each study site. We collected aquatic invertebrate samples from 73 wetlands during 2005 and from 75 sites during 2006.

Limnological and Phytoplankton Sampling

Surface (dip) water samples were taken from the center of each wetland once during late May or early June, and again in late July or early August each year. Samples were acidified to a pH of 2 using concentrated sulfuric acid, then frozen. Samples were analyzed by the Minnesota Department of Agriculture chemistry lab (St. Paul, MN) for total Kjeldahl, nitrate-nitrite, and ammonia nitrogen, as well as total phosphorus. Additional water was collected at the same time as the surface samples for total dissolved phosphorus (TDP) and phytoplankton abundance, measured as chlorophyll *a* (Chla). TDP samples were collected by filtering lake water through GF/F glass fiber filters (0.7 µm nominal pore size) and immediately freezing filtered water. Chla samples were collected in the field by filtering lake water through a GF/F glass fiber filter. The filters were then wrapped in tin foil and immediately frozen. In the lab TDP was determined using high-temperature persulfate digestion followed by ascorbic-acid colorimetry (APHA 1989). Chla was measured in the lab using a 24 h, alkaline-acetone extraction, followed by fluorometric analysis (APHA 1989). Turbidity and specific conductance were measured in the field with a portable nephelometer and a conductivity meter.

Submerged Macrophyte Surveys

Species richness, frequency of occurrence, and community-scale biomass of submerged macrophytes were assessed using techniques of Jessen and Lound (1962), and Deppe and Lathrop (1992). In each wetland, submerged macrophytes were sampled in early August at 20 stations along 4 transects. Two throws were made at each station using a weighted plant rake, and frequency of occurrence was recorded for each plant species. We then calculated a whole-wetland score (number of times each species was collected in 40 rake throws; resulting species scores were then summed across all submerged plant taxa); values hereafter referred to as “submerged plant score”. We also measured the total plant biomass (all species combined) for the first rake throw at each station. Metaphyton (e.g. *Cladophora* spp.) and macroalgae (e.g. *Chara* spp.) were assessed along with vascular plant species during these surveys.

Waterfowl Surveys

Waterfowl numbers were assessed during the breeding season and brood-rearing period using helicopter survey techniques (Cordts 2002). Indicated breeding pairs (lone male, pairs, and flocked males < 6) were tallied by species on each wetland during early May 2005 and 2006. Groups by species were also recorded. We assumed that all individuals were seen using the helicopter survey technique (Ross 1985, Cordts 2002). We also counted the number of waterfowl broods on each wetland during late-June or July 2005 and 2006. Broods were recorded by species where possible and number of ducklings was estimated.

Amphibian Surveys

We sampled larval amphibians concurrently with fish (using the same gear used to sample fish, described above). In each wetland, we determined the total number of larval frogs, larval salamanders, and painted turtles captured with the 3 trap nets and 1 gill net set in each wetland during late July. Results are expressed as the total number of individuals captured in each study site.

Periphyton Measurements

Periphyton biomass (Chla) was determined by deploying artificial substrates for 5 weeks in 2005 and 4 weeks in 2006 (4 weeks was found to be sufficient to get maximum growth without sloughing of periphyton). Sampling devices were set out in mid-June and collected in late-July each year. These devices were constructed out of a polyester braided rope (6.35 mm thick, 1.5 m long) with a brick anchor attached to one end and a float on the other. Along the rope, 3 vinyl microscope slides were attached using zip-ties at 10, 50, and 90 cm below the surface because we hoped to assess whether periphyton abundance differed with depth. Total height of the sampling device was 1.5 m when placed vertically in the water column. Three devices were deployed in each wetland, near locations where invertebrates were sampled. Each sample was carefully removed from the water column to limit disturbance to the periphyton, placed in a container of well water and stored in a dark cooler until taken to the lab and processed within 12 hours. Chla was measured in the lab using a 24 h, alkaline-acetone extraction, followed by fluorometric analysis (APHA 1989).

RESULTS

Fish Communities

Sites within the HI landscape generally contained fewer fish species on average than sites within the LO landscape during both 2005 (4.00 vs. 5.61 species) and 2006 (3.82 vs. 5.57 species; Figure 2). The HI landscape had 19 sites and 21 sites with 3 or fewer fish species in 2005 and 2006, respectively, while the LO landscape had just 6 sites with 3 or less species of fish in both years. Maximum number of fish species sampled in sites within the HI landscape was 10, while the LO landscape had 3 sites with 11 or more fish species in 2005. In 2006, the maximum number of species in the HI landscape increased to 11, but dropped to 9 in the LO landscape. Twenty-seven different fish species were sampled across both the HI and LO landscapes. We sampled 23 species of fish across the HI landscape sites, and 21 species of fish across the LO landscape sites in 2005. In 2006, we sampled 2 fewer species in both the HI (21 species) and LO (19 species) landscape sites.

Study areas (HI and LO) shared 14 species of fish across years, including black bullhead, black crappie (*Pomoxis nigromaculatus)*, bluegill (*Lepomis macrochirus)*, brown bullhead (*Ameiurus nebulosos)*, brook stickleback (*Culaea inconstans)*, central mudminnow (*Umbra limi)*, fathead minnow, golden shiner (*Notemigonus crysoleucas)*, largemouth bass, northern pike, pumpkinseed (*Lepomis gibbosus)*, tadpole madtom (*Noturus gyrinus)*, white sucker, and yellow perch (*Perca flavescens)*. Species common to both study areas in 2005, but not in 2006, included common carp, Iowa darter (*Etheostoma exile)*, and walleye. In 2006, common carp and walleye were sampled in the HI landscape but not in the LO landscape, while Iowa darter was found in the LO but not in the HI landscape. Green sunfish (*Lepomis cyanellus)* was common to both landscapes in 2006. However, in 2005, green sunfish was sampled only in the LO landscape. Species that were present in the HI landscape, but never present in the LO landscape, included bigmouth buffalo (*Ictiobus cyprinellus)* (2005 only), channel catfish (*Ictalurus punctatus)* (2005 only), freshwater drum (*Aplodinotus grunniens)* (both years), orangespotted sunfish (*Lepomis humilis)* (2005 only), shorthead redhorse (*Moxostoma macrolepidotum)* (both years), and yellow bullhead (*Ameiurus natalis)* (both years). Species present in the LO landscape, but never present in the HI landscape, included blacknose shiner *Notropis heterolepis* (2006 only), brassy minnow (*Hybognathus hankinsoni)* (both years), and northern redbelly dace (*Phoxinus eos)* (both years). Common shiner (*Luxilus cornutus)* was found in the LO but not in the HI landscape in 2005, and vice versa in 2006.

There were also some interesting differences between study landscapes with respect to proportions of the 4 major fish community types. The HI landscape contained 3-4 fishless sites, while only 1 site was fishless in LO landscape (Table 2). The number of planktivore-only (P) sites was similar in both areas, but sites with planktivores, benthivores, and piscivores (PBP) were approximately twice as numerous (n=15-16) as the number of sites containing planktivores and benthivores (PB) (n=8) in the HI Landscape (Table 2). Proportion of sites with common carp also differed between the 2 study landscapes; we sampled 12-14 sites with carp in the HI landscape, but only 0-2 sites in the LO region. Common carp were sampled in 2 additional sites in the HI landscape in 2006. Carp were absent in 1 site in the LO landscape where they were captured in 2005; also, during 2006, we were unable to sample the only other LO landscape site that contained carp in 2005.

Within each of the major fish community types, the biomass of each guild (i.e., planktivore, benthivore, piscivore) was higher in the HI landscape sites than in the LO landscape sites in both 2005 and 2006 (Figure 3). Most striking was the pattern for benthivores, which were 2.9-5.3 times higher in the HI than LO sites in both the PB, and PBP community types. Planktivores in the P sites (HI and LO landscapes), and benthivores in the PBP sites (HI landscape only) also were lower in 2006 compared to 2005. This inter-annual variation may reflect differential over-winter mortality or summerkill between years, less basin interconnectivity in spring 2006 (drier, less snowpack), or spatial-temporal sampling effects (much hotter during 2006 fish sampling—less active, restricted to deep-water refuges). It is also interesting to note that planktivore biomass was always low in the presence of piscivores (i.e., predators), suggesting that piscivory may be an important structuring mechanism in these wetlands, as it is in larger lakes (Tonn and Magnuson 1982, Robinson and Tonn 1989).

Future analyses will focus on predicting fish community characteristics from basin characteristics and geomorphic setting, including the study site's relationship to characteristics of the surface water drainage network, surrounding land uses, as well as anthropogenic factors and features. Other focus areas will include: 1) exploring relationships between fish community "types" and wetland characteristics (invertebrate communities, submerged plant vs. algal dominance, etc.); 2) better understanding inter-specific interactions among wetland fish species (i.e., roles of predation and competition); and 3) understanding the relative roles of watershed features (e.g., agricultural land use) and fish in determining alternative equilibria in wetland ecosystems (clear vs. turbid, etc.).

Aquatic Invertebrates

We assessed potential relationships among fish communities, characteristics of wetland study sites, and wetland invertebrates graphically and using model selection procedures (Anderson and Burnham 2002). In our case, "best" models were selected from combinations of fish community characteristics, wetland features, study site location (focus area), and other site attributes that have been previously shown to influence aquatic invertebrates. In all cases, model fit was assessed using AICc (Anderson and Burnham 2002). To reduce the number of comparisons, we combined taxa, creating 3 aggregate variables including "macroinvertebrates" (common aquatic insects including Diptera, Coleoptera, Odonata, selected Hemiptera, and others), "amphipods" (Gammarus lacustris and Hyallela azteca), and "zooplankton" (primarily Daphnia spp.). Here, we describe preliminary results of analyses using relative abundance of these 3 aggregate taxa from July 2005 (lab processing of 2006 samples has only recently been completed).

Crustacea, aquatic insects, water mites, and snails were the most common aquatic invertebrates collected from study wetlands during 2005. In general, invertebrate taxa in these wetland communities were similar to those reported from recent work in wetlands in Minnesota (Hanson and Riggs 1995, Zimmer et al. 2000, Zimmer et al. 2002).

Relative abundance of macroinvertebrates in our study wetlands was best predicted by models including only mass of submerged aquatic macrophytes (plants) (R^2=0.28); however, a model including mass of planktivorous fish and plants performed nearly as well (R^2=0.30) (Table 3). By a wide margin, our best zooplankton model also included mass of plants and planktivorous fishes (R^2=0.36; Table 4). All of our amphipod models explained < 5 percent of observed variance; here, our best model included only mass of planktivorous fishes (R^2=0.03; Table 5). Alternative amphipod models, including total mass of planktivorous and benthivorous fishes (R^2=0.02) or a single variable depicting water clarity status (turbid or clear), achieved similar fits to observed data (Table 5).

Results indicated that mass of plants and planktivorous fish were important determinants of macroinvertebrates and zooplankton in our study wetlands during 2005, but these influences may interact in complex ways. For example, macroinvertebrate abundance was most influenced by plant mass, but also reflected abundance (mass) of planktivorous fish (Figure 4). All of our preliminary models explained < 5 percent of observed variability in amphipods. In no cases did our other invertebrate models explain more than approximately 30 percent of observed variability in 2005 data. Along with results of amphipod models, this probably indicates that important determinants of invertebrate community structure were not accounted for in our preliminary analyses.

We will continue to develop data from vertical column and activity trap samples collected during 2006. We have nearly completed enumeration of the samples collected in 2006. We expect that aquatic invertebrates will constitute an important response variable in our analyses of fish and land use effects. We will also explore potential influences of spatial and hydrological variables (such as distance to other water bodies and position in a watershed) on wetland invertebrate characteristics in these study sites.

Limnological Characteristics

In 2005, average total phosphorus (TP) concentrations (µg/L) increased between June and July in both the HI and LO landscapes (HI: 20% increase, LO: 55% increase), and were 7.2 and 5.6 times higher in the HI than LO landscape sites in June and July, respectively (Figure 5). In 2006, average total phosphorus (TP) concentrations (µg/L) again increased between June and July in the HI and LO landscapes (HI: 18% increase,
LO: 60% increase). The HI landscape sites had 4.9 and 3.6 times higher TP level than the LO landscape site in June and July, respectively (Figure 5).

TP averaged 22.2-34.5 μg/L across months in the LO landscape in 2005, sometimes falling within the range of TP values favoring persistent, clear-water, macrophyte-dominated conditions (Moss et al. 1996). In contrast, TP in the HI landscape averaged 161-193 μg/L across months in 2005, levels where basins can exhibit either clear-water, macrophyte-dominance or turbid, phytoplankton dominance (Moss et al. 1996). In 2006, TP exhibited a greater range in values, averaging 12.5-240 μg/L across months in the LO landscape and 24.5-609 μg/L across months in the HI landscape. Despite this greater range and higher TP values in 2006, TP exceeded 50 μg/L in 6 of 36 sites and 150 μg/L in just 1 site in the LO landscape. This contrasts strongly with the HI landscape, where TP exceeded 50 μg/L in 34 of 39 sites, and 150 μg/L in 15 of 39 sites. Data from the UK and elsewhere suggests that TP <150 μg/L is required for dominance by a diverse macrophyte community, and that macrophyte communities exhibit higher stability at TP <50 μg/L (reviewed in Madgwick 1999). Thus, many of the study sites (~56%) in the HI landscape have TP concentrations within a range where there is considerable potential for restoring macrophyte dominance through fish community manipulations, etc.

In 2005, mean turbidity was 2.4 times higher in the HI landscape sites than in the LO landscape sites in both June and July (Figure 6). Turbidity increased by approximately 50% between June and July in the HI landscape and by 12% in the LO landscape, reflecting a seasonal increase in phytoplankton abundance. In 2006, HI landscape sites had 2.7 and 2.8 times higher mean turbidity than the LO landscape sites in June and July, respectively. There was a 51% increase between June and July in the HI landscape and 44% increase in the LO landscape sites.

LO landscape sites were relatively clear (<5 NTUs), with just 9 sites having a turbidity >5 NTUs in both 2005 and 2006. In contrast, the HI landscape sites exhibited considerably more variability, with some sites characterized by clear water (10 sites had a turbidity < 5 NTUs in 2005, and 15 in 2006), but many sites characterized by very turbid water (14 sites had a turbidity >20 NTUs in 2005, and 11 in 2006).

Distributions of chlorophyll *a* (Chla) concentrations (μg/L) between the 2 landscapes were dramatically different in both 2005 and 2006 (Figure 7). In June 2005, there were only 9 sites in the LO landscape with Chla >15 μg/L, while there were 23 sites in the HI landscape with Chla >15 μg/L. In the HI landscape, this increased to 27 sites in July, with 10 sites exceeding 90 μg/L Chla. In June 2006, the LO landscape had only 5 sites with Chla >15 μg/L, while the HI landscape had 22 sites. In the HI landscape, this increased to 26 in July, with 9 sites exceeding 90 μg/L Chla (highest was 534.5 μg/L).

As for turbidity, there were several sites in the HI landscape with low levels of Chla (e.g., 10 sites in July 2005 with Chla <15 μg/L). These data suggest that alternative conditions of clear water, macrophyte dominance and turbid, phytoplankton dominance are represented in both landscapes, although the proportion of turbid sites is considerably higher in the HI landscape.

Submerged Macrophytes

Submerged plant biomass was similar between the HI and LO landscapes, as evidenced by overlapping standard errors (Figure 8). In contrast, the average submerged plant score showed that the coverage and diversity of macrophytes was higher in the LO landscape than in the HI landscape (68 vs. 32 in 2005, 74 vs. 40 in 2006). Although mean biomass of submerged plants was similar among study areas, the distribution of submerged plant biomass differed appreciably (only 2006 data summarized; Figure 9). In the HI landscape, there were many sites with low plant biomass (about half of the sites). The remainder of sites had intermediate to high plant biomass (long right tail). In contrast, in the LO landscape we observed only a few sites with low or high submerged plant biomass, with most sites having intermediate to high plant biomass.

Waterfowl Use

Despite high variability, more puddle and diving duck breeding pairs were observed within the HI landscape area, especially during 2005 (Figure 10). Overall, for both puddle and diving ducks combined, breeding pairs were observed on 92 and 77 percent of wetland study sites in HI and LO landscapes (respectively) during 2005 (Table 6). This trend continued during 2006 when breeding pairs were observed on 90 and 71 percent of wetlands in HI and LO landscapes. As with breeding pairs, more duck broods were also observed within the HI landscape area; this geographical contrast was much more obvious during the first year of our study (2005; Figure 10). Overall, duck broods were observed on 44 and 23 percent of wetland study sites in HI and LO landscapes (respectively) during 2005. This trend continued during 2006 when breeding pairs were observed on 45 and 36 percent of wetlands in HI and LO landscapes (Table 6).

Amphibian Populations

Our current analysis is limited to comparisons between landscapes and relative to fish community characteristics, as we do not yet have final data on the surrounding land cover types. Results from 2005 and 2006 indicated that highest abundances of tadpoles and larval salamanders were found in fishless wetlands in both HI and LO landscapes (Figures 11,12). Across all types of fish communities, the general abundance pattern for both tadpoles and salamanders in 2005 was fishless > planktivores (P) > planktivores/benthivores (PB) > planktivores/benthivores/piscivores (PBP) (Figure 11). A similar pattern was observed in 2006, but only in the HI landscape (Figure 12).

Few tadpoles or salamanders were captured in wetlands with piscivores (PBP sites) in either study landscape in both 2005 and 2006 (Figures 11,12). Finally, we observed no consistent relationships between relative abundance of painted turtles and fish community types. However, the lowest abundance of painted turtles occurred in the HI landscape PBP sites in 2005, and PB and PBP sites in 2006.

Periphyton Distribution and Dynamics

In 2005, sites in the HI landscape (534 μg/L) had a higher average periphyton biomass per wetland than those in the LO landscape (266 μg/L). HI landscape sites had a wider range of periphyton biomass than LO landscape sites (Figure 13). Periphyton biomass generally decreased with water depth (slide positions: top, middle, bottom) (Figure 14).

During 2006, lakes in the HI landscape (918 μg/L) again had a higher average periphyton biomass per wetland than those in the LO landscape (109 μg/L). Wetlands in the HI landscape also had a wider range of periphyton biomass values than those in the LO landscape (Figure 13). One wetland in Grant County (HI) had an average biomass of 5727 μg/L, whereas the highest value in Polk County (LO) was 411 μg/L. There were no consistent trends between periphyton biomass and water (slide) depth (Figure 14). Overall, results show that there is considerable variance within each of the landscapes, especially among sites in the HI landscape.

Model selection and model averaging showed that macrophyte biomass was the best single predictor of periphyton biomass at all depths in 2005 (data not shown). No fish parameters were present in any of the candidate models, thus fish were poor predictors of periphyton biomass. Nutrient (particularly phosphorus) and macroinvertebrate variables occurred repeatedly in candidate models predicting periphyton biomass at each depth. Therefore, preliminary results show that periphyton biomass is influenced by both top-down (macroinvertebrates) and bottom-up (macrophytes and nutrients) factors.

SUMMARY

Data gathered during 2005 and 2006 indicated that fish populations occurred in nearly all wetland study sites. Diverse fish communities were common and often contained combinations of planktivorous, benthivorous, and piscivorous species. As far as we know, our research is the first to simultaneously measure direct and indirect influences of fish on prairie wetland characteristics in the sense that, in addition to site-level effects, we are also assessing fish communities in response to landscape characteristics at several spatial scales.

Previously, we reported results of preliminary analyses indicating that attributes of adjacent landscapes were poor predictors of fish populations in study wetlands; however, fish communities did reflect wetland size and depth, along with site-level influences of piscivores (Hanson et al. 2006). Preliminary data summarized here suggests that wetland fish abundance and/or community type were associated with important components of wetland study sites including aquatic invertebrates, limnological characteristics (such as water clarity and phytoplankton abundance), and relative abundance of amphibians. Early results also indicated that fish influences differ among feeding guilds (planktivores – fathead minnows, benthivores – black bullheads). For example, increasing mass of benthivorous fishes was significantly associated with declining mass of submerged aquatic plants during 2005 (Hanson et al. 2006) and 2006 (data not shown here); in contrast, no similar relationships was observed between submerged plants and planktivorous fishes in either study year. It is also notable that fish influences are complex and often interact with other wetland characteristics. For example, during 2005, planktivore mass was negatively associated with relative abundance of macroinvertebrates in study wetland. However, our best invertebrate models also showed strong evidence of interactions between mass of planktivorous fish and submerged plant mass; this indicates that macroinvertebrates were most strongly suppressed in wetlands with high mass of planktivorous fish and low mass of submerged plants.

Finally, location was also an important determinant of wetland characteristics; study sites often differed dramatically in key ecological features between our HI and LO landscape study areas. For example, benthivorous fish mass, turbidity, total phosphorus, and phytoplankton concentrations tended toward higher values in our HI (Grant County) study sites. In part, this probably reflects an increasing gradient of nutrient availability along a statewide SW to NE trajectory, but may also relate to differences in proportions of agriculture or other anthropogenic differences between these regions. It may also indicate greater vulnerability of Grant County wetlands to shift to turbid-water states. Future analyses will focus on clarifying relationships among landscape cover types, surface-water connectivity, site-level geomorphic setting, and other spatial characteristics of wetland sites and associated fish communities, along with additional clarification of site-level influences of fish populations.

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Table 1. Landscape features captured using existing GIS layers or digitized using 2003 FSA air photos and 1 square mile land use maps as primary references.

Description	Definitions	Polygon source	Our label	FSA label
Grasslands: CRP, pasture, WPA and WMA uplands	· Grassy, does not include row crops or hay/alfalfa, but does include pasture · Established grassy uplands on WPAs and WMAs	Digitized	GRA	NC, FWS, DNR, CRP
	· Vegetated portions of right-of-ways associated with transportation	Buffered		
Woodlands	· Forested areas, with ground cover of greater than 75% mature trees	Digitized	WDL	NC, FWS, DNR
Shrubs	· Shrubby area mixed with grassy area, woodland area with ground cover of less than 75% mature trees	Digitized	SBL	NC, FWS, DNR
Row crops and hay	· Tilled crops, generally corn, soybeans, and small grains · Areas that are hayed annually including alfalfa and wild hay	Digitized	AGR	HEL, MHEL, NW
Non-study site lakes	· Entire area of lake or wetland including emergent vegetation (Lakes, Type IV, V wetlands, bogs with at least 10% open water and lakes)	Digitized	LKS	W
Non-study site wetlands	· All non-Type IV or V wetlands, and bogs with <10% open water; minimum size of 0.1 ha to be digitized	Digitized	WTL	W, CW, FW
Study sites	· Open water portion of the wetland	Digitized	OWT	W
	· Emergent vegetation along basin margins (use GPS reference points as guide). Includes cattails, sedges, Phragmites spp.	Digitized	EAV	
	· Islands with trees and shrubs	Digitized	ISL	
	· Emergent vegetation in the interior of basins (cattail islands)	Digitized	CTI	
Streams	· Continuously wetted and intermittent streams.	Quick Themes	CST	No label
Ditches	· Ditches containing water in fields (straight/linear, and contain water that you can see on an air photo) · Ditches associated with public roads and driveways	Digitized	DWT	No label
Farmsteads	· Active and abandoned farmsteads/homesteads and associated buildings and shelterbelts etc. regardless of size, but not the associated woodlands	Digitized	FST	NC
Roads	· Transportation surfaces	Quick Themes	RDS	No label
Other impervious surfaces	· Gravel pits and parking lots, towns	Digitized	OIS	No label
Driveways	· Driveways associated with farmsteads/homesteads	Digitized	DVW	No label

Table 2. Number of sites corresponding to each of the four major fish community types within the HI ("Grant County") and LO ("Polk County") landscapes in 2005 and 2006. Also tabulated is the number of sites falling within the P-Benthivores and P-B-Piscivores community types that also contain common carp.

	HI		LO	
Fish community "type"	No. of sites	No. of sites with common carp	No. of sites	No. of sites with common carp
2005				
Fishless	3	-	1	-
Planktivores	10	-	9	-
P-Benthivores	8	4	17	0
P-B-Piscivores	16	8	9	2
2006				
Fishless	4	-	1	-
Planktivores	12	-	10	-
P-Benthivores	8	5	16	0
P-B-Piscivores	15	9	8	0

Table 3. Results of model selection procedures using macroinvertebrate data gathered from study wetlands during 2005. Model fit was assessed using AICc values and resulting evidence ratios. Plant Mass was the best indicated model (R^2=0.28; indicated in bold).

Model terms	K	n	AICc	Evidence ratio
Plant Mass	**3**	**73**	**26.27**	**1.0**
Planktivore Mass	3	73	45.09	12,206.0
Benthivore Mass	3	73	46.96	31,085.4
Planktivore+Benthivore Mass	3	73	39.79	866.9
Water clarity (turbid or clear)	3	73	45.57	15,537.2
July Chlorophyll a	3	73	43.08	4481.0
Plant Mass * Planktivore Mass	5	73	26.47	1.1
Plant Mass * Benthivore Mass	5	73	30.48	8.2
Plant Mass * Plank + Benth Mass	5	73	29.30	4.5

Table 4. Results of model selection procedures using zooplankton (*Daphnia*) data gathered from study wetlands during 2005. Model fit was assessed using AICc values and resulting evidence ratios. Plant Mass* Planktivore Mass (interaction term) was the best indicated model (R^2=0.31; indicated in bold).

Model terms	K	n	AICc	Evidence ratio
Plant Mass	3	73	124.32	17.6
Planktivore Mass	3	73	142.88	187765.7
Benthivore Mass	3	73	131.67	695.5
Planktivore+Benthivore Mass	3	73	135.20	4057.0
Water clarity (turbid or clear)	3	73	136.62	8256.6
July Chlorophyll a	3	73	132.74	1188.7
Plant Mass * Planktivore Mass	**5**	**73**	**118.58**	**1.0**
Plant Mass * Benthivore Mass	5	73	125.77	36.3
Plant Mass * Plank + Benth Mass	5	73	128.40	135.6

Table 5. Results of model selection procedures using amphipod data gathered from study wetlands during 2005. Model fit was assessed using AICc values and resulting evidence ratios. Planktivore Mass was the best indicated model (R^2=0.03; indicated in bold).

Model terms	K	n	AICc	Evidence ratio
Plant Mass	3	73	51.46	3.3
Planktivore Mass	3	73	49.05	1.0
Benthivore Mass	3	73	51.29	3.1
Planktivore+Benthivore Mass	3	73	49.63	1.3
Water clarity (turbid or clear)	3	73	50.40	2.0
July Chlorophyll a	3	73	51.33	3.1
Plant Mass * Planktivore Mass	5	73	51.10	2.8
Plant Mass * Benthivore Mass	5	73	54.80	17.7
Plant Mass * Plank + Benth Mass	5	73	53.67	10.1

Table 6. Breeding pair and duck brood characteristics in the HI (Grant County) and LO (Polk County) study landscape during 2005-2006.

	Study Landscape			
	HI		LO	
Characteristic	2005	2006	2005	2006
Number of sites with at least one breeding pair	34	35	27	25
Total number of study sites	37	39	35	35
Percentage of sites with at least one breeding pair	92%	90%	77%	71%
Number of sites with at least one brood	16	17	8	13
Total number of study sites	36	38	35	36
Percentage of sites with broods	44%	45%	23%	36%

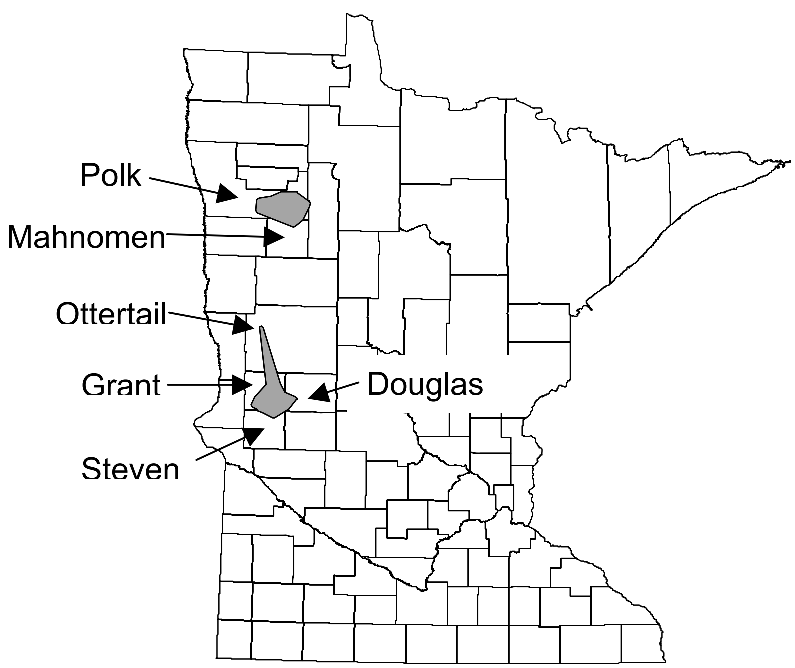

Figure 1. Map showing locations of study landscapes. Study areas are defined by a polygon drawn around the outermost 1-mile buffers surrounding each of the study sites.

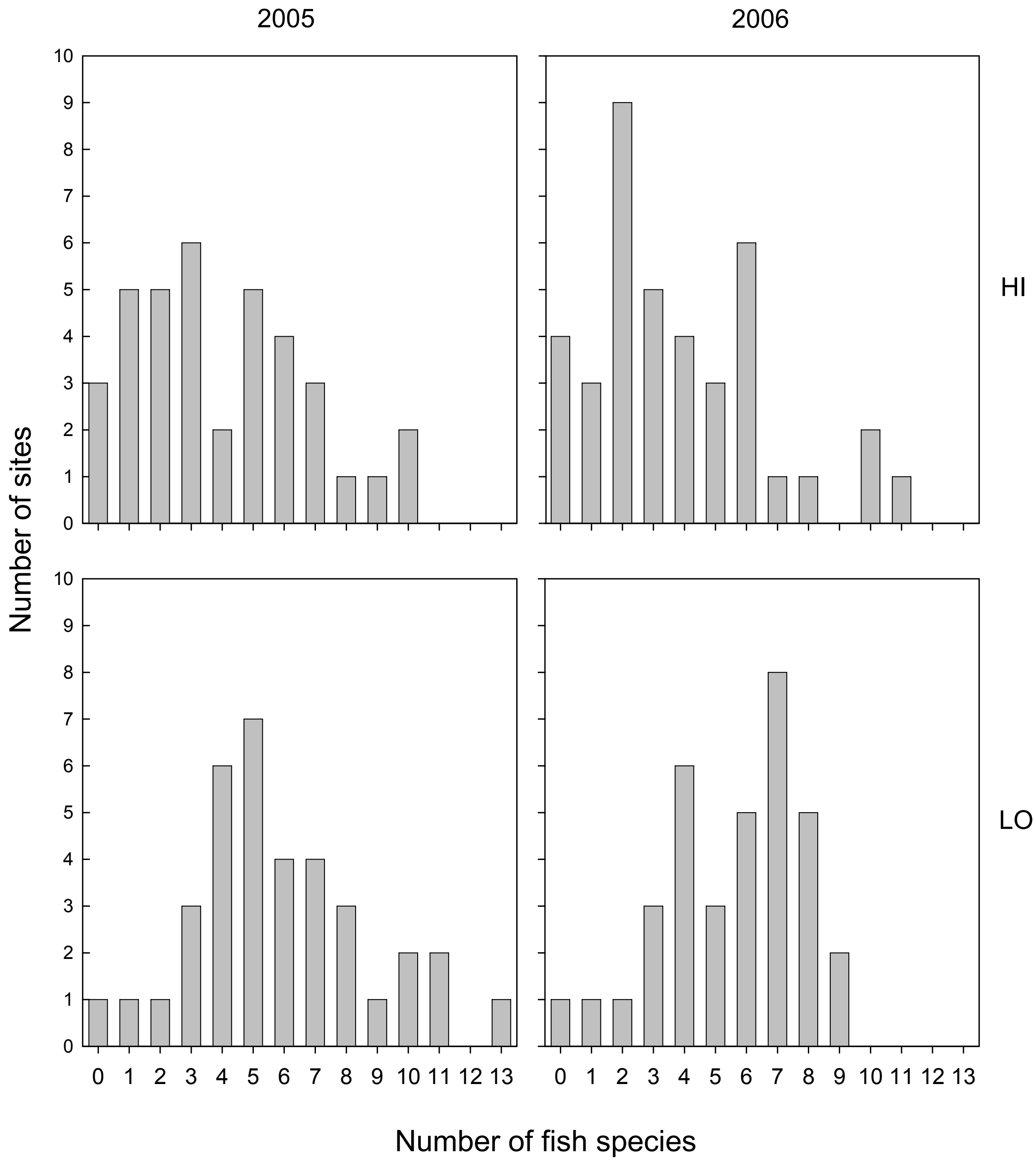

Figure 2. Frequency distributions showing fish species richness for study sites located in the HI landscape ("Grant Co" – top panels) and LO landscape ("Polk Co." – bottom panels) in summer 2005 and 2006.

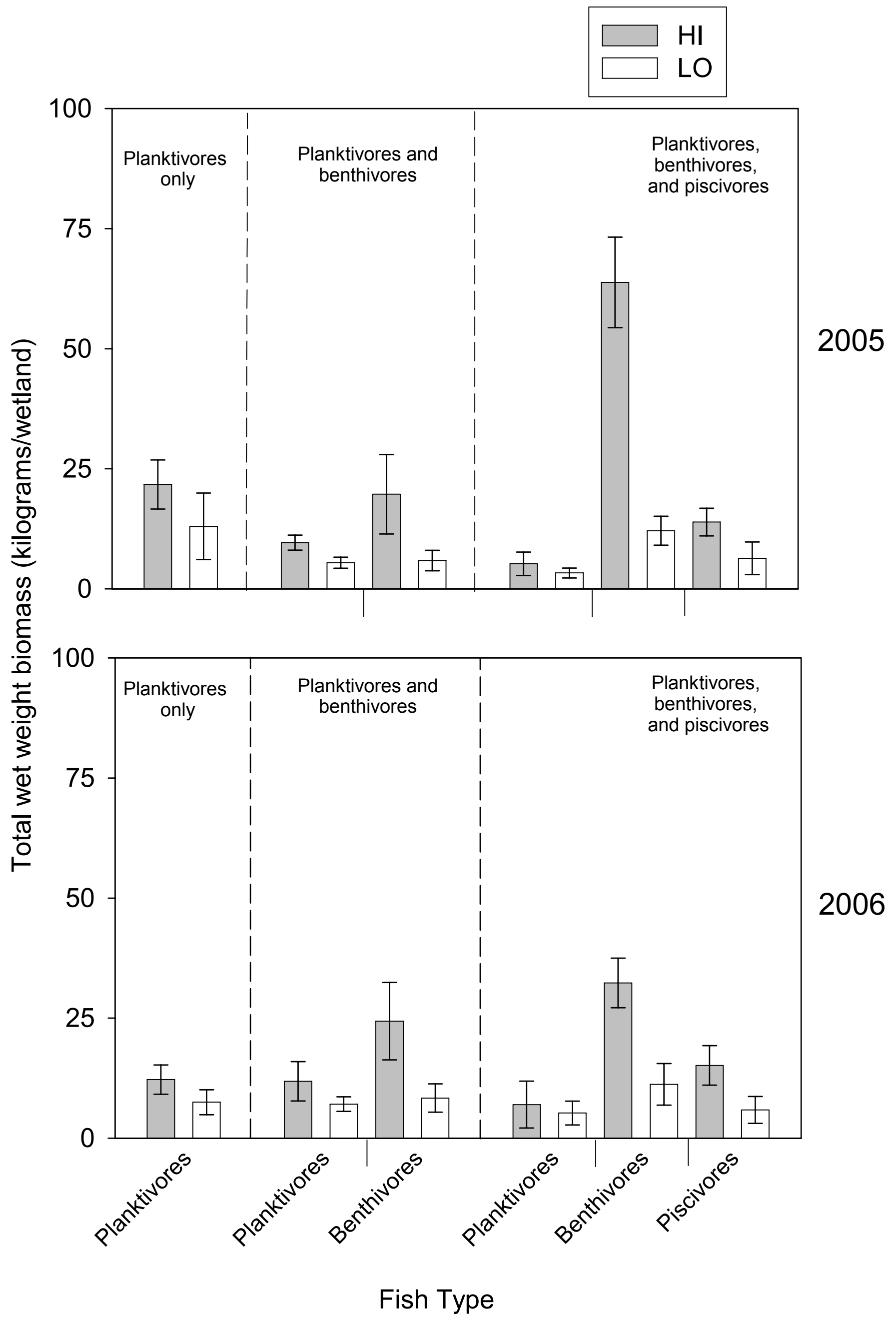

Figure 3. Average fish biomass (+/- 1 SE), summarized by guild (i.e., planktivores, benthivores, piscivores), for each of the major fish community types for the HI landscape ("Grant Co" – gray bars) and LO landscape ("Polk Co." – white bars) in 2005 (top panel) and 2006 (bottom panel).

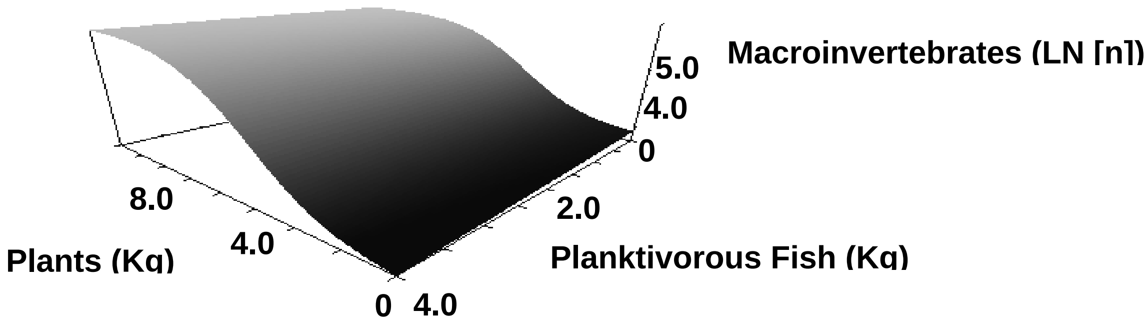

Figure 4. Relationship among macroinvertebrate abundance, mass of submerged aquatic plants (plants) and mass of planktivorous fish measured in 73 study wetlands during July 2005. Smoothed surface depicts predicted values derived using nonparametric multiplicative regression model (Hyperniche [McCune and Mefford 2004]).

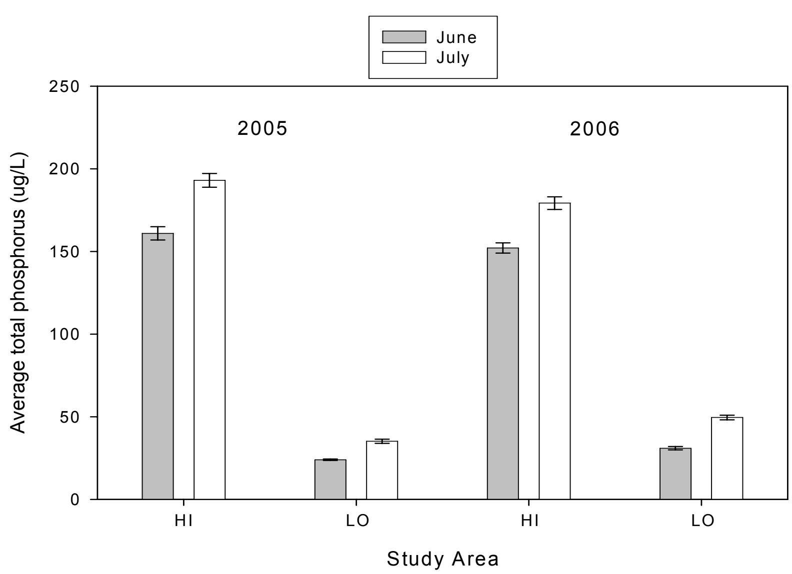

Figure 5. Average total phosphorus concentration (μg/L) for study sites in the HI landscape ("Grant Co") and LO landscape ("Polk Co") in June (black bars) and July (grey bars) 2005 and 2006. Error bars are +/- 1 SE.

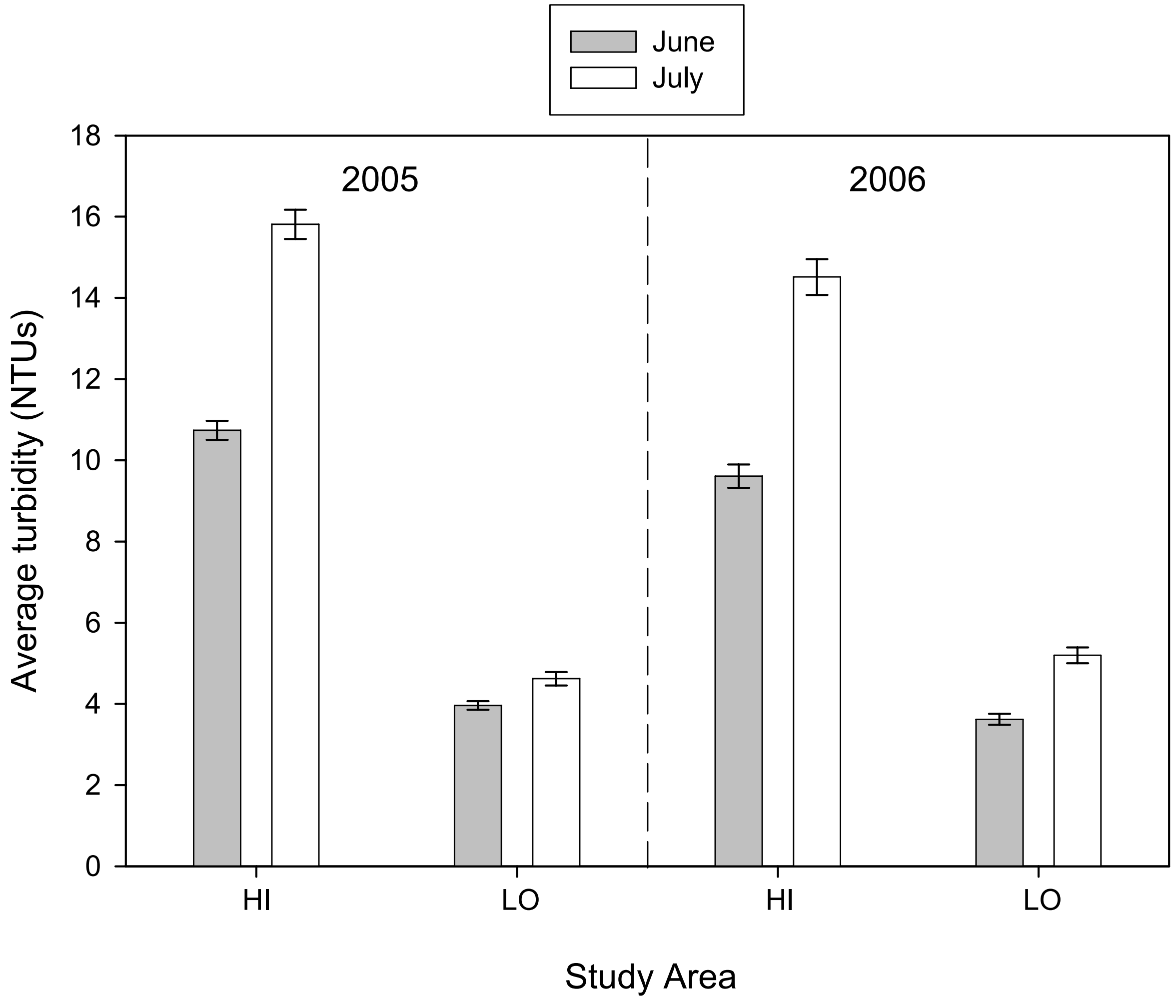

Figure 6. Average turbidity readings (NTUs) for study sites in the HI landscape ("Grant Co") and LO landscape ("Polk Co") in June (black bars) and July (grey bars) 2005 and 2006. Error bars are +/- 1 SE.

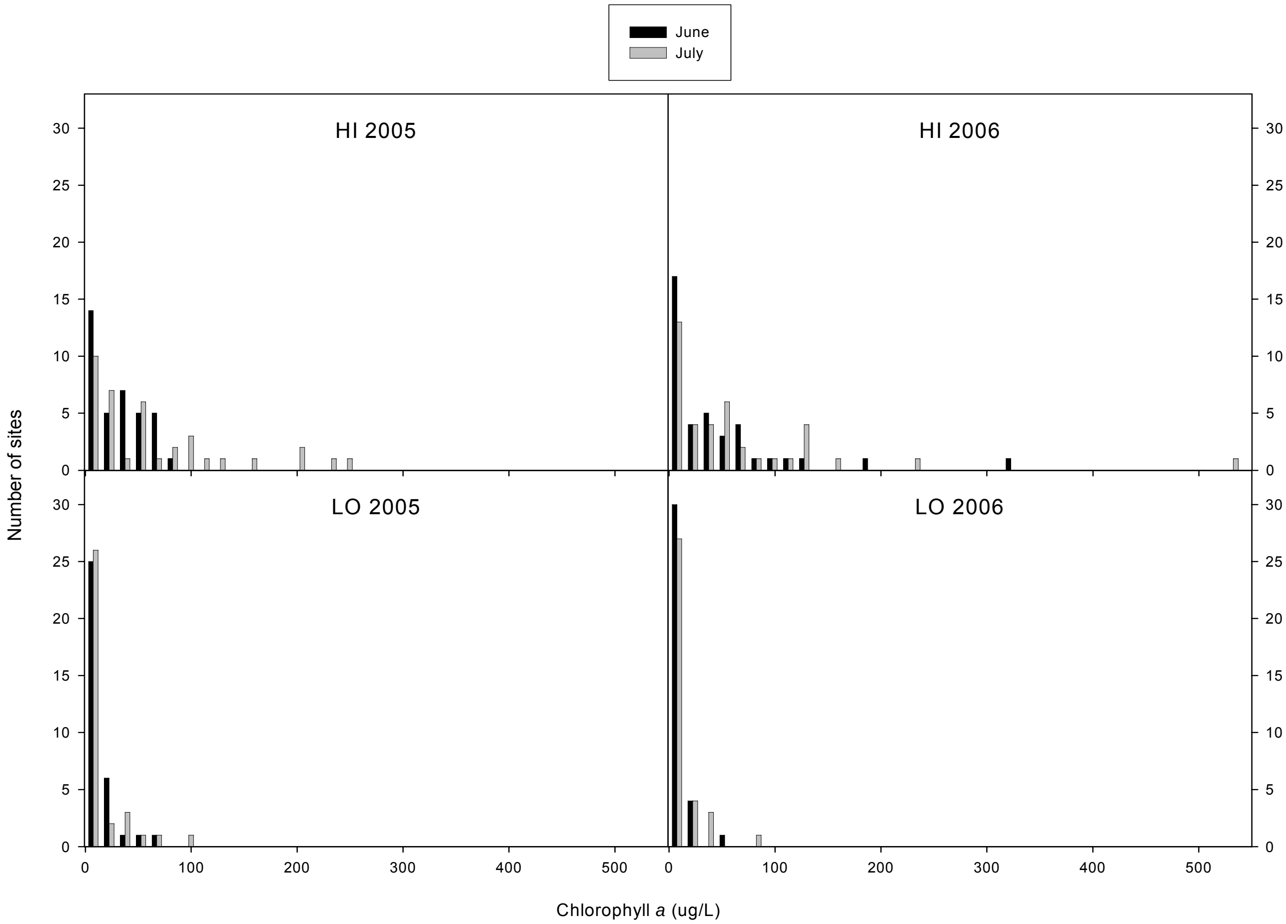

Figure 7. Histogram showing the distribution of average chlorophyll *a* concentrations (µg/L) observed in study sites located in the HI landscape (“Grant Co” – top panels) and LO landscape (“Polk Co” – bottom panels) in June (black bars) and July (gray bars) 2005 and 2006.

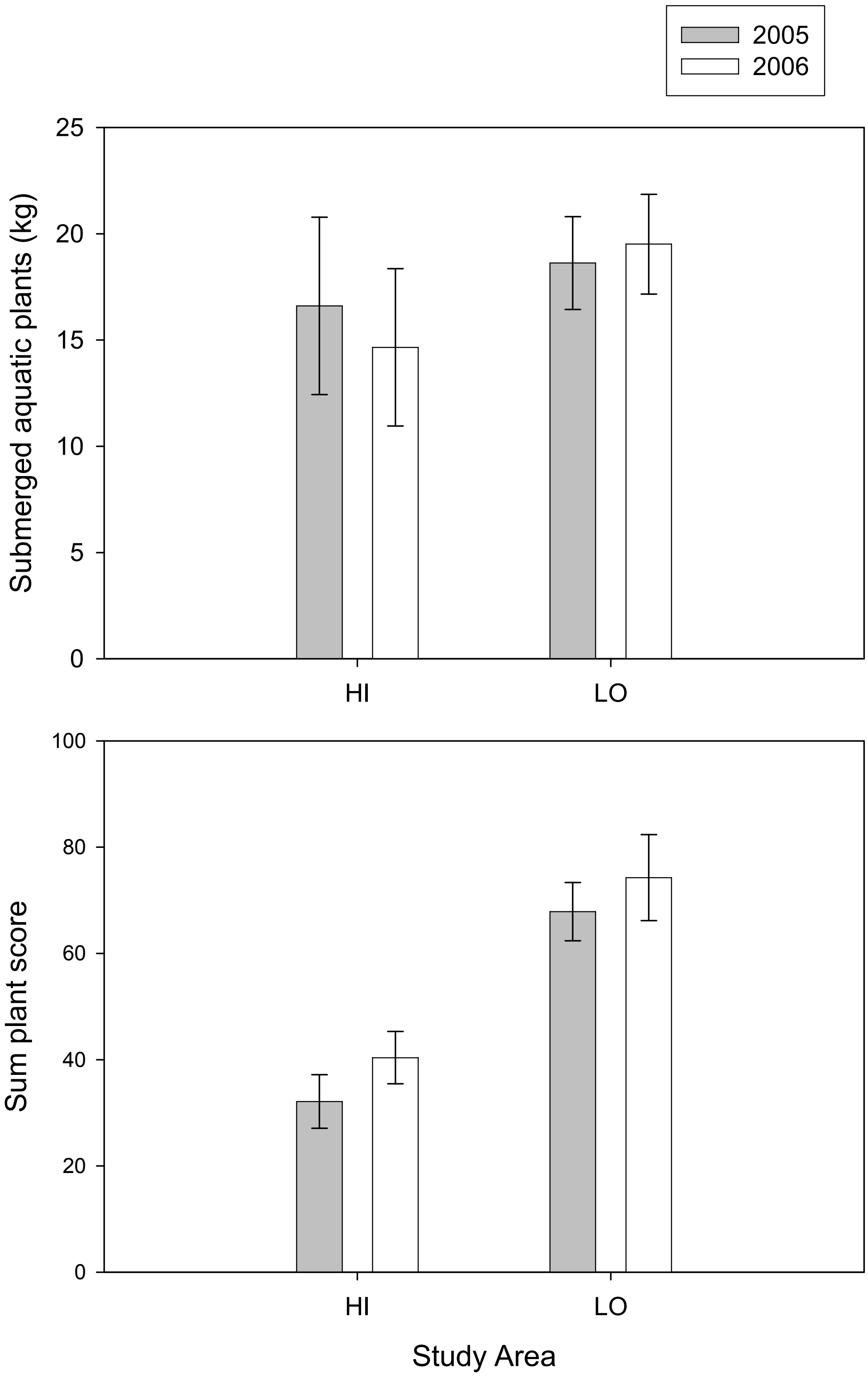

Figure 8. Average sum of submerged plant biomass (top panel) and average submerged plant score (bottom panel) for study sites in the HI landscape ("Grant Co") and LO landscape ("Polk Co") in 2005 and 2006. Error bars are +/- 1 SE.

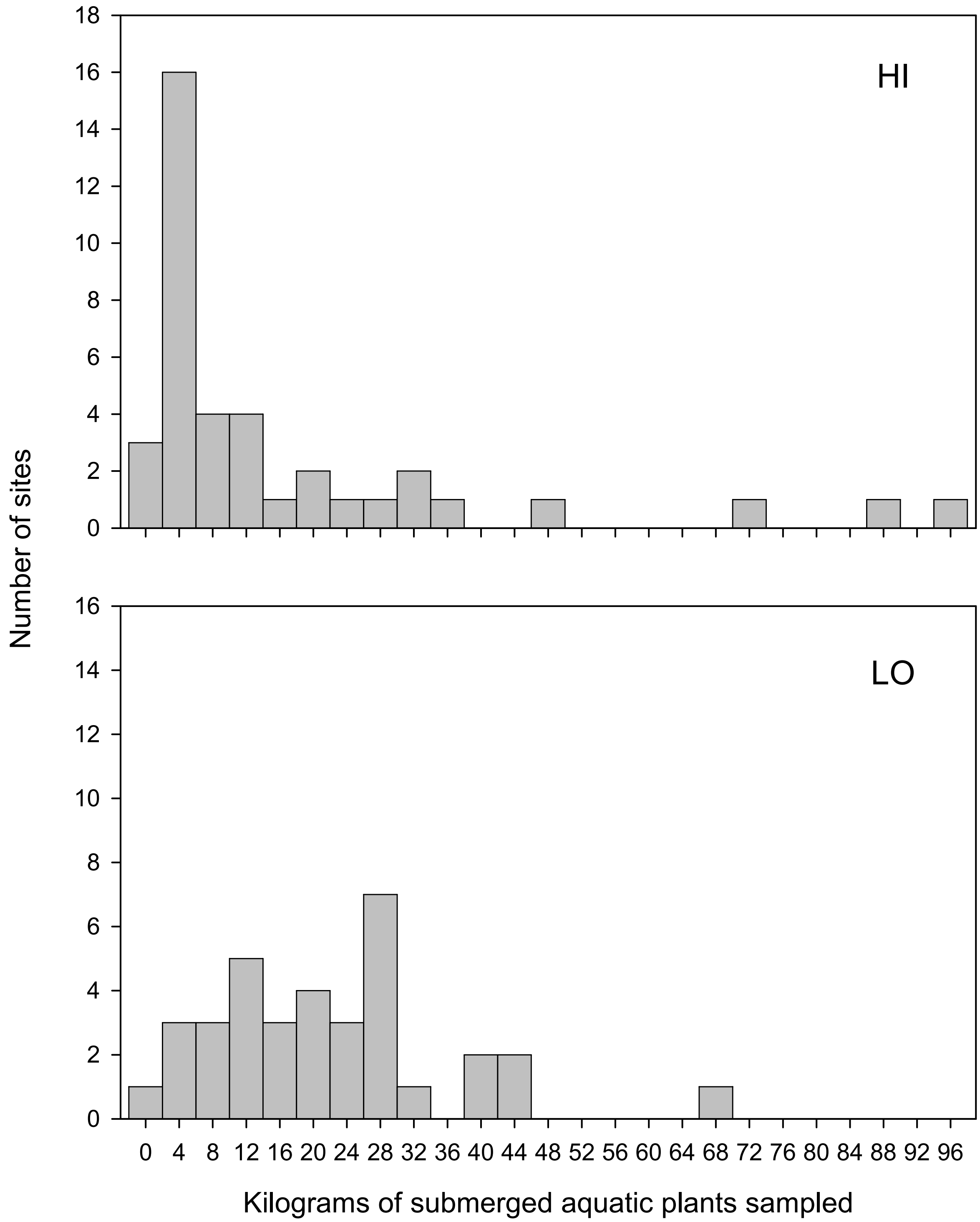

Figure 9. Histograms showing distribution of submerged plant biomass (summed across stations) within study sites in the HI landscape focus area ("Grant Co" - top panel) and LO landscape focus area ("Polk Co" – bottom panel) in summer 2006.

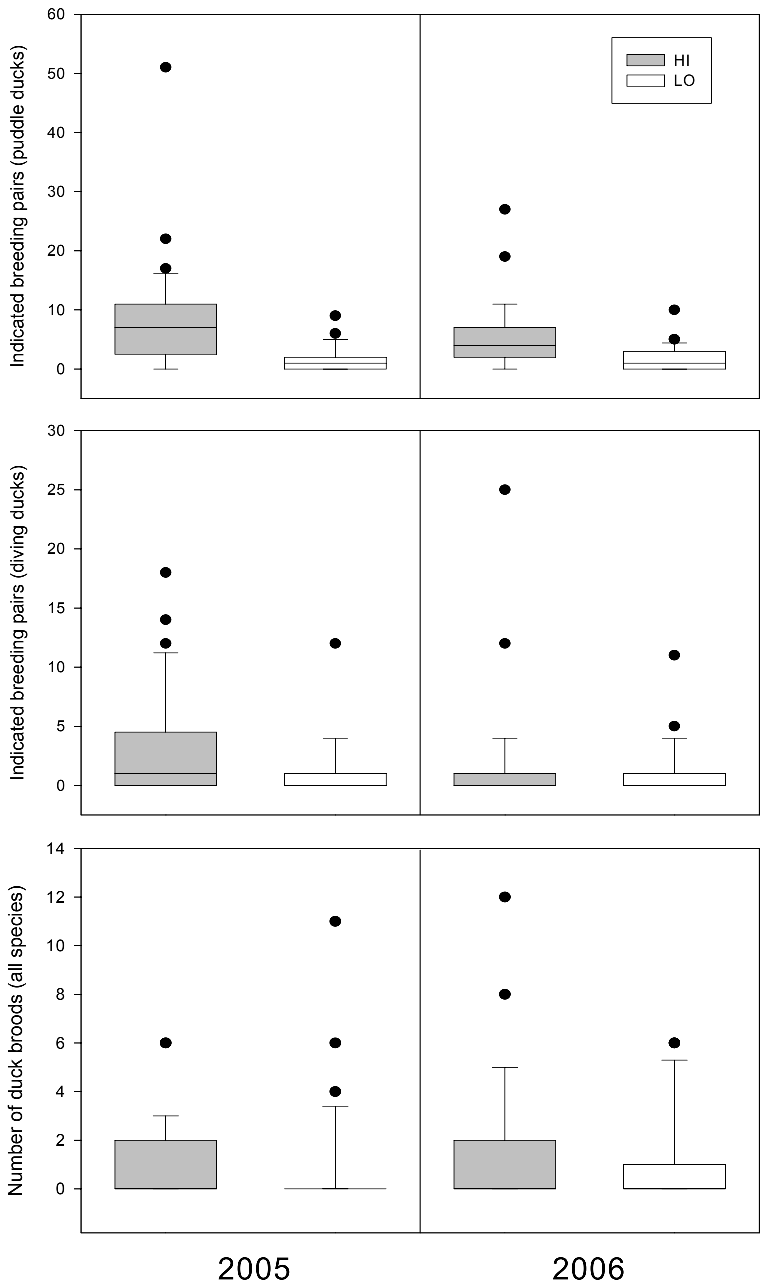

Figure 10. Number of indicated breeding duck pairs and duck broods observed on wetland study sites during 2005 and 2006. Box plots depict median values (central horizontal line), along with the 10th, 25th, 75th, and 90th percentiles and outliers beyond 10 and 90 percentiles (indicated by whiskers). Left-hand bars indicate ducks observed in HI study landscape; right-hand bars indicate ducks observed in LO study landscape.

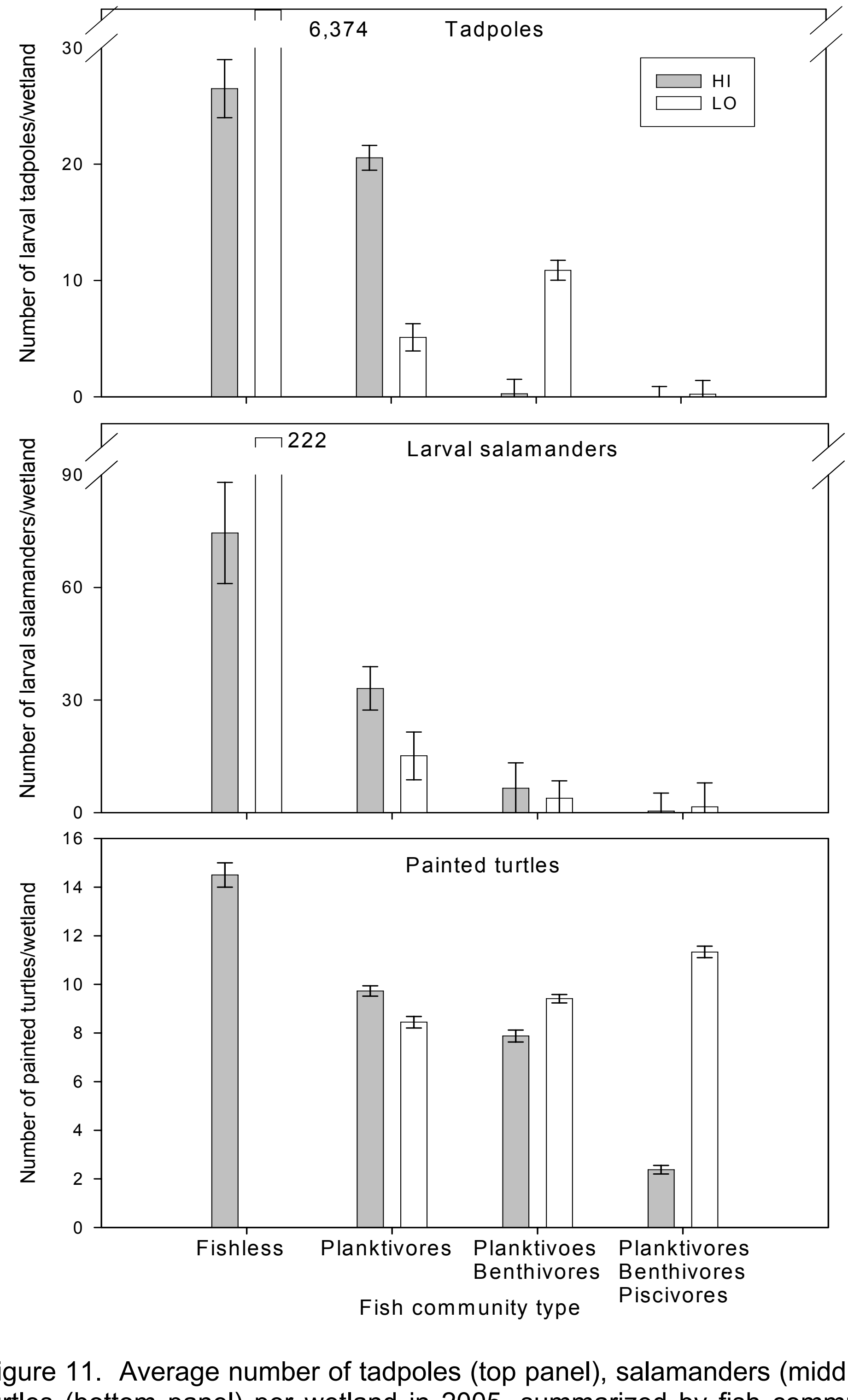

Figure 11. Average number of tadpoles (top panel), salamanders (middle panel), and painted turtles (bottom panel) per wetland in 2005, summarized by fish community types for the HI landscape ("Grant Co" – gray bars) and LO landscape ("Polk Co." – white bars). Error bars represent +/- 1 SE.

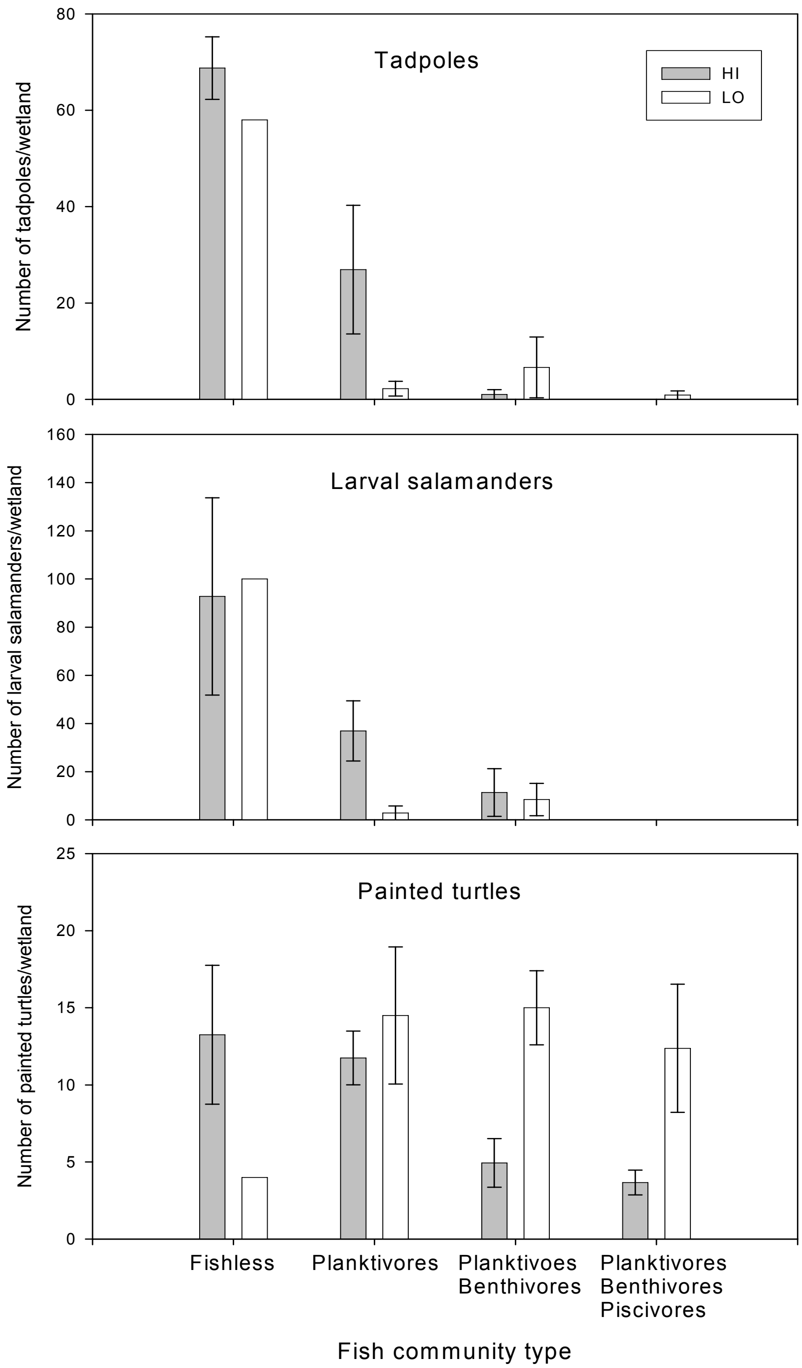

Figure 12. Average number of tadpoles (top panel), salamanders (middle panel), and painted turtles (bottom panel) per wetland in 2006, summarized by fish community types for the HI landscape (“Grant Co” – black bars) and LO landscape (“Polk Co.” – grey bars). Error bars represent +/- 1 SE.

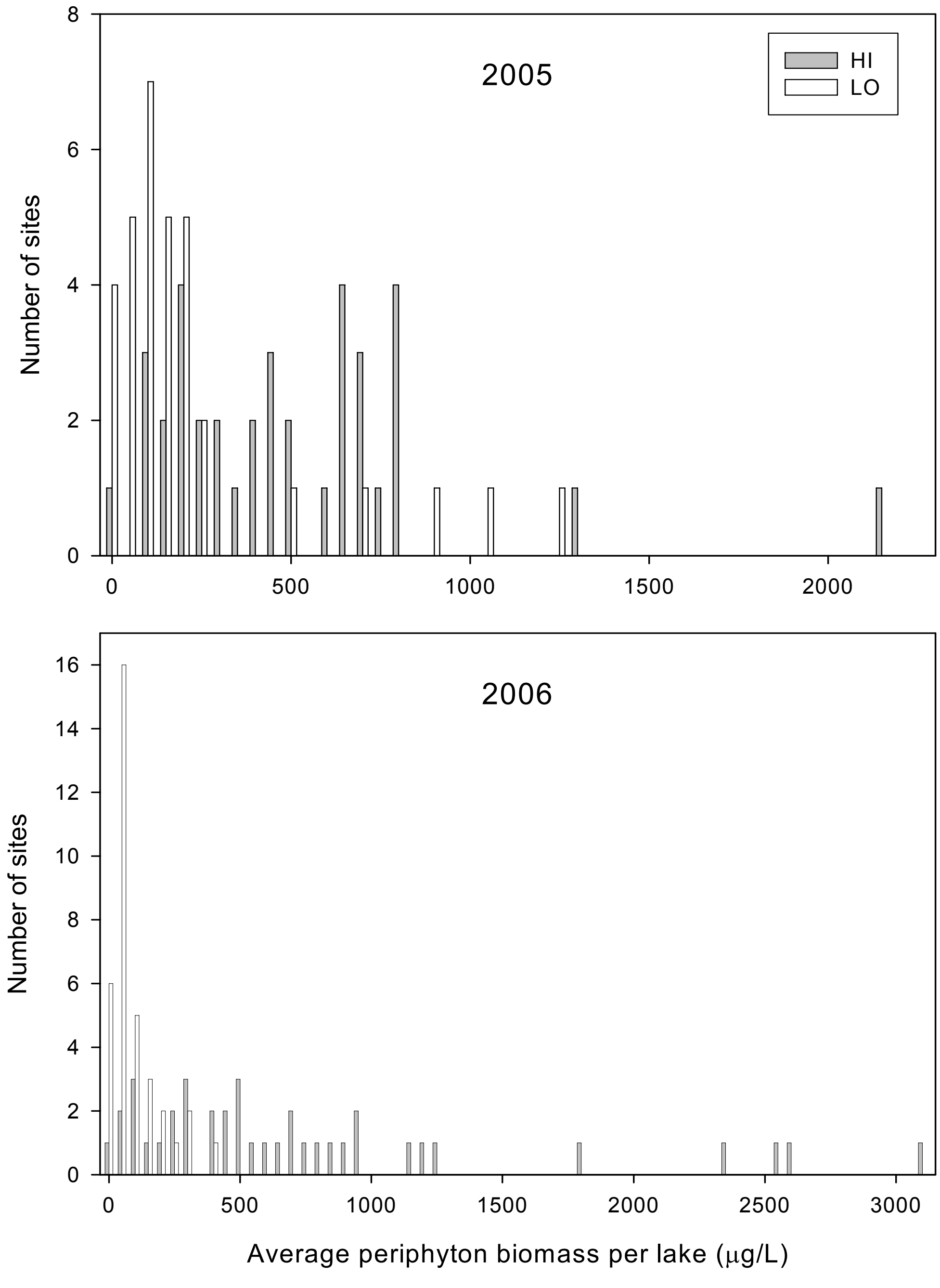

Figure 13. Histograms showing the distribution of chlorophyll *a* concentrations (ppb) of periphyton samples (average per lake) collected from study sites located in the HI landscape ("Grant Co" – gray bars) and LO landscape ("Polk Co" – white bars) in 2005 (top) and 2006 (bottom). There was a value of 5727 ug/L in the HI landscape in 2006 but it is not shown in the histogram.

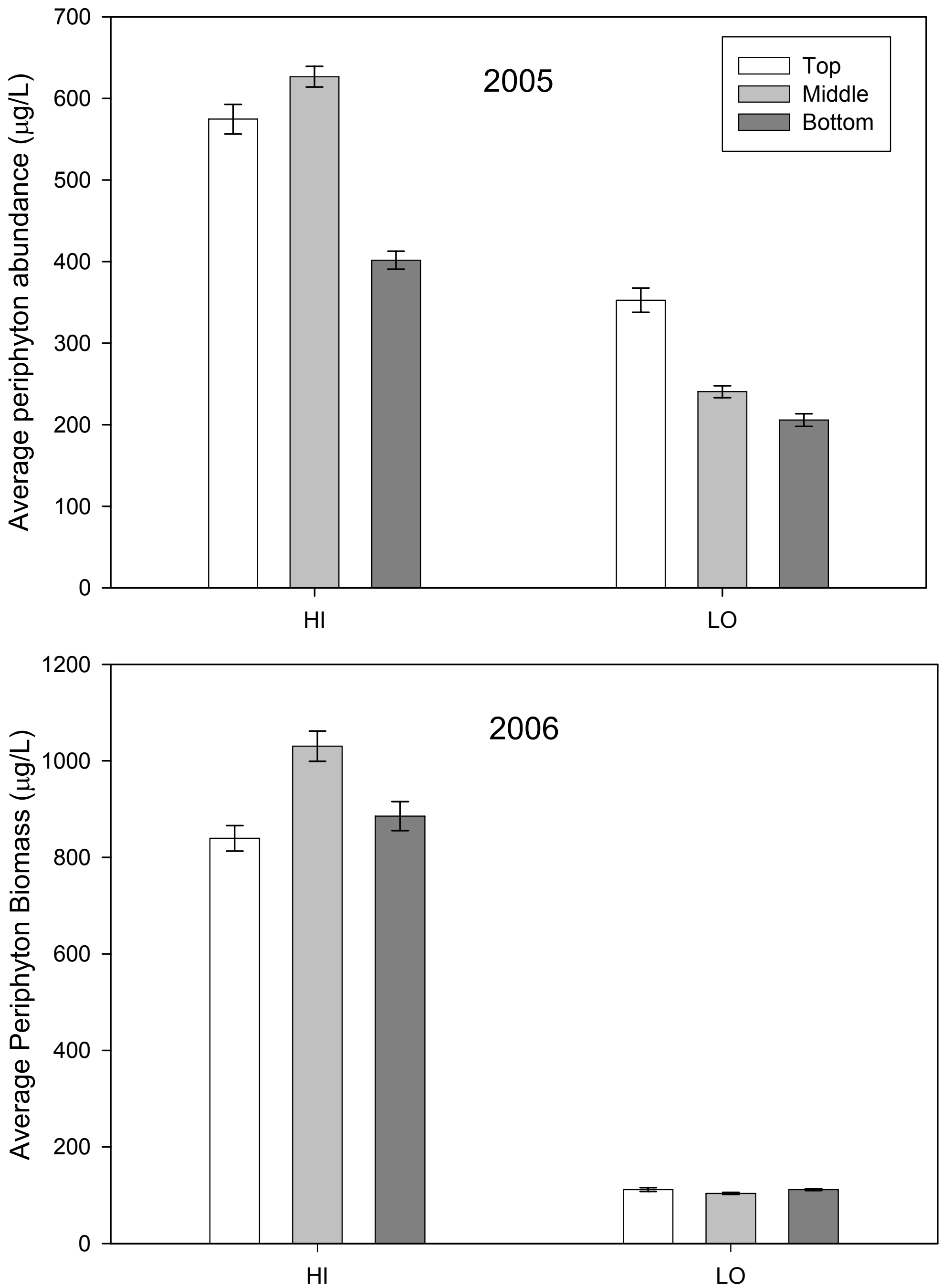

Figure 14. Average chlorophyll *a* concentration (ppb) of periphyton samples on the top slide (black), middle slide (light grey), and bottom slide (dark grey) from study sites located in the HI landscape (Grant Co.) and LO landscape (Polk Co.) in 2005 (top) and 2006 (bottom). Error bars represent +/- 1 sample SE.

THE VALUE OF FARM PROGRAMS FOR PROVIDING WINTER COVER AND FOOD FOR MINNESOTA PHEASANTS

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SUMMARY OF FINDINGS

The purpose of this study was to determine how much winter habitat is needed to sustain local populations of ring-necked pheasants (*Phasianus colchicus*) over a range of winter conditions. We estimated relative abundance of pheasant populations on 36 study areas using roadside surveys. In addition, we estimated amounts of winter cover, winter food, and reproductive cover on each study area by cover mapping to a Geographic Information System (GIS). During 2003-2006, pheasant population indices varied in association with weather and habitat. A preliminary evaluation indicated that mean pheasant indices were positively related to habitat abundance in most, but not all, regions. Four consecutive mild winters have hampered our ability to estimate winter habitat needs. Future work will include continued pheasant surveys for 1 additional year, improved estimates of habitat abundance, and more complex analysis of the association between pheasant indices and habitat parameters. Final products of this project will include GIS habitat models or maps that managers can use to target habitat development efforts where they may yield the greatest increase in pheasant numbers.

INTRODUCTION

Preferred winter habitat for ring-necked pheasants (*Phasianus colchicus*) in the Midwest includes grasslands, wetlands, woody cover, and a dependable source of food (primarily grain) near cover (Gates and Hale 1974, Trautman 1982, Perkins et al. 1997, Gabbert et al. 1999). However, emergent wetlands and woody habitats that are large enough to provide shelter during severe winters have been extensively removed from agricultural landscapes, and grasslands and grain stubble are inundated by snow during some years. During severe winters, pheasants without access to sufficient winter habitat are presumed to perish or emigrate to landscapes with adequate habitat. Birds that emigrate >3.2 km (2 miles) from their breeding range are unlikely to return (Gates and Hale 1974).

Almost 400,000 ha (1 million acres) of cropland in Minnesota's pheasant range are currently retired under the Conservation Reserve Program (CRP). Wetland restorations, woody habitats, and food plots are eligible cover practices in the CRP, but most appear inadequate in size, design, or location to meet pheasant habitat needs. Furthermore, small woody covers commonly established on CRP lands may reduce the quality of adjacent grass reproductive habitat without providing intended winter cover benefits.

Pheasants use grasslands for nesting and brood rearing, and we previously documented a strong relationship between grassland abundance and pheasant numbers (Haroldson et al. 2006). However, information is lacking on how much winter habitat is needed to sustain pheasant populations during mild, moderate, and severe winters. The purpose of this study is to quantify the relationship between amount of winter habitat and pheasant abundance over a range of winter conditions. Our objectives are to: 1) estimate pheasant abundance on study areas with different amounts of reproductive cover, winter cover, and winter food over a time period capturing a range of winter severities (≥5 years); 2) describe annual changes in availability of winter cover as a function of winter severity; and 3) quantify the association between mean pheasant abundance (over all years) and amount of reproductive cover, winter cover, and winter food.

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METHODS

We selected 36 study areas of contrasting land cover in Minnesota's core pheasant range to ensure a wide range of habitat configurations. Study areas averaged 23 km^2 (9 $miles^2$) in size, and were selected to vary in the amount of winter cover, winter food, and reproductive cover. We defined winter cover as cattail (*Typha spp.)* wetlands ≥4 ha (10 acres) in area (excluding open water), dense shrub swamps ≥4 ha (10 acres) in area, or planted woody shelterbelts ≥0.8 ha (2 acres) in area, ≥60 m (200 feet) wide, and containing ≥2 rows of conifers (Gates and Hale 1974, Berner 2001). Winter food was defined as grain food plots left unharvested throughout the winter and located ≤0.4 km (1/4 mile) from winter cover (Gates and Hale 1974). Reproductive cover included all undisturbed grass cover ≥6 m (20 feet) wide. To facilitate pheasant surveys, 9 study areas were selected in each of 4 regions located near Marshall, Windom, Glenwood, and Faribault, Minnesota (Figure 1).

We estimated amounts of winter cover, winter food, and reproductive cover on each study area by cover mapping to a GIS from recent aerial photographs. In addition, we mapped large habitat patches within a 3.2-km (2-mile) buffer around study area boundaries to assess the potential for immigration to and emigration from study areas. We used Farm Service Agency's GIS coverages of farm fields (Common Land Units) as base maps, and edited field boundaries to meet the habitat criteria of this project. Cover types were verified by ground-truthing all habitat patches visible from roads. Because cover mapping of cattail wetlands, shrub swamps, and undisturbed grasslands is still in progress, for this progress report we made preliminary estimates of the amounts of these habitats from GIS coverages of the National Wetlands Inventory (NWI), Wildlife Management Areas (WMAs), Waterfowl Production Areas (WPAs), and CRP enrollments. We recognize that not all cattail wetlands, shrub swamps, and undisturbed grasslands are included in these GIS coverages.

We plan to estimate availability of winter cover during moderate–severe winters using aerial surveys. When fallen or drifted snow has inundated small (4–6 ha [10–15 acre]) cattail wetlands for ≥2 weeks, a sample of winter cover patches on all affected study areas will be inspected by helicopter to determine 1) availability of any remaining cover within the patch, and 2) presence of pheasants within the patch.

We estimated relative abundance of pheasant populations on each study area using roadside surveys (Haroldson et al. 2006). Roadside surveys consisted of 16–19 km (10–12 mile) routes primarily on gravel roads (≤6 km [4 miles] of hard-surface road). Observers drove each route starting at sunrise at an approximate speed of 24 km/hour (15 miles/hour) and recorded the number, sex, and age of pheasants observed. Surveys were repeated 10 times on each study area during spring (20 April–20 May) and summer (20 July–20 August). Surveys were conducted on mornings meeting standardized weather criteria (cloud cover <60%, winds ≤16 km/hour [10 miles/hour], temperature ≥0^{o}C [32^{o}F], dew present) 1–2 hours before sunrise; however, surveys were completed even if conditions deteriorated after the initial weather check. We attempted to survey all study areas within a region on the same days, and observers were systematically rotated among study areas to reduce the effect of observer bias.

Observers carried Global Positioning System (GPS) receivers while conducting roadside surveys to record their time and position throughout each survey (track logs), and to record the location of observed pheasants (waypoints). We inspected all track logs for each observer to ensure that surveys were conducted at the correct time, location, and speed of travel.

For each study area and season, we calculated a population index (pheasants counted/route) from the total number of pheasants counted/total survey distance driven over all 10 repetitions. We standardized the index to pheasants/161 km (pheasants/100 miles) to adjust for variation in survey distance among study areas. We evaluated temporal trends in pheasant abundance by calculating mean percent change in population indices by region and in total. We interpreted trends as statistically significant when 95% confidence intervals of percent change did not include 0.

To evaluate the effect of habitat on pheasant abundance, we calculated a cover index for each study area:

$$CI = [(UG/Max)x4 + (WCwFP/Max)x4 + (WCwoFP/Max)x2 + (FP/Max)] / 11$$

where UG = undisturbed grass (% of study area)
WCwFP = winter cover near a food plot (number of patches)
WCwoFP = winter cover without a nearby food plot (number of patches)
FP = food plot (number of patches)
Max = maximum observed value among all 36 study areas.

The cover index combined the effects of reproductive cover, winter cover, and winter food into a single weighted average (weight based on a preliminary estimate of relative importance). Potential values of cover index ranged from 0.0 (poorest habitat) to 1.0 (best habitat). We acknowledge that the cover index is an oversimplification, and we used it only to make simple, 2-dimentional plots for this early progress report. For each region, we evaluated the association of cover indices to pheasant population indices using simple linear regression.

RESULTS

We identified and mapped 321 patches of winter cover on the 36 study areas and surrounding 3.2-km (2-mile) buffers. Severity of winter weather was mild during all 4 winters (2002-2006) of this study. As a result, even the least robust patches of winter cover (e.g., 4-ha [10-acre] cattail wetlands) remained available to pheasants throughout the 4 winters of this study.

Spring 2006 Surveys

Observers completed all 360 scheduled surveys (10 repetitions on 36 study areas) during the spring 2006 season. Despite strong efforts by surveyors to select days that best met weather standards, weather conditions were not consistent among surveys, ranging from excellent (calm, clear sky, heavy dew) to poor (wind >16 km/hour [10 miles/hour], overcast sky, no dew, rain, or frost). Over all regions, 92% of the surveys were started with at least light dew present, which was greater than previous years (78-91%). Seventy-four percent of surveys were started under clear to partly cloudy skies (<60% cloud cover), 96% reported wind speeds <16 km/hour (10 miles/hour), and 98% of surveys were started on mornings with temperatures >0^{o}C (32^{o}F). Among regions, Glenwood experienced the least dew (16% of surveys started with no dew), whereas Faribault experienced the least cloud cover (only 7% of surveys started with cloud cover $\geq$60%).

Pheasants were observed on all 36 study areas during spring 2006, but abundance indices varied widely among areas from 32.5–474.5 pheasants observed per route (Table 1). Over all study areas, the mean pheasant index was 165.8 birds/route, a 69% increase (95% CI: 56–82%) from spring 2005 and the highest observed during the 4 years of this study (Table 2). Total pheasants/route varied among regions from 91.1 in the Faribault region to 234.3 in the Windom region (Table 2). Compared to 2005, total indices increased significantly in all regions, with the greatest increase in Marshall (101%; 95% CI: 76–126%) and the smallest increase in Windom (35%; 95% CI: 18–52%).

Hens were relatively abundant among study areas in spring 2006. The overall hen index averaged 97.5/route, a 95% increase (95% CI: 72–118%) from 2005 (Table 2). Among regions, the hen index ranged from 46.8/route in Faribault to 143.9/route near Windom. Hen indices increased significantly from 2005 in all regions, more than doubling in Faribault (132% increase; 95% CI: 56–208%) and Marshall (127% increase; 95% CI: 77–157%; Table 2). The observed hen:rooster ratio varied from 0.5 to 2.7 among study areas (Table 1). Fewer hens than roosters were observed on 1 study area in the Marshall region, 4 areas in Glenwood, and 4 areas in Faribault.

Summer 2006 Surveys

Observers completed all 360 scheduled surveys during the summer 2006 season. Weather conditions during the summer surveys ranged from excellent (calm, clear sky, heavy dew) to poor (light or no dew, overcast sky). Over all regions, 75% of the surveys were started with medium-heavy dew present, which was lower than 2005 (81%), 2004 (87%), and 2003 (81%). Large regional differences in dew conditions were observed this year, ranging from 90% of surveys with medium-heavy dew present in Marshall to only 62% in Windom and 66% in Glenwood. For all regions combined, 64% of the surveys were started under clear skies (<30% cloud cover), and 75% reported wind <6 km/hour (4 miles/hour). In comparison, 96% of the statewide August Roadside Surveys were started under medium-heavy dew conditions, 89% under clear skies, and 76% with winds <6 km/hour (4 miles/hour). The less desirable weather conditions reported in this study probably reflect the limited availability of 10 suitable survey days within the 31-day period.

Pheasants were observed on all 36 study areas during 2006, but abundance indices varied widely from 18.6–537.3 pheasants observed per route (Table 3). Over all study areas, the mean pheasant population index of 161.9 birds/route was not significantly different from 2005 (150.9 birds/route). Total pheasant indices varied among regions from 81.7 birds/route in the Faribault region to 280.9 birds/route in Marshall (Table 4). Compared to 2005, total indices increased significantly only in the Marshall region (Table 4).

The overall hen index (28.7 hens/route) was similar to last year (26.3 hens/route), and varied among regions from 12.2 in the Faribault region to 49.1 near Marshall (Table 4). Hen indices increased 60% (95% CI: 22–98%) in the Marshall region, but were not significantly higher than 2005 in the Glenwood, Faribault, or Windom regions (Table 4). In contrast, overall and regional cock indices increased significantly except in the Faribault region (Table 4). The observed hen:rooster ratio varied from 0.2 to 6.3 among study areas (Table 3), and averaged 1.8 overall. Fewer hens than roosters were observed on 2 study areas in the Glenwood and Faribault regions and 4 study areas in the Windom region.

The 2006 overall brood index (23.1 broods/route) was similar to 2005 (23.6 broods/route), with regional indices ranging from 11.4 in Faribault to 38.9 in Marshall (Table 4). Regional brood indices were also similar to 2005 (Table 4). Mean brood size averaged 4.8 chicks/brood overall, but varied among regions from 3.9 in Windom to 5.3 in Faribault. Mean brood size in 2006 increased 21% (95% CI: 1–41%) over that in 2005 in the Marshall region, was similar to 2005 in Glenwood and Faribault, and declined 12% (95% CI: -23 to -1%) in Windom (Table 4). On average, 27.9 broods were observed for every 100 hens counted during spring surveys, a 33% (95% CI: -45 to -21%) decrease from last year. This brood recruitment index (broods/100 spring hens) varied among regions from 18.7 in Windom to 35.9 in Marshall. Brood recruitment indices decreased significantly only in the Marshall (95% CI: -61 to -31%) and Faribault (95% CI: -61 to -31%) regions (Table 4).

Habitat Associations

The mean pheasant index (total pheasants/route averaged over summer 2003–2006) was significantly related to the cover index only in the Marshall region (Figure 2). Cover index explained 60% of the variation in pheasant indices in the Marshall region, 28% in Windom, 18% in Faribault, and 6% in Glenwood.

DISCUSSION

A high spring hen population in 2006 was expected given the relatively mild winter of 2005-2006 (the 5th consecutive mild winter), but the magnitude of the increase was greater than expected. Weather during the reproductive period was warmer and drier than average, conditions conducive for increased nest success and chick survival. However, brood size increased only in the Marshall region and the brood recruitment index (broods/100 spring hens) declined in 2006. Nevertheless, total pheasant indices remain high due to above-average carryover of adults from 2005 plus average chick recruitment in 2006.

At this early stage in our evaluation, we cannot explain the weak association between summer pheasant indices and habitat abundance on the Glenwood and Faribault study areas (Figure 2). However, preliminary habitat estimates based on GIS coverages of the NWI, WMAs, WPAs, and CRP enrollments appear to have omitted much more winter and reproductive cover on the Glenwood and Faribault study areas than on Marshall and Windom study areas. Habitat estimates will be improved as we finish cover mapping the study areas. In addition, future analyses of pheasant-habitat associations will use multiple regression models that treat reproductive cover, winter cover, and winter food as independent predictor variables.

Our study design requires at least 1 severe winter to estimate pheasant winter cover needs. After 4 consecutive mild winters, we have observed relatively high, stable pheasant populations on all study areas. We expect pheasant populations to decline following a severe winter, with the largest declines on study areas with the least amount of winter cover. Unless the coming winter (2006-2007) is severe, we may not be able to fully accomplish Objective 1 of this study. Furthermore, the significant loss of CRP contracts expected during 2007-2009 will preclude an extension of this study.

We plan to continue to survey pheasant populations during spring and summer 2007. In addition, we will continue annual cover mapping of all 36 study areas. During the next moderate-severe winter, we will assess winter habitat availability in relation to snow depth and drifting. Finally, we will attempt to build a multiple regression model using data extracted from a previous pheasant habitat study (Haroldson et al. 2006).

ACKNOWLEDGMENTS

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Table 1. Pheasant population indices and sex ratios (female:male) after 10 repeated surveys (n) on 36 study areas in Minnesota, spring 2006.

			Birds/route[a]			
Region	Study area	n	Total	Cocks	Hens	F:M ratio
Marshall	1	10	226.6	71.1	155.5	2.2
	2	10	267.5	121.3	146.3	1.2
	3	10	340.8	137.9	202.9	1.5
	4	10	362.5	106.2	256.3	2.4
	5	10	104.2	46.7	57.5	1.2
	6	10	232.1	75.0	157.1	2.1
	7	10	155.5	42.3	113.2	2.7
	8	10	128.7	68.3	60.4	0.9
	9	10	85.1	35.1	50.0	1.4
Glenwood	10	10	79.0	35.0	44.0	1.3
	11	10	81.4	24.6	56.8	2.3
	12	10	169.1	98.6	70.5	0.7
	13	10	114.8	64.3	50.4	0.8
	14	10	158.8	60.5	98.2	1.6
	15	10	215.7	108.8	106.9	1.0
	16	10	96.2	53.3	42.9	0.8
	17	10	37.2	19.8	17.4	0.9
	18	10	184.3	77.5	106.8	1.4
Windom	19	10	474.5	148.9	325.5	2.2
	20	10	396.3	157.6	238.7	1.5
	21	10	173.7	58.4	115.3	2.0
	22	10	219.0	103.2	115.8	1.1
	23	10	357.4	148.5	208.9	1.4
	24	10	136.0	64.0	72.0	1.1
	25	10	129.0	46.3	82.7	1.8
	26	10	166.7	61.4	105.3	1.7
	27	10	56.5	26.1	30.4	1.2
Faribault	28	10	207.5	79.2	128.3	1.6
	29	10	65.2	37.0	28.2	0.8
	30	10	58.9	40.3	18.5	0.5
	31	10	125.5	70.6	54.9	0.8
	32	10	68.5	33.8	34.7	1.0
	33	10	79.3	36.6	42.7	1.2
	34	10	131.6	61.0	70.7	1.2
	35	10	50.4	26.5	23.9	0.9
	36	10	32.5	13.3	19.2	1.4

[a]Route length standardized to 161 km (100 miles).

Table 2. Regional trends (% change) in pheasant population indices on 36 study areas in Minnesota, spring 2003–2006.

Region	Group	n	Birds/route[a] 2003	2004	2005	2006	% change 2005-2006	95% CI
Marshall	Total pheasants	9	87.2	116.3	110.4	211.4	101	±25
	Cocks	9	43.1	47.4	47.7	78.2	72	±27
	Hens	9	44.1	68.9	62.7	133.2	127	±30
Glenwood	Total pheasants	9	100.9	113.0	84.5	126.3	67	±24
	Cocks	9	48.7	47.2	40.2	60.3	55	±22
	Hens	9	52.2	65.9	44.3	66.0	86	±41
Windom	Total pheasants	9	162.3	179.7	167.6	234.3	35	±17
	Cocks	9	69.4	75.8	65.0	90.5	37	±17
	Hens	9	92.9	103.9	102.6	143.9	36	±24
Faribault	Total pheasants	9	70.3	86.0	57.3	91.1	72	±37
	Cocks	9	37.1	47.1	33.5	44.3	44	±30
	Hens	9	33.2	38.8	23.8	46.8	132	±76
All	Total pheasants	36	105.2	123.8	104.9	165.8	69	±13
	Cocks	36	49.6	54.4	46.6	68.3	52	±11
	Hens	36	55.6	69.4	58.3	97.5	95	±23

[a]Route length standardized to 161 km (100 miles).

Table 3. Pheasant population indices and sex ratios (female:male) after 10 repeated surveys (n) on 36 study areas in Minnesota, summer 2006.

Region	Study area	n	Birds/route[a] Total	Cocks	Hens	F:M ratio	Chicks/ route[a]	Broods/ route[a]	Chicks/ brood	Broods/100 Summer hens	Broods/100 Spring hens
Marshall	1	10	537.3	42.3	75.9	1.8	419.1	65.5	6.4	0.862	0.421
	2	10	421.7	46.7	64.2	1.4	310.8	52.5	5.9	0.818	0.359
	3	10	166.0	14.6	31.1	2.1	120.4	27.2	4.4	0.875	0.134
	4	10	313.0	18.5	61.5	3.3	233.0	51.0	4.6	0.829	0.199
	5	10	363.3	38.3	59.2	1.5	265.8	45.0	5.9	0.761	0.783
	6	10	267.9	18.9	53.8	2.9	195.3	35.8	5.4	0.667	0.228
	7	10	100.0	9.1	20.9	2.3	70.0	20.9	3.3	1.000	0.185
	8	10	253.5	35.1	50.0	1.4	168.3	38.6	4.4	0.772	0.639
	9	10	105.3	12.3	25.4	2.1	67.5	14.0	4.8	0.552	0.281
Glenwood	10	10	35.4	6.1	4.0	0.7	25.3	8.1	3.1	2.000	0.184
	11	10	152.1	7.2	22.5	3.1	122.3	19.5	6.3	0.868	0.344
	12	10	299.0	14.3	38.1	2.7	246.7	41.0	6.0	1.075	0.581
	13	10	107.8	7.8	23.5	3.0	76.5	15.7	4.9	0.667	0.310
	14	10	138.6	14.5	24.1	1.7	100.0	19.3	5.2	0.800	0.196
	15	10	197.2	28.2	35.6	1.3	133.3	33.3	4.0	0.935	0.312
	16	10	105.7	10.5	11.4	1.1	83.8	18.1	4.6	1.583	0.422
	17	10	24.8	5.0	4.1	0.8	15.7	2.5	6.3	0.600	0.143
	18	10	128.7	12.9	23.4	1.8	92.5	15.3	6.1	0.653	0.143
Windom	19	10	228.4	14.2	54.2	3.8	160.0	32.6	4.9	0.602	0.100
	20	10	172.0	19.7	43.5	2.2	108.8	28.0	3.9	0.643	0.117
	21	10	83.2	10.0	21.6	2.2	51.6	14.7	3.5	0.683	0.128
	22	10	151.6	28.0	40.6	1.5	83.0	21.7	3.8	0.533	0.187
	23	10	239.7	50.7	46.6	0.9	142.4	33.1	4.3	0.710	0.158
	24	10	76.0	27.5	14.5	0.5	34.0	11.0	3.1	0.759	0.153
	25	10	100.9	28.5	16.4	0.6	56.1	13.1	4.3	0.800	0.158
	26	10	281.6	41.2	52.6	1.3	187.7	44.7	4.2	0.850	0.425
	27	10	41.7	13.0	4.3	0.3	24.3	7.8	3.1	1.800	0.257
Faribault	28	10	127.4	9.0	20.3	2.3	98.1	18.9	5.2	0.930	0.147
	29	10	22.8	8.9	2.0	0.2	11.9	3.0	4.0	1.500	0.105
	30	10	77.4	6.5	11.3	1.8	59.7	11.3	5.3	1.000	0.609
	31	10	168.0	13.8	25.7	1.9	128.5	19.8	6.5	0.769	0.360
	32	10	77.5	6.8	16.7	2.5	54.1	11.7	4.6	0.703	0.338
	33	10	90.5	2.6	16.4	6.3	71.6	12.1	5.9	0.737	0.283
	34	10	118.4	11.0	11.8	1.1	95.6	18.4	5.2	1.556	0.261
	35	10	18.6	3.5	3.5	1.0	11.5	1.8	6.5	0.500	0.074
	36	10	35.0	8.3	2.5	0.3	24.2	5.8	4.1	2.333	0.304

[a]Route length standardized to 161 km (100 miles)

Table 4. Regional trends (% change) in pheasant population indices on 36 study areas in Minnesota, summer 2003–2006.

Region	Group	n	Birds/route[a] 2003	2004	2005	2006	% change 2005-2006	95% CI
Marshall	Total pheasants	9	142.6	114.9	190.5	280.9	54	±51
	Cocks		12.7	13.5	10.5	26.2	161	±107
	Hens		25.6	20.5	32.3	49.1	60	±38
	Broods		22.3	16.8	35.0	38.9	19	±34
	Chicks/brood		4.6	4.8	4.2	5.0	21	±20
	Broods/100 spring hens		59.9	29.8	77.2	35.9	–46	±15
Glenwood	Total pheasants	9	139.9	57.9	135.7	132.1	117	±189
	Cocks		9.2	8.3	8.0	11.8	73	±55
	Hens		23.5	12.3	20.7	20.8	8	±39
	Broods		20.2	8.3	17.2	19.2	30	±52
	Chicks/brood		5.0	4.1	6.1	5.2	–13	±19
	Broods/100 spring hens		44.7	14.7	42.8	29.3	–17	±38
Windom	Total pheasants	9	283.5	179.8	187.0	152.8	–5	±28
	Cocks		25.9	23.6	13.8	25.9	85	±43
	Hens		50.9	36.2	37.4	32.7	–5	±25
	Broods		36.2	24.2	29.4	23.0	–2	±36
	Chicks/brood		5.4	5.0	4.6	3.9	–12	±11
	Broods/100 spring hens		47.1	29.1	30.2	18.7	–20	±33
Faribault	Total pheasants	9	164.6	54.4	90.5	81.7	1	±32
	Cocks		9.5	13.0	8.0	7.8	4	±24
	Hens		23.6	13.1	14.8	12.2	–15	±24
	Broods		23.6	6.8	12.6	11.4	7	±36
	Chicks per brood		5.5	5.0	5.5	5.3	1	±18
	Broods/100 spring hens		85.4	18.6	71.0	27.6	–46	±15
All	Total pheasants	36	182.6	101.7	150.9	161.9	42	±46
	Cocks		14.3	14.6	10.1	17.9	81	±32
	Hens		30.9	20.5	26.3	28.7	12	±16
	Broods		25.6	14.0	23.6	23.1	13	±17
	Chicks/brood		5.1	4.7	5.1	4.8	–0	±8
	Broods/100 spring hens		59.3	23.1	55.3	27.9	–33	±12

[a]Route length standardized to 161 km (100 miles).

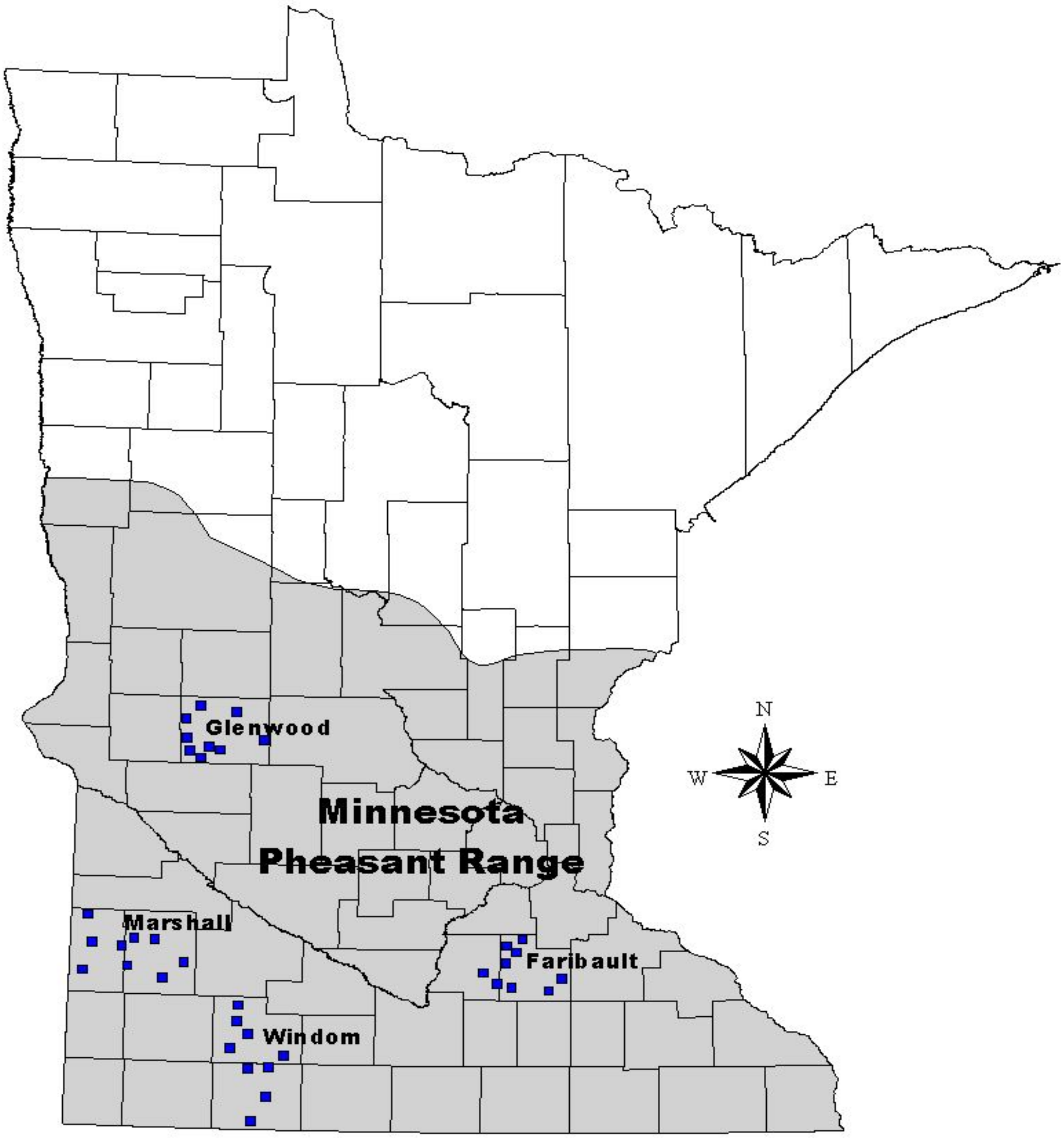

Figure 1. Locations of winter-habitat study areas within Minnesota's pheasant range, 2003-2006.

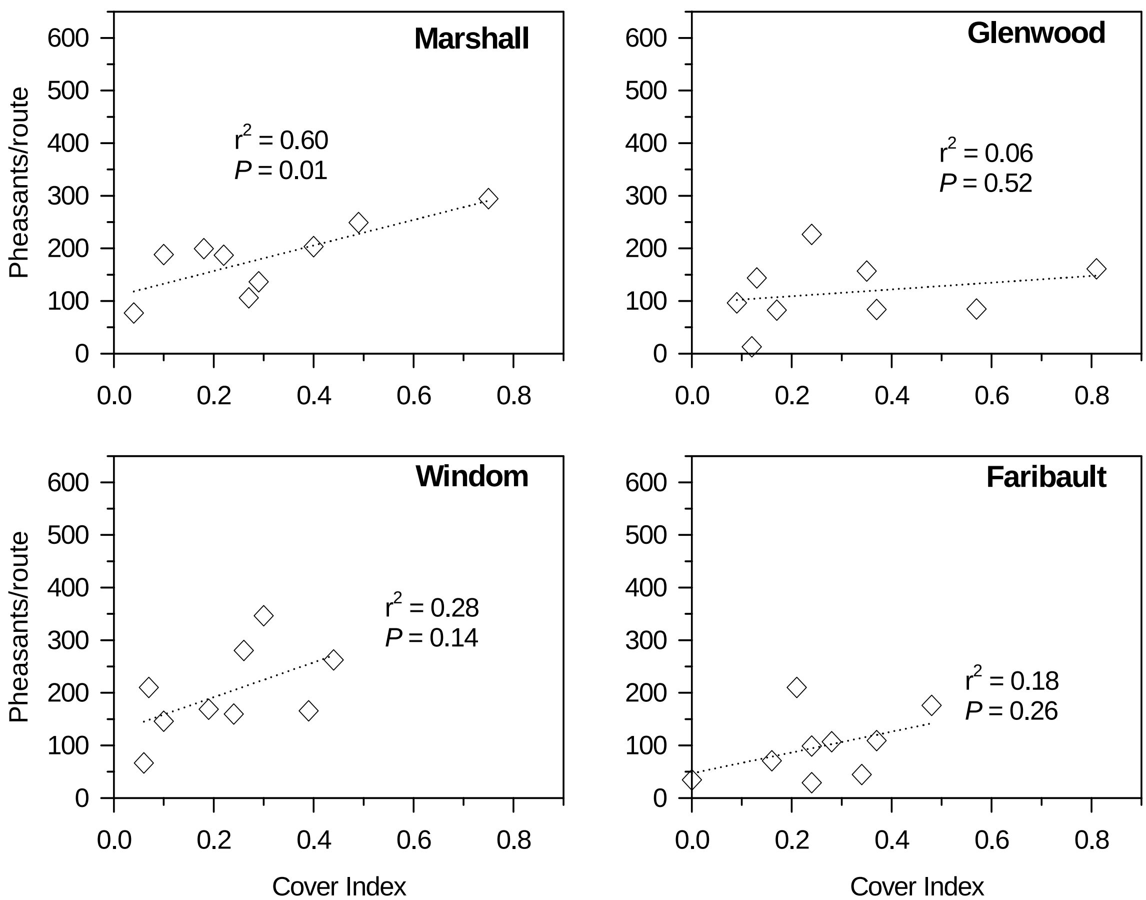

Figure 2. Relationship between relative pheasant abundance (pheasants counted/route) and amount of habitat (cover index) on 9 study areas in 4 regions in Minnesota during summer 2003-06. Route length was standardized to 161 km (100 miles).

SURVIVAL AND HABITAT USE OF EASTERN WILD TURKEYS TRANSPLANTED TO NORTHWESTERN MINNESOTA

Chad J. Parent, Brett J. Goodwin,[1] and Richard O. Kimmel

SUMMARY OF FINDINGS

Eastern wild turkeys (*Meleagris gallopavo sylvestris*) were not historically common in Minnesota. Public interest to restore extirpated populations to Minnesota generated an intensive trap-and-transplant program. Public demand for turkey populations is spreading northward, but our understanding of wild turkey ecology in northern habitat is inadequate. To address this, we released 59 female (radioed) and 19 male (not radioed) wild turkeys at two study areas in Red Lake and Pennington Counties, MN, USA. Locations were obtained on female turkeys 3-4 days/week in the winter (1 January 2006 to 31 March 2006) and 1-2 days/week the rest of the year (non-winter). We estimated survival, habitat use, home range, and productivity based on data in 2006. Overall survival was 22% (annual), 38% (winter), and 59% (non-winter). Cropland habitat had the most turkey locations (55%) followed by deciduous forests of oak, aspen, and white birch (27%), marsh (9%) and grassland (9%). Turkeys tended to stay close to farmsteads and rural residences with 65% of locations in Pennington County and 75% in Red Lake County found within 400 m of a farmstead. Twelve turkeys at the 2 study areas were located enough (≥ 20 locations) for home range analyses. We found that annual core home ranges were small compared to similar research: 168 ± 179 ha (mean ± SD) for Pennington County and 119 ± 58 ha (mean ± SD) for Red Lake County. Seven turkeys attempted to nest with 4 having successful clutches.

INTRODUCTION

In Minnesota, eastern wild turkeys were historically restricted to the southern part of the state with persistence of these populations dependent on winter severity (Leopold 1931). Public interest in northward expansion of turkeys in Minnesota has led to the establishment of sustainable populations as far north as Mahnomen and Norman counties in the northwest, and the St. Croix River valley south of Duluth (Figure 1).

Physiologically, turkeys should be able to survive in northern Minnesota if food is available (Haroldson 1996, Haroldson et al. 1998, Coup and Pekins 1999). Prince and Gray (1986) suggest that hens are capable of surviving 8 days without food. This is particularly important in northern Minnesota, as snowfall can cover food sources for extended periods. Snowfall deeper than 30 cm has been observed to abate turkey movement and make food hard for turkeys to find (Austin and DeGraff 1975). Finally, snowfall can also effect reproduction. Porter (1983) attributed severe winter conditions in southeastern Minnesota to reduced hatching success. It is unknown if translocated turkey populations are self-sustaining in northern Minnesota. The objectives of this study are to examine wild turkey survival, habitat use, and productivity during the first year following release in northern Minnesota.

METHODS

During winter 2006, Minnesota Department of Natural Resources (MNDNR) captured 59 female and 19 male wild turkeys using cannon nets at sites in southeastern Minnesota. The turkeys were weighed, aged (juvenile or adult), and leg-banded. Female turkeys were equipped with a backpack style radio-transmitter (95 - 104 g, 40 cm whip antenna) with a battery life of

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approximately 3 years and a movement (mortality) sensitive switch (Advanced Telemetry Systems-ATS, Isanti, MN, USA). Males were not radioed because of their higher resilience to severe weather (Gray and Prince 1988). Within 2 days of capture, turkeys were transported to study areas in Red Lake County and Pennington County, MN and released. Both study areas are located in the Aspen Parkland Ecological Classification System subsection (MNDNR 2006). The landscape is composed of lacustrine plain and historic beach ridges formed by Glacial Lake Agassiz (MNDNR 2006). Based on level III GAP land cover classification data, both study areas were approximately 82% cropland with the remaining 18% of the study areas composed of nearly equal amounts of grassland, oak forest, aspen or white birch forests, and marshes (Table 2).

Monitoring of Turkeys

Radioed birds were monitored 3-4 times/week during winter and 1-2 times/week during non-winter. Winter was defined as 1 January through 31 March (Kassube 2006). Wild turkeys were located via triangulation from roads using ≥3 azimuths acquired within 15 minutes (Hubbard 1999). Attempts to keep triangulation angles within 45 to 135 degrees were carried out whenever possible. Due to the lack of roads at the Red Lake study area, this was not always feasible.

Mortality date was assumed to be the midpoint of the last known date the turkey was alive and the date of the first mortality signal. During the nesting season a mortality signal was assumed to be an incubating hen; a follow up was conducted 30 days from the original mortality signal or when the hen left the nesting site. Efforts were made to retrieve the radio and examine the bird as soon as was possible after a mortality signal. Upon recovering the radio and dead bird, an investigation to determine the probable cause of mortality was conducted, i.e. feathers, hair, tracks, carcass condition, marks on radio (Thogmartin and Schaeffer 2000). If a turkey carcass was recovered we examined crop contents.

Survival

We calculated annual (1 January through 31 December), winter (1 January through 31 March), and non-winter (1 April through 31 December) survival rates. Hens that died < 7 days post-release were censored from the analyses because of potential trapping stress or transmitter harness complications (Vangilder 1996, Miller et al. 1998). To investigate impacts of weather conditions on mortality rates, weather data were extracted from the Minnesota Climatology Working Group (MCWG 2006) weather stations. For the Red Lake County study area the weather station was located in the same township as the release site. For the Pennington County study area the weather station is located in a neighboring township.

Habitat Use

Triangulation data were converted to spatial data using Location Of A Signal (LOAS, Ecological Software Solutions). Habitat associated with each individual turkey location from 2006 (n=321) was determined from level III GAP land cover data downloaded from the MDNR Data Deli (deli.dnr.state.mn.us). These associations provide an estimate of turkey habitat use. As we had no *a priori* study area, we used the turkey locations most distant from the release sites (9 km) to determine the extent of our two study areas. Study areas were the area contained in a 9 km radius circle around the release point (26,890 ha each). From the study area, we calculated habitat availability, which was compared to our estimate of habitat use. To estimate farmstead use, we counted turkey locations that fell within 400 m of a farmstead or rural residence.

Home Range

Locations from LOAS were examined in BIOTAS (Ecological Software Solutions) to estimate turkey home ranges at the two study areas. Home ranges were estimated from the complete year's locations and also from seasonal (winter, non-winter) subsets of the locations. A fixed kernel was applied to the locations and yielded 95, 75, and 50 percent confidence regions. Only turkeys with ≥20 locations were considered for analysis. Twelve turkeys were included in home range analyses, with 6 turkeys at each study area.

Productivity

To prevent disturbance of nesting females, mortality signals during the nesting season were treated as nesting attempts. The number of poults per hen were estimated from personal observations and reports by landowners of young turkeys in the field after poults had fledged.

RESULTS

Survival

In 2006, 14 of 59 (23.7%) hens were censored from the study due to early mortality, which reduced the sample size to 45 hens. In 2006, 6 turkeys at the Pennington County study area and 4 turkeys at the Red Lake County study area survived into the next year (2007). Annual survival was 27% and 17% respectively at each study site and 22% overall (Table 1). Overall winter survival was 38%, 36% at the Pennington County study area and 39% at the Red Lake County study area (Table 1). Overall non-winter survival was 59%; 75% at the Pennington County study area and 44% at the Red Lake County study area (Table 1). Mortality at both study areas increased with snowfall and low temperatures (Figure 2). Substantial snowmelt and warming occurred in April with a concurrent decrease in mortality. Scavenging prevented us from identifying some mortalities. Sources of mortality were attributed to avian (6%) or mammalian (22%) predation, vehicle collision (2%), severe weather (4%), and unknown (66%).

Habitat Use

Turkeys were found most often in cropland, followed by oak forests, marsh, aspen and birch forests, and grassland (Table 2). Turkeys were found in cropland in 52.6% of locations. Cropland made up 82.3% of the study area. In contrast, 33% of locations were in oak and marsh habitats, which make up only 6% of the study areas.

Sixty-five percent of the locations at the Pennington County study area and 75% of the locations at the Red Lake County study area were located within 400 m of farmsteads and rural residences. Farmsteads and rural residences, along with a 400 m buffer, comprise 261 ha (0.9% of study area) of the Pennington County study area and 729 ha (2.7% of study area) of the Red Lake County study area.

Home Range

Core home range (50% confidence region; Gitzen et al. 2006) in Pennington County was 168 ha ± 179 (mean ± St. Dev) for the whole year, 136 ha ± 92 during the winter, and 316 ha ± 420 during the non-winter period (Table 3). Home ranges increased 180 ha during non-winter periods, and were at their peak size during this period.

Core home range in Red Lake County was 119 ha ± 58 annually, 118 ± 133 during the winter, and 120 ± 133 during non-winter periods (Table 4). Home range increased 2 ha during the non-winter period.

Productivity

During 2006, 5 adults and 2 juveniles nested (4 hens at the Red Lake County study area and 3 at the Pennington County study area). We assume hatching occurred between 19 June 2006 and 29 June 2006. Of the hens that nested, 57% (4/7) appeared to have been successful. Our observations indicate that 13 poults were hatched at the Pennington County study area and 9 at Red Lake County study area.

DISCUSSION

Survival

Severe winter conditions can reduce wild turkey survival (Wunz and Hayden 1975, Porter et al. 1983, Haroldson et al. 1998). In 2006, we assume that handling stress and unfamiliarity with surroundings contributed to winter mortality observed in this study. Winter mortality was linked to weather conditions; as temperature increased and snow cover decreased, turkey mortality declined (Figure 2). An increase in survival during spring is not common, but changes in survival rates between winter and spring have been reported to be negligible (Porter 1988, Roberts et al 1995, Wright et al 1996). However, in Ontario, Nguyen et al (2003) observed increased survival during the spring. Non-winter mortality was predominately due to predation. Scavenged carcasses indicated mortality was likely due to avian or mammalian predators.

Nguyen et al (2003) observed 28% survival during the first year of a release in Ontario. Kane (2003) observed 22% annual survival in central Minnesota during the first year of a release coinciding with mild winter conditions. Our estimates of survival were similar with 22% annual survival, which may in part be due to 2006 having a mild winter. It remains to be seen what survival rates will be like during a winter with colder temperatures and more snow remaining on the ground for longer periods.

Habitat Use

Most turkey locations occurred in croplands. However, it is unclear if turkeys are selecting croplands or simply moving through them because they are so abundant on the landscape. During winter, it is unlikely that cropland was a preferred habitat since snow depth was >30 cm (MCWG 2006) at both study areas. At this snow depth, unless snow was blown from fields, turkey movements would be slowed and it would be difficult for turkeys to find food left on the ground (Austin and DeGraff 1975). Finally, interpretation of the location data as an indicator of turkey habitat selection is complicated by turkey flocking behavior, since data might be inflated because each member of the flock would be counted in a particular habitat type.

At the Pennington County study area, oak forest and marsh habitats emerge as important habitat types. This conclusion is based on the large proportion of locations in these habitat types compared to the low percent of the study area that are oak forest or marsh. We expected turkeys to use oak forest because acorns are an important food source (Palmer et al 1969). Based on incidental observations, acorns were abundant in 2006. We suspect that turkeys may have used marsh habitat as cover from predators and nesting habitat (e.g., Lazarus and Porter 1985).

At the Red Lake County study area, grassland and marsh habitat locations were used by turkeys. We suggest turkeys may have selected these habitats for nesting or for food. Grasslands consisting of alfalfa and grains were used by turkeys in Wisconsin (Paisley and Kubisiak 1994). Lazarus and Porter (1985) identified mesic plant communities (i.e. marsh) as nesting sites by turkeys in southern Minnesota. Marsh habitat was used for nesting by 1 turkey at the Red Lake study area.

In this study, hen turkeys used farmsteads and rural residences at a high rate; especially considering that farmsteads made up a small proportion of the 2 study areas. Most farmsteads in the study areas have ranching or agriculture, which could provide a consistent source of food. Crop contents from turkeys during winter (n = 6) included corn, suggesting some possible feeding on stored grains. Crop depredation by wild turkeys is a concern when they use farmstead habitats (Paisley and Kubisiak 1994). Public acceptance of future wild turkey releases in northern Minnesota will likely be influenced by farmstead use by turkeys.

Home Range

While a number of studies have estimated turkey home range size (e.g., Lewis 1963, Porter 1977, Brown 1980) most are conducted in areas quite different from northern Minnesota. Studies in Minnesota include Porter (1978, 1980) in southeastern Minnesota and McMahon and Johnson (1980, 1982) in east-central Minnesota. All except Porter (1978) reported home ranges larger than those we observed. Porter (1980) and McMahon and Johnson (1980, 1981) reported larger mean home range sizes (year long mean home range of 100 ha, winter mean home range of 750 ha, and winter mean home range of 596 ha respectively). Only Porter (1978) reported smaller mean home range sizes (100 ha). An explanation for the differences could be our use of kernel estimators, while the other studies used Minimum Convex Polygon. Minimum Convex Polygon estimates are known to be larger than kernel estimates (Aebischer et al 1993). Additionally, both our study areas have a higher concentration of agriculture or ranching landuse than the other studies. We have shown that turkeys tend to be found near farmsteads likely due to availability of food or shelter. As a result, turkeys would not need to move far between food and shelter resulting in smaller home ranges.

Nesting and Recruitment

Nesting success in our study was lower than Porter (1978). Because our turkeys were released between January and March, it is possible that an unfamiliarity of the area could result in fewer nesting attempts.

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Table 1. Annual and seasonal survival of wild turkeys at the Red Lake County and Pennington County study areas, Minnesota, 19 January 2006 through 30 October 2006.

	Annual	Winter[1]	Non-Winter[2]
Pennington	(6/22) 27%	(8/22) 36%	(6/8) 75%
Red Lake	(4/23) 17%	(9/23) 39%	(4/9) 44%
Overall	(10/45) 22%	(17/45) 38%	(10/17) 59%

[1] 1 January – 31 March
[2] 1 April – 31 December

Table 2. Proportion of habitat types used by wild turkeys, Minnesota, 2006.

	Pennington County		Red Lake County	
Habitat type	Percent of study area	Percent of loc. in habitat	Percent of study area	Percent of loc. in habitat
Cropland	82.3	52.6	81.9	56.2
Grassland	5.9	6.1	5.9	23.1
Aspen or White Birch	4.3	7.6	2.7	3.1
Oak	3.3	17.7	4.2	5.8
Marsh	2.9	15.2	2.5	11.6
Lowland Shrub	0.8	0	0.1	0
Upland Shrub	0.6	0	0.6	0
Aquatic	0.4	0	0	0
Black Ash	0.3	0.74	0.2	0
Developed	0	0	1.5	0

Table 3. Fixed-kernel home ranges (hectares) for wild turkeys on the Pennington County study area, with 50, 75, and 95% confidence regions for 6 turkeys with ≥ 20 locations, Minnesota, 2006.

	50%		75%		95%	
	Mean	St. Dev	Mean	St. Dev	Mean	St. Dev
Annual	168	179	511	543	1268	1258
Winter	136	92	295	182	695	368
Non-Winter	316	420	731	909	1564	1800

Table 4. Fixed-kernel home ranges (hectares) for wild turkeys on the Red Lake County study area, with 50, 75, and 95% confidence regions for 6 turkeys with ≥ 20 locations, Minnesota, 2006.

	50%		75%		95%	
	Mean	St. Dev	Mean	St. Dev	Mean	St. Dev
Annual	119	58	257	206	941	682
Winter	118	133	256	291	912	1028
Non-Winter	120	133	381	457	1139	1287

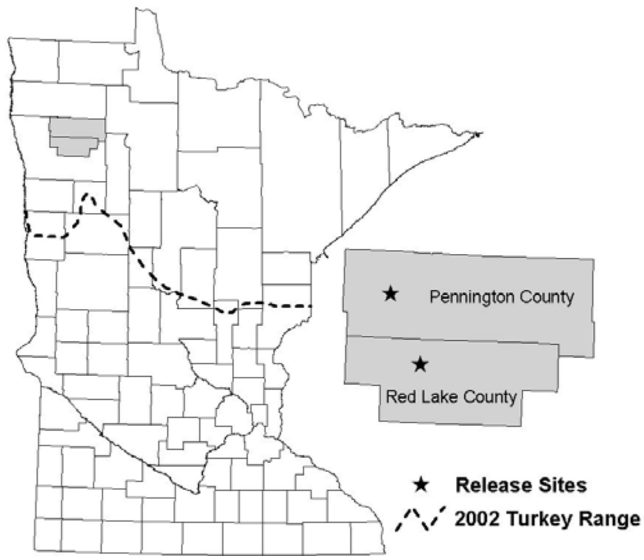

Figure 1. Wild turkey release sites (study areas) in Red Lake and Pennington County, Minnesota in 2006 and the northern range of turkeys in Minnesota in 2002.

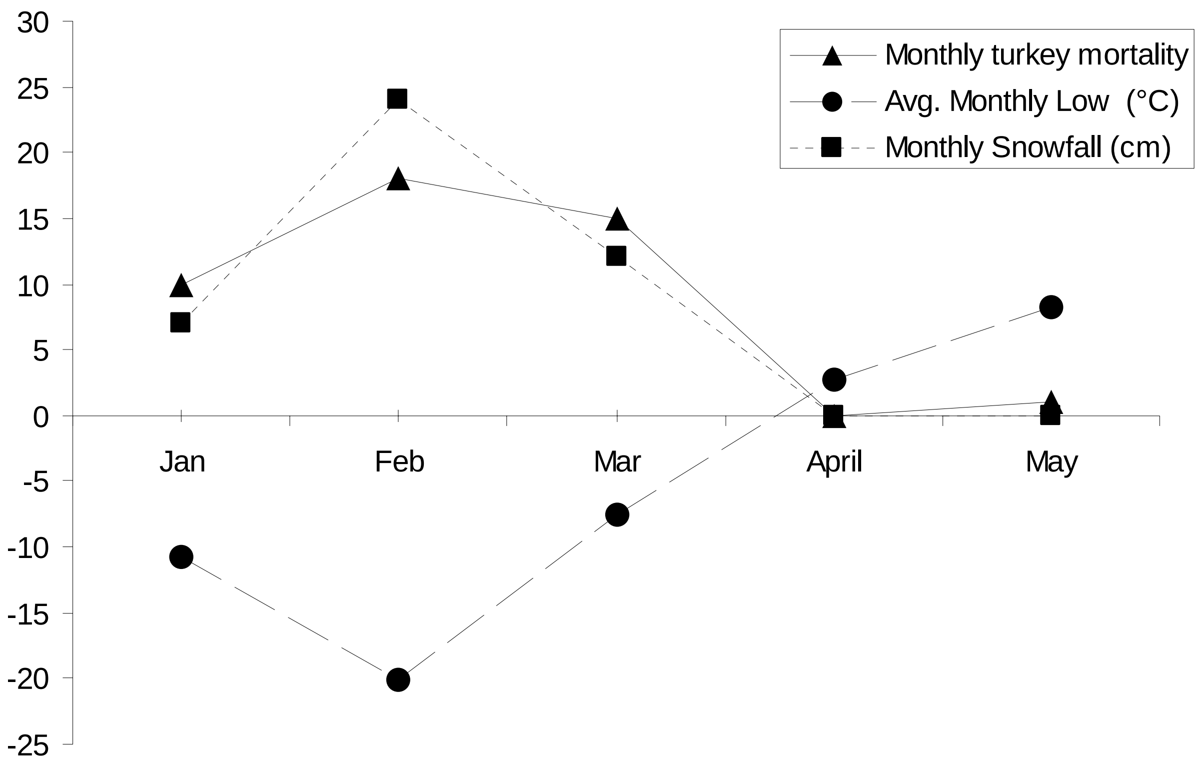

Figure 2. Monthly turkey mortality (%), total monthly snowfall (cm), and average monthly low temperatures (°C), Minnesota, 2006.

MONITORING VEGETATION TO ASSESS CHANGES IN RELATION TO WHITE-TAILED DEER DENSITIES

Emily J. Dunbar and Marrett D. Grund

SUMMARY OF FINDINGS

High densities of white-tailed deer (*Odocoileus virginianus*) result in overbrowsing of forest vegetation. Intensive browsing can change forest ecosystem structure and composition by reducing palatable plant species and increasing unpalatable plant species. Past studies have examined differences in forest vegetation using exclosures between areas with no deer and high densities of deer. Few studies have investigated impacts of forest composition and structure by different or declining densities of deer. This study will examine impacts of declining deer density on forest vegetation at Itasca State Park. This report summarizes the first 2 years of data collection (2005-2006). Three plot arrays were established and sampled in 2005. Seven more plot arrays were added in 2006 and 10 arrays were sampled during summer and will be resampled in future years. Most plot arrays at Itasca State Park were unique in composition. Thus, results should not be compared among sites, but over time within each plot array. Overall, plot arrays were not highly diverse, averaging 2.3 using the Shannon-Weiner Index (0-5). Density and frequency of plant species was fairly low, with many species occurring in small numbers. Herbaceous reproduction was observed infrequently, although reproducing plants were taller on average than non-reproducing plants. Browsing mainly occurred on woody species rather than herbaceous species. Time series analyses will be used in future reports to determine changes in forest vegetation over time in individual plot arrays.

INTRODUCTION

In recent years, white-tailed deer populations reached high densities in many areas of Minnesota. Overabundant deer generate a variety of problems for both humans and forest ecosystems (Cote et al. 2004). Intensive deer browsing, resulting in reduced regeneration or even exclusion of some plant species, directly affects the distribution and richness of both understory and overstory forest vegetation (Rooney 2001) and could impact Minnesota's sustainable forest management certification. Alterations in plant populations may lead to a variety of changes in community structure including increased populations of unpalatable or browse-resilient species, the elimination of preferred woody and herbaceous species, and a decrease in resources for other wildlife (Horsley et al. 2003, Rooney and Waller 2003). Over time, intensive deer browsing can cause a forest ecosystem to succeed to an alternate state, characterized by unpalatable tree species and a ground layer of ferns, grasses, and sedges (Horsley et al. 2003).

Past studies examined differences in forest community structure between no deer inside exclosures and high densities of deer outside exclosures (Wisdom et al. 2006). Few studies investigated changes in forest structure with differing deer densities (Horsley et al. 2003, Tremblay et al. 2006) or declining deer densities. This study will assess impacts of deer browsing on forest vegetation and changes in vegetation due to a declining deer population in Itasca State Park in northwestern Minnesota. In 2005, Itasca State Park was selected as a study area for an alternative deer management research project. Antler-point restriction regulations were implemented during the regular firearms season. Alternative deer management was proposed to reduce deer densities. Our goal was to measure and monitor ecosystem-level effects caused by overabundant deer at Itasca State Park. Our secondary goal was to develop a forest vegetation monitoring protocol that could be used in other areas of Minnesota.

OBJECTIVES

- To determine the impacts of deer browsing at Itasca State Park;
- To assess changes in forest vegetation due to a declining deer population at Itasca State Park; and
- Develop a forest vegetation monitoring protocol for use in Minnesota.

METHODS

Vegetation sampling was conducted at Itasca State Park, in northwestern Minnesota, during July 2005 and 2006. A 16 x 16 grid was placed in the center of the park using Geographical Information Systems (GIS). Three plot arrays were selected in 2005 and 7 additional plot arrays were selected in 2006 (Figure 1) using a random number generator. Thus, we collected data from 10 plot arrays in 2006. Each sampling plot array contained a 50 x 50m (2500-m^2) plot and 5, 1-m^2 subplots. Plots were permanently marked with 0.6-m pieces of rebar at the center, at the corners of the 2500-m^2 sampling plot, and at a pair of diagonal corners of each 1-m^2 subplot (Figure 2).

Data were recorded from each 1-m^2 subplot and 2 m radius plot at the corners of the 2500-m^2 plot, and transects originating at each subplot. We intend to collect data annually at the arrays for at least 5 years. In each 1-m^2 subplot, all woody and herbaceous species (< 2.54 dbh and < 1.5-m tall) were identified and counted. Percent cover of each plant species was recorded using Daubenmire cover classes (Daubenmire 1959). Heights of woody or herbaceous plants were also recorded. We also recorded percent cover of bryophytes and lichens, tree seedlings, rock, and litter. Litter depth was measured and recorded using a meter stick at the center of each subplot.

Photographs were taken above each subplot and also in each cardinal direction to measure forest structure. At each corner of the 2500-m^2 plot, all trees and shrubs (> 1.5-m tall and/or between 2.54 and 12.7 cm dbh) within a 2-m radius of the permanent marker were identified to species, and height and dbh recorded. Percent overstory canopy was estimated using a spherical densitometer at the centers of subplots and a Graphical Resource Solutions densitometer (GRS) at 5, 5-m intervals along transects in each cardinal direction from subplot centers.

Slope, aspect, topographic position, and visual evidence of natural disturbance history (fire scars, insect/disease infestation, blow downs, etc.) were recorded for each sampling plot array. Abiotic differences can lead to differing plant compositions and subsequently, vary deer usage within forest ecosystems. If abiotic differences exist between the plot arrays, results will be compared on an individual array basis, rather than across arrays. To determine if the plot arrays were similar in plant species composition, Renkonen Similarity Index (RSI) was used. This index is robust in regards to sample size and species diversity and is one of the top quantitative similarity coefficients available to ecologists (Wolda 1981). The index ranges from zero (no similarity) to 100 (complete similarity) (Wolda 1981). The index was calculated by transforming number of plants for each species into percentages, using the following formula;

$$P = \Sigma_i \text{ minimum } (p_{1i}, p_{2i})$$

where P = Percentage similarity between sample 1 and 2

p_{1i} = Percentage of species *i* in community sample 1

p_{2i} = Percentage of species *i* in community sample 2

RSI was calculated using the subplots, 2-m radius plots, and overstory canopy along transects.

Plot arrays in 2005 and 2006 were measured for diversity using Shannon-Wiener function. Shannon-Wiener index is sensitive to changes in rare species in a community and ranges from zero (no diversity) to 5 (high diversity) (Peet 1974). The index was calculated using the following formula:

$$H' = -\Sigma_{i=1} (p_i)(\log_2 p_i)$$

where H' = Index of species diversity

p_i = Proportion of total sample belonging to ith species

Shannon-Wiener function of diversity was calculated using the subplots, 2-m radius plots, and overstory canopy along transects.

Density and frequency of plant species were calculated in the subplots and 2-m radius plots for both years. Frequency of the overstory canopy plant species was also recorded in both years. Estimates of forest horizontal cover were obtained in each subplot using a cover board. Plant reproduction was sampled in subplots by the presence/absence of flowers or fruit of each plant (i.e. Canada mayflower (*Maianthemum canadense*). Browsing intensity was recorded for each plant in subplots and 2-m radius plots. Browsing intensity was ranked based on percent of stems browsed and height of plant:

1. Not Browsed – no visible browsing damage
2. Light – 0 to 25% of seedling stems are browsed
3. Moderate – 25 - 50% of stems are browsed
4. Heavy - more than 50% of stems are browsed and the plant is severely hedged, but it is taller than 15 cm
5. Severely browsed – no seedlings of the species within the plot are >15 cm tall and seedlings are severely hedged

RESULTS AND DISCUSSION

A total of 42 plant species were recorded and 949 individual plants were sampled in 3 plot arrays in 2005 In 2006, 71 plant species were recorded and 3,515 individual plants were sampled in the 10 plot arrays. Overall, 2006 RSI scores ranged from dissimilar (5) to somewhat similar (68). The mean RSI score was 30, suggesting there was little similarity among plot arrays (Table 1). In 2006, the most similar subplots were in plot arrays 3 and 10 (Table 1). The plant species composition within 2-m radius plots in 2005 was dissimilar. The similarity of the plant composition of the 2-m radius plots in 2006 ranged from very dissimilar (0) to highly similar (95). In 2006, the similarity of plant composition in the overstory canopy, recorded from transects of each subplot, ranged from 8 to 76. The most similar plot arrays with regards to overstory canopy were arrays 2 and 10. Sample sizes of overstory canopy data were too low in 2005 to calculate similarity.

Average Shannon-Weiner diversity score of plot arrays in 2006 was 2.31, which indicates moderate vegetative diversity. Plot array 5 was most diverse (2.85) and the least diverse plot array was 7 (1.61). Average Shannon-Weiner Index scores associated with subplots within plot arrays was 3.49 (range 2.78 – 3.91) in 2005 and 3.63 (range 1.93 – 4.25) in 2006. The average diversity of the 2-m radius plots within plot arrays during 2005 was 1.48 (range 0.88 – 2.00) and 1.10 (range 0.36 – 2.12) in 2006. The average diversity of the overstory canopy in plot arrays in 2006 was 2.19 (range 1.1 - 3.16). Sample sizes were too low in 2005 to calculate overstory diversity.

In 2005, Canada mayflower had the highest density (4.9 stems/m^2) in the subplots and the average plant density among subplots was 1.1 stems/m^2 (Table 2). Similar to 2005, we found that Canada mayflower had the highest density in subplots (6.4 stems/m^2) and the average plant density was 0.69 stems/m^2 in 2006 (Table 2). The most frequently observed plant species in the subplots in 2005 was sedge (*Carex* spp.). In 2006, the most frequently observed species in the subplots was mountain ricegrass (*Oryzopsis asperfolia*) (Table 2). In the 2005 2-m radius plots, sugar maples (*Acer saccharum*) had highest density (1,393 stems/ha) and the average plant density in 2005 was 517 stems/ha (Table 3). The most frequently observed species was beaked hazelnut (*Corylus cornuta*) (Table 3). In the 2006 2-m radius plots beaked hazelnut had the highest density (1,971 stems/ha) the average density in 2006 was 513 stems/ha (Table 3). The most frequently encountered species in 2005 was ironwood (*Ostrya virginiana*) (Table 3). In overstory canopy, red pines (*Pinus resinosa*) were the most frequently sampled species in 2005 (Table 3). In 2006, the most frequently observed species in the overstory was aspen (*Populus* spp*.)*.

In 2005, the average horizontal cover was 53%. Plot array 1 had the highest horizontal cover (80%) and plot array 3 had the lowest horizontal cover (29%) (Table 1). In 2006, the average horizontal cover was also 53%. Plot array 6 had the highest horizontal cover (90%) and plot array 7 had the lowest horizontal cover (19%) (Table 1).

Plant reproduction was sampled in subplots by the presence/absence of flowers or fruit of each plant. In 2005, 5 plant species had plants that were in the reproductive stage; big-leaf aster (*Aster macrophyllus)*, downy yellow violet (*Viola pubescens)*, early meadow-rue (*Thalictrum dioicum)*, large-flowered bellwort (*Uvularia grandiflora*), and twisted stalk (*Streptopus lanceolatus*) (Table 4). Four of the 5 species had low flowering sample sizes ($n \leq 2$). The average height of bellwort flowering plants was 38.1 cm (n = 15) and the average height of non-flowering plants was 21.8 cm (n = 26) (Table 4). In 2006, 5 plant species had plants that were in the reproductive stage; bluebead lily (*Clintonia borealis*), Canada mayflower, large-flowered bellwort, jewelweed (*Impatiens capensis*), and twisted stalk (Table 4). Three of the 5 species had low flowering sample sizes ($n \leq 3$). The average height of flowering bellwort plants was 37.7 cm (n = 13) and non-flowering plants was 27.5 cm (n = 161) (Table 4). The average height of flowering Canada mayflower plants was 8.5 cm (n = 16) and non-flowering plants was 5.0 cm (n = 302) (Table 4).

Browsing intensity was measured in subplots and sapling plots. In 2005, we found that most browsing was concentrated on tree seedlings in the subplots. Most species browsed had small sample sizes (n<10) or had low browse intensity (<2.0). Species browsed that had browsing intensity greater than 2.0 included mountain maple (*A. spicatum)* and ironwood (Table 5). Mountain maple had an average browsing intensity of 2.6 (n=6) and ironwood had an average browsing intensity of 2.7 (n=7). In the 2-m radius plot, no species had an average browsing intensity >1.1. In 2006, species browsed in subplots that had higher browsing intensities ($\geq$ 2.0) included aspen and choke cherry (*Prunus virginiana*) (Table 5). Aspen had an average browsing intensity of 2.9 (n=28) and choke cherry had an average browsing intensity of 2.0 (n=4) (Table 5). In the 2-m radius plots, mountain maple, choke cherry, and red elm (*Ulmus rubra*) had browsing intensities > 2.0 (Table 5). Mountain maple had an average browsing intensity of 2.1 (n=20), choke cherry had a browsing intensity of 3.0 (n=1), and red elm had an average browsing intensity of 2.0 (n=2) (Table 5).

Due to time constraints, we reported frequency data and other descriptive statistics available to summarize the data collection thus far. Time series models will be used in future analyses to determine changes in forest vegetation and to account for differences in plant composition between plot arrays. We believe times series analysis models will facilitate determining “indicator” plant species that may increase in abundance and distribution under lower deer densities.

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Table 1. Similarity and diversity of plots sampled at Itasca State Park, Minnesota, July 2005, 2006.

Year	Plot array	Area sampled	RSI score[a]	Diversity	% Horizontal[b] cover
2005	1	Subplot	28 (2)	3.91	80
		2 m radius	20 (2)	2	
	2	Subplot	31 (3)	2.78	50
		2 m radius	14 (3)	1.57	
	3	Subplot	46 (1)	3.79	29
		2 m radius	32 (1)	0.88	
2006	1	Subplot	64 (8)	3.97	82
		2 m radius	72 (10)	1.55	
		Transects	47 (8)	2.65	
	2	Subplot	46 (7)	3.19	35
		2 m radius	90 (7)	1.37	
		Transects	76 (10)	1.9	
	3	Subplot	71 (10)	3.64	35
		2 m radius	19 (9)	0.84	
		Transects	53 (5)	2.58	
	4	Subplot	49 (6)	4.25	62
		2 m radius	95 (8)	0.81	
		Transects	66 (9)	2.35	
	5	Subplot	54 (1,3)	3.9	46
		2 m radius	57 (2)	1.5	
		Transects	53 (3)	3.16	
	6	Subplot	60 (1)	3.66	90
		2 m radius	80 (8)	0.83	
		Transects	47 (2)	1.37	
	7	Subplot	46 (2)	1.93	19
		2 m radius	90 (2)	0.88	
		Transects	47 (5)	2.01	
	8	Subplot	64 (1)	4.21	53
		2 m radius	95 (4)	0.72	
		Transects	61 (4)	2.5	
	9	Subplot	49 (1)	3.95	40
		2 m radius	43 (1)	2.12	
		Transects	66 (4)	2.32	
	10	Subplot	71 (3)	3.56	69
		2 m radius	72 (1)	0.36	
		Transects	76 (10)	1.1	

[a] RSI score = highest score for the area sampled and corresponding plot array
[b] % Horizontal cover = average cover for plot array

Table 2. Density and frequency of plant species sampled in subplots at Itasca State Park, Minnesota, July 2005, 2006.

		2005		2005	
Species	Common name	Density[a]	Frequency[b]	Density[a]	Frequency[b]
Acer rubrum	Red maple	0.93	0.53	0.92	0.20
Acer saccharum	Sugar maple	3.80	0.53	4.42	0.40
Acer spicatum	Mountain maple	0.40	0.13	0.58	0.22
Actaea rubra	Red baneberry			0.02	0.02
Amelanchier	Juneberry spp.			0.12	0.06
Amphicarpa bracteata	Hog-peanut	0.13	0.07	0.78	0.14
Anemone canadensis	Canada anemone			0.02	0.02
Anemone cylindrica	Thimbleweed				
Anemone quinquefolia	Wood anemone	0.13	0.07	1.26	0.06
Apocynum androsaemifolium	Spreading dogbane			0.08	0.04
Aralia nudicaulis	Wild sarassparilla	0.47	0.27	0.84	0.42
Aralia racemosa	American spikenard	0.27	0.07		
Arisaema triphyllum	Jack in the pulpit			0.28	0.02
Asarum canadense	Wild ginger			0.48	0.14
Aster macrophyllus	Big-leaf aster	4.80	0.47	4.42	0.62
Athyrium felix-femina	Lady fern	0.27	0.07	0.18	0.06
Betula papyrifera	Paper birch			0.04	0.02
Carex	Sedge spp.	4.67	0.67		
Caulophyllum thalictroides	Blue cohosh			0.02	0.02
Circea alpine	Enchanted nightshade			0.46	0.06
Clintonia borealis	Bluebead lily	0.87	0.13	0.60	0.20
Cornus canadensis	Bunchberry			0.16	0.02
Cornus	Dogwood spp.			0.30	0.14
Corylus americana	American hazelnut	0.33	0.07	0.06	0.02
Corylus cornuta	Beaked hazelnut	2.07	0.47	1.26	0.44
Dirca palustris	Leatherwood			0.10	0.04
Dryopteris carthusiana	Spinulose woodfern			0.06	0.04
Equisetum arvense	Horsetail fern	0.47	0.07	0.28	0.06
Fragaria	Wild strawberry spp.	1.33	0.20	2.48	0.50
Fraxinus nigra	Black ash			0.38	0.20
Fraxinus pennsylvanica	Green ash			0.06	0.04
Galium boreale	Northern bedstraw			0.02	0.02
Galium triflorum	Sweet-scented bedstraw			0.14	0.08
Gymnocarpium dyropteris	Oak fern			0.06	0.02
Hepatica americana	Liverleaf	0.13	0.07	1.14	0.18
Impatiens capensis	Jewelweed			0.20	0.02
Lathyrus ochroleucus	Pale vetchling			0.02	0.02
Lathyrus venosus	Woodland vetch			0.28	0.18
Mainthemum canadense	Canada mayflower	4.93	0.47	6.36	0.58
Matteuccia struthiopteris	Ostrich fern			0.34	0.12
Oryzopsis asperfolia	Mountain rice grass	0.93	0.33	3.80	0.66
Osmorhiza claytonii	Sweet cicley	0.73	0.20	0.70	0.24
Ostrya virginiana	Ironwood	0.47	0.20	0.16	0.08
Parthenocissus vitacea	Woodbine			0.02	0.02
Picea glauca	White spruce			0.02	0.02
Polystichum acrostichoides	Christmas fern	0.07	0.07	0.20	0.04
Populus tremuloides	Trembling aspen	0.07	0.07		
Populus	Aspen spp.			0.56	0.04
Prunus viginiana	Choke cherry			0.08	0.02
Pteridium aquilinum	Bracken fern	0.87	0.33	0.74	0.26
Quercus macrocarpa	Bur oak			0.13	0.08
Quercus rubra	Red oak	0.33	0.20	0.06	0.06
Ribes	Gooseberry spp.			0.22	0.08
Rubus acridens	Red raspberry	3.13	0.33	1.42	0.40
Rubus allegheniensis	Common blackberry	0.73	0.20	0.50	0.20
Rubus pubescens	Dwarf red blackberry			0.02	0.02
Sanicula canadensis	Black snakeroot			0.02	0.02
Sanicula marilandica	Maryland sanicle			0.04	0.02

Table 2. continued.

Smilacina racemosa	False Solomon's seal			0.02	0.02
Solidago	Goldenrod spp.	0.07	0.07		
Streptopus lanceolatus	Twisted stalk	3.53	0.40	2.00	0.48
Taraxacum	Dandelion spp.			0.02	0.02
Thalictrum dioicum	Early meadow-rue	1.33	0.33	1.44	0.46
Tilia americana	American basswood			0.04	0.02
Toxicodendron rydbergii	Posion ivy	0.07	0.07		
Trientalis borealis	Star flower	0.27	0.07	0.06	0.06
Triillium	Trillium spp.			0.18	0.06
Ulmus rubra	Red elm			0.10	0.04
Uvularia grandiflora	Large-flowered bellwort	2.73	0.53	3.52	0.58
Uvularia sessilifolia	Sessile-leaved bellwort			1.00	0.26
Vaccinium angustifolium	Lowbush blueberry	0.07	0.07	0.46	0.12
Vicia americana	American vetch			0.02	0.02
Viola	Wild violet spp.	0.07	0.13	0.50	0.12
Viola pubescens	Downy yellow violet	0.13	0.07		

[a] density reported as stem/m^2
[b] frequency reported as number of plots with plant present/total number of plots

Table 3. Density and frequency of plant species in 2 m radius plots and frequency of plant species on transects at Itasca State Park, Minnesota, July 2005, 2006.

		2005			2006		
Species	Common name	Sapling density[a]	Sapling frequency[b]	Canopy frequency[c]	Sapling density[a]	Sapling frequency[b]	Canopy frequency[c]
Acer rubrum	Red maple	1062	0.200	0.003			0.024
Acer saccharum	Sugar maple	1393	0.333	0.093	557	0.600	0.119
Acer spicatum	Mountain maple				378	0.267	0.001
Betula papyrifera	Paper birch	199	0.067	0.063			0.093
Corylus cornuta	Beaked hazelnut	1261	0.400		1971	0.600	
Fraxinus nigra	Black ash				239	0.400	0.114
Fraxinus pennsylvanica	Green ash						0.026
Fraxinus	Ash spp.	199	0.133				
Ostrya virginiana	Ironwood	331	0.133	0.047	438	0.867	0.037
Picea glauca	White spruce			0.003			
Pinus resinosa	Red pine			0.130			0.056
Pinus strobes	White pine						0.019
Populus tremuloides	Trembling aspen	66	0.067		916	0.600	
Populus grandidentata	Big-toothed aspen	531	0.133			0.133	
Populus spp.	Aspen spp.			0.087			0.283
Prunus viginiana	Choke cherry				20	0.067	
Quercus macrocarpa	Bur oak	66	0.067		60	0.133	0.035
Quercus rubra	Red oak						0.037
Quercus	Oak spp.			0.063			
Rubus allegheniensis	Common blackberry	66	0.067				
Tilia Americana	American basswood						0.062
Ulmus rubra	Red elm				40	0.133	0.008

[a] density reported as stems/ha
[b] frequency reported as number of plots with plant present/total number of plots
[c] frequency reported as number of points on transect with plant present/total number of points

Table 4. Plant reproduction in subplots sampled at Itasca State Park, Minnesota, July 2005-2006.

Species	Common name	Flowering?	N	2005 height (cm)	N	2006 height (cm)
Aster macrophyllus	Big-leaf aster	Yes	1	24.00		
		No	71	12.24		
Clintonia borealis	Bluebead lily	Yes			1	27.00
		No			29	17.97
Impatiens capensis	Jewelweed	Yes			1	50.00
		No			9	15.56
Mainthemum canadense	Canada mayflower	Yes			16	8.47
		No			302	5.03
Thalictrum dioicum	Early meadow-rue	Yes	1	50.00		
		No	19	33.42		
Uvularia grandiflora	Large-flowered bellwort	Yes	15	38.13	13	37.69
		No	26	21.80	161	27.46
Streptopus lanceolatus	Twisted stalk	Yes	2	32.00	3	41.33
		No	51	14.94	97	15.81
Viola pubescens	Downy yellow violet	Yes	2	24.00		
		No	0	0.00		

Table 5. Browsing intensity of plants sampled in subplots and 2 m radius plots at Itasca State Park, Minnesota, July 2005, 2006.

Species	Common name	2005 subplot	2006 subplot	2 m radius
Acer spicatum	Mountain maple	2.3 (6)[a]		2.1 (20)
Ostrya virginiana	Ironwood	2.7 (7)		
Populus	Aspen spp.		2.9 (28)	
Prunus viginiana	Choke cherry		2.0 (4)	3.0 (1)
Ulmus rubra	Red elm			2.0 (2)

[a] = sample size

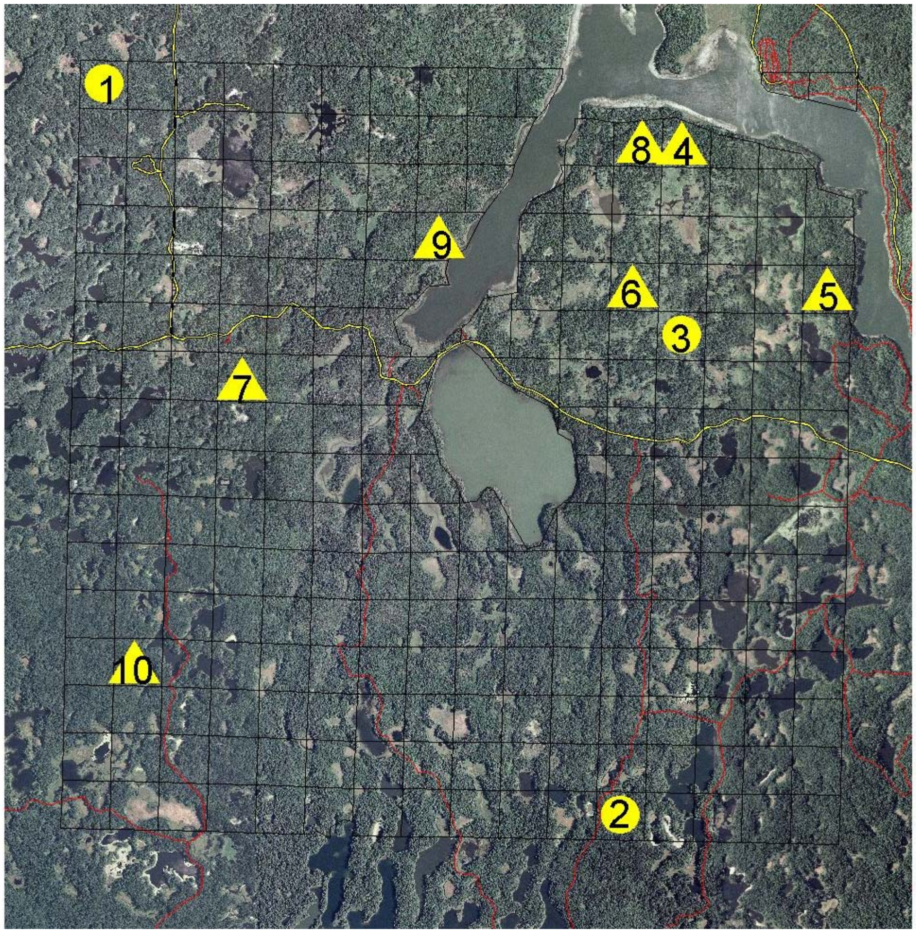

Figure 1. Plot arrays sampled at Itasca State Park, Minnesota, July 2005, 2006.

MANAGEMENT IMPLICATIONS ASSOCIATED WITH HUNTER PREFERENCES TOWARD ALTERNATIVE HUNTING REGULATIONS IN MINNESOTA

Marrett D. Grund, Lou Cornicelli, and David Fulton[1]

SUMMARY OF FINDINGS

Recreational hunting is the primary tool to manipulate white-tailed deer (*Odocoileus virginianus*) populations. In some areas of Minnesota, the number of antlerless deer harvested by hunters under the current seasonal framework is not adequate to reduce deer densities toward population goals. As a result, we surveyed hunters to assess preferences toward regulations that may be more effective at increasing the numerical antlerless deer harvest. We found hunters supported early antlerless-only seasons and ranked early antlerless-only seasons higher than other hunting regulations that we presented in the survey. However, hunters ranked antler-point restriction and earn-a-buck regulations at relatively high levels when we presented regulations that could be used in deer population reduction management scenarios. Our findings suggest that implementing early antlerless-only seasons would be a logical first step toward managing overabundant deer populations followed by antler-point restriction or earn-a-buck regulations. We believe that a public outreach effort may be required if earn-a-buck regulations are implemented as hunter support for this regulation was relatively low. To maintain long-term hunter satisfaction, we speculate that implementing a regulation that protects bucks may be a necessary management component while managing deer densities at prescribed goal levels.

INTRODUCTION

State wildlife agencies rely on recreational hunting to manage deer populations (Woolf and Roseberry 1998). Historically, most state wildlife agencies allowed hunters to harvest 1 antlered deer per year and then restricted antlerless harvests through allocating limited quotas of antlerless licenses. The allowable number of antlerless deer to be harvested depended on, in large part, where the deer population density was relative to a predetermined population goal. Over the past 70 years, deer management has changed from augmenting population growth of deer through habitat protection, hunting regulations, and predator control to serious concerns about how best to limit deer densities and the consequent impacts of deer on society (Conover 1997) and forest ecosystems (Garrott et al. 1993). Today, many state wildlife agencies allow hunters to harvest 1 antlered deer and multiple antlerless deer, but are finding that the majority of hunters are unwilling or unable to harvest more than 1 deer. Consequently, managing overabundant white-tailed deer has emerged as 1 of the most challenging issues in natural resource conservation this past decade (McShea et al. 1997).

Although state wildlife agencies rely on hunters to manage deer populations, previous research suggests that most hunters do not typically perceive hunting as a population management tool (Decker and Connelly 1990). Hunters consider hunting as a recreational activity and consequently, regulations associated with hunting are often debated as to how they affect recreational opportunity, not deer population management goals. Thus, assessing hunter opinions regarding hunting regulations is an important step in the process of implementing innovative management strategies to improve deer population management if the goal is to maintain hunter satisfaction.

This study was conducted by the Minnesota Cooperative Fish and Wildlife Research Unit, Department of Fisheries, Wildlife, and Conservation Biology at the University of Minnesota. A detailed final report, which includes broader deer management issues and strategies, is

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available from Dr. David Fulton (Fulton et al. 2006). This report summarizes key findings from the original report (Fulton et al. 2006) and we suggest a framework for implementing alternative deer hunting regulations based specifically on managing overabundant deer populations.

OBJECTIVES

- Describe hunter effort in Minnesota in 2004 including: type of land hunted, hunting methods and locations, and number of years hunting;
- Describe hunting satisfaction with deer hunting in Minnesota in 2004, and identify activities and experiences that affect hunting satisfaction;
- Determine Minnesota deer hunter support for various regulatory changes that might lead to more mature bucks in the deer population; and
- Determine deer hunter preference for regulatory changes when a finite number of choices are presented to the respondent.

METHODS

Sampling

The study sample was divided into 4 strata: Northwest, Transition Zone, East Central, and South East (Figure 1). These areas represented locations where alternative harvest strategies may be necessary to control and manage deer population growth. Samples were drawn using stratified random sampling of 2004 licensed deer hunters that were >17 years of age in the Electronic Licensing System (ELS) database. At the time of license purchase, hunters were asked to indicate which permit area they intended to hunt most often. Deer harvest data indicated ~90% of successful hunters harvested a deer in the permit area they indicated that they would hunt most often (L. Cornicelli, unpublished data). For this reason, we used responses to the question of which permit area they intended to hunt most often as the basis for stratification of our sample. The target sample size for firearm deer hunters who hunted in each region was 700 (*n*=2,800 statewide). An initial stratified random sample of 6,000 individuals (1,500 in each region) was drawn from the ELS database.

Survey Design

The survey contained 4 sections. The first section contained questions that assessed recent hunter experiences and general perceptions about hunting deer in Minnesota. The second section included questions to quantify hunter support for alternative deer hunting regulations, and the third section focused on past deer hunting experience.

In the fourth section, we provided hunters with different population management scenarios and queried them about what changes in deer hunting regulations were most preferable. Hunters were presented with 5 scenarios related to Minnesota deer management. In total, there were 7 choices within each management scenario, but each hunter was presented only 3 choices in which they were asked to rank preference in descending order (1, 2, 3). Each choice was assigned at random using a balanced incomplete block design (Cochran and Cox 1957), which allowed for the same number of choices represented in all 6,000 surveys. The option of ‘doing nothing’ was not a choice under any scenario as the intent of the instrument was to gauge acceptance of regulation change. However, the options of ‘not hunting’ or ‘moving to another area’ were offered as choices on some scenarios.

This final section of the survey was not designed to gauge hunter support on an issue; rather, it was designed to elucidate a rank-ordering of preferences for management alternatives in response to a specific deer management scenario. We developed 5 scenarios that we believed would occur in Minnesota and asked hunters to rank their preferences for regulation change. The scenarios were:

1. The deer population is stable and within population goals. It is currently being managed so that either-sex licenses are available over the counter and hunters can also buy additional antlerless permits. Based on requests from some hunters, this area will be managed in the future for more mature bucks.
2. The deer population is currently 25% above the management goal. The current strategy of allowing 5 deer per hunter has not been effective in lowering the deer population. A new strategy needs to be developed that lowers the deer population to goal levels within 3 to 5 years.
3. The deer population is currently 50% above the management goal. The current strategy of allocating 5 deer per hunter has not been effective in lowering the deer population. A new strategy needs to be developed that lowers the deer population to goal levels within 3 to 5 years.
4. The deer population is stable or below the population goal and the harvest rate on 1½ year-old bucks is high. Consequently, a low percentage of the buck population lives beyond 1½ years. Currently, buck licenses are available over the counter, either-sex permits are available through the lottery, and hunters can only kill 1 deer. Based on requests from hunters, this area may be managed in the future to protect young bucks and allow them to get to the next age class.
5. Antler point restriction regulations are currently being used by several states to encourage antlerless harvest and protect 1½-year-old bucks. The number of hunters and sporting organizations interested in antler-point restriction regulations seems to be increasing in Minnesota. While the harvest rate of bucks varies in Minnesota, the majority of the bucks killed during the firearm season are 1½ years old. Typically, 50 to 75% of the 1½ year-old buck population is harvested during the firearm season.

Choices were designed to be representative of regulations that might be adopted for that management scenario. For example, earn-a-buck regulations have the potential to decrease deer populations; therefore, earn-a-buck was not a choice in the scenarios where the deer population was stable and/or within goal range. Also, the choice of moving the deer season out of the rut was not presented in the scenarios where the deer population was 25% or 50% above goal density because that regulation likely would not lower deer populations appreciably. Conversely, moving the season was presented as a choice when the scenario suggested the deer population was within goal levels and the desire was to manage for more mature bucks.

We analyzed choice data at 2 levels. First, we consolidated choices into 7 'packages' (e.g., all possible antler-point restriction regulation choices) and looked at the grand mean for each package. Second, we used the mean of the ranks to distinguish between preferred choices by scenario and survey area. We did not include scenario 5 in the consolidation because it was a scenario that included only antler-point restriction regulations and we observed a difference in means between scenarios 1 through 4 and scenario 5 ($t = -5.28$, $p < 0.001$).

Using this approach, we were able to identify both the specifically preferred choice (e.g., antler point restriction with party hunting vs. antler point restriction without party hunting) and preferences for major regulatory changes (e.g., antler point restrictions vs. earn-a-buck). A mean close to 1 implied a preferred choice while a mean approaching 3 indicated a non-preferred choice.

Data Collection

Data were collected using a mail-back survey questionnaire following the process outlined in Dillman (2000). The process involved development of a survey that was relatively easy and was not time consuming to complete. The first 3 sections of the survey were relatively easy to complete; however, the fourth section did require more thought and consideration as it asked hunters to rank order several scenarios that may have had only slight differences between the choices. In total, 3 attempts were made to contact potential respondents. The first mailing was sent in late October, 2005. In the initial attempt, a cover letter, survey

questionnaire, and postage-paid envelope were sent to participants. The cover letter attempted to convey the importance of completing and returning the survey. Approximately 30 days later, a second survey, postage-paid envelope and new cover letter was sent to non-respondents. Approximately 8 weeks after the first mailing, a third mailing was sent to non-respondents with another survey, postage-paid envelope, and cover letter. Returned surveys were collected through March, 2006.

Survey Instrument

The survey was a 16-page (14 pages of questions), self-administered questionnaire (Fulton et al. 2006). The survey was organized into 4 sections and addressed the following topics: 1) Minnesota deer hunting experiences, 2) Deer management in Minnesota, 3) Past hunting experiences, and 4) Choice preferences for deer season options and regulatory changes.

Data Entry and Analysis

The data entry template was designed using the Questionnaire Programming Language version 5 (http://qpl.gao.gov) that allowed for online data entry at any computer with internet access. Data were entered by University of Minnesota undergraduate students where 1 student would enter data and another would proof data entered from the same survey. This method assured 2 individuals reviewed each survey, which decreased data entry errors. Data were analyzed using the Statistical Program for the Social Sciences (SPSS 14). For the statewide level, descriptive statistics and frequencies were computed. Regional level results were compared using chi-square tests, analysis of variance (ANOVA), and cross-tabulations. The choice portion of the survey (Section 4) was analyzed using ANOVA.

Variable Weights and Margin of Error

The study sample was drawn from a stratified random sample of individuals who indicated they hunted in 1 of 4 regions. Therefore, data were weighted to reflect the proportion of hunters sampled within each region and the proportion of regional respondents. For total estimates, data were weighted based on these proportions.

The margin of error for this survey was calculated using the formula provided by Scheaffer et al. (2000). We opted to calculate a maximum error rate, which implied a 50:50 split between responses. Overall, our stratified error rate for this survey was 0.3% and ranged from 3.3% to 3.5% at the regional level. If respondents were treated as a simple random sample drawn statewide, the error estimate was 1.7%. Overall, samples sizes were adequate to draw conclusions both in total and by individual survey areas.

RESULTS AND DISCUSSION

Survey Response Rate

Of the 6,000 questionnaires mailed, 426 were undeliverable, which resulted in 5,574 valid surveys. A total of 3,293 deer hunters completed and returned the questionnaire, yielding an overall response rate of 59%.

Characteristics of Minnesota Deer Hunters

Throughout the regions of Minnesota that we surveyed, we found that virtually all (99%) deer hunting license buyers hunted deer in 2004. In total, deer hunters had approximately 25 years (SD=14 years) of deer hunting experience and the average age of Minnesota hunters was 39 years old. Hunters had approximately 2 years less experience in southeastern Minnesota, which was statistically different ($P < 0.05$) than the other 3 regions, but probably had little affect on practical deer hunting skills. Also in the southeast, a smaller percentage (13%) of respondents used public-owned lands for hunting deer than the other 3 regions (range=27–30%). In all regions that we surveyed, approximately 90% of respondents hunted in the same areas every year, which indicated that they might not be willing to move if new regulations were implemented in their traditional area.

We found that approximately 10% of Minnesota hunters only hunt “big bucks” and another 6% hunt only legal bucks throughout the hunting season. Further, another 21% of hunters are willing to harvest a big buck early, than any deer later in the hunting season. Since nearly 60% of the total firearm deer harvest occurs during the opening weekend (L. Cornicelli, unpublished data), these data indicate that almost 40% of Minnesota deer hunters are not willing to hunt antlerless deer during the period when the vast majority of deer are being harvested in Minnesota. Only 25% of hunters indicated that they were willing to hunt antlerless deer first, and then hunt for antlered deer after an antlerless deer was harvested. These results are encouraging because it indicates that there are many more hunters who could harvest antlerless deer if the DNR implements a regulation that requires or encourages the harvest of antlerless deer.

Perceptions of Deer Populations

Hunters were evenly divided with regards to satisfaction related to “buck quality” in the area that they hunt (Table 1). Interestingly, while antler characteristics of bucks differ among regions in Minnesota (Grund 2004), there were no regional differences in buck quality satisfaction data (Fulton et al. 2006). Bucks likely have larger antlers at younger ages in southern Minnesota due to better soils, more abundant high-quality food, and more mild winters than in northern Minnesota (Grund 2004). These findings may suggest that Minnesota hunters define “buck quality” based on their expectations of what they experienced in the field from prior observations of bucks in that area. In other words, a 6-point buck in northern Minnesota may be defined as a “quality buck” whereas a 6-point buck in southeastern Minnesota would not be considered a “quality buck”. Further analysis to examine this relationship is warranted.

Hunter opinion with regards to the number of “mature bucks” in the area that they hunt was different than the perceptions about “buck quality”. About half of the respondents agreed that there were not enough “mature bucks” in the area that they hunt (Table 1). It is important to point out that “buck quality” may mean different things to different people (Duda et al. 2002). Whether our respondents interpreted “mature bucks” simply as antlered deer or as large-racked bucks is unknown, but half of the hunters that we surveyed indicated that there were not a sufficient number of bucks in the area that they hunted.

Approximately 77% of hunters were satisfied with the number of antlerless deer in the area they hunted, which suggests that Minnesota hunters are not requesting the DNR to restrict the antlerless harvest. The 4 regions of Minnesota that we surveyed had relatively high densities of deer, so this finding was expected. However, about 67% of hunters indicated that they were satisfied with the hunting season because of the number of deer in the area that they hunt (Table 1). This indicates that hunter satisfaction will likely decline as the result of reduced deer densities because hunter satisfaction is often related to the number of deer observed in the field by hunters (Thomas et al. 1973). We believe the ability of hunters to redefine satisfaction on a factor unrelated to overall deer numbers is paramount for responsible deer management to occur in the future.

Perceptions of Hunting Regulations

Although only 50% of respondents indicated they were not satisfied with the number of mature bucks in the area that they hunted, approximately 66% indicated that they would support a regulation that increased the proportion of bucks in the area that they hunt. In contrast, only 13% of hunters indicated they opposed a regulation that would increase the proportion of bucks in the area that they hunt (Table 1). Apparently, there is a discrepancy between current satisfaction levels related to the number of "mature bucks" in current deer populations (Table 1) versus a hunter's willingness to increase the proportion of "mature bucks" in deer populations. There are >5 times as many hunters supportive of implementing a regulation to increase the proportion of "mature bucks" as there are opposing such a regulation (Table 1). This finding presents a challenge because the only regulatory options the DNR has to choose from is to increase the number of bucks through decreased mortality of bucks or increased mortality of antlerless deer (Grund 2004). To increase the proportion of bucks in a population through reducing buck mortality, the DNR would need to adopt some hunting regulation that would reduce hunting pressure on the buck population. To increase the proportion of bucks in a population through increasing antlerless deer mortality, which would ultimately reduce deer densities, the DNR would need to increase hunter pressure on the antlerless deer population while maintaining an equal amount of hunting pressure on the buck population. However, buck harvest mortality rates and deer density are inversely related under either-sex deer seasons (Roseberry and Woolf 1991), so some regulation would likely be required to reduce hunting pressure on bucks while deer densities decline.

In terms of support for alternative hunting regulations, hunter support exceeded opposition for early antlerless-only seasons, antler-point restriction regulations, and eliminating cross-tagging of bucks (Table 1). In general, about 2 hunters opposed buck license lottery regulations, moving the season outside the rut, and eliminating cross-tagging of all deer for every hunter that supported such regulations. These results suggest that a majority of hunters are willing to support some new hunting regulations, but other regulations will receive minority support if the regulation is simply implemented without educational/outreach efforts. The degree to which hunter support toward a regulation may be affected by educational efforts is unknown, but definitely warrants an attempt with coinciding research.

Choosing an Alternative Hunting Regulation

Overall, hunters indicated a clear preference for going hunting, even though they may not agree with changing regulations. In our sample, the option of not hunting in an area if regulations were adopted consistently ranked below all other options. The early antlerless season ranked highest (mean = 1.6/3.0), followed by antler point restrictions (mean = 1.8/3.0), earn-a-buck (mean = 1.8/3.0), move the deer season (mean = 1.8/3.0), continue to hunt despite objecting to regulations (mean = 2.0/3.0), buck license lottery (mean = 2.2/3.0), and will not hunt in the area if regulations are implemented (mean = 2.6/3.0).

Management Scenario 1: Population at Goal but Manage for Mature Bucks—We observed distinct trends in that hunters seemed willing to accept regulation changes so long as they were able to continue hunting every year. In this scenario, the least restrictive antler-point restriction regulation ranked highest, followed by moving the season out of the rut and then the most restrictive antler-point restriction regulation. Buck license lotteries and changing hunting locations if regulations were enacted ranked very low overall. Consequently, in this scenario, it appeared hunters would be accepting of some regulation change so long as they were able to pursue bucks every year. When faced with the choice of a buck license lottery, which would mean a hunter would not obtain an annual buck license annually, hunters tended to rank this option lower than the others.

Overall, the following regulatory options were ranked as follows:

1. Antler-point restriction regulation to protect 50% of the yearling buck population and no buck party hunting (mean = 1.7/3.0).
2. Antler-point restriction regulation to protect 75% of the yearling buck population and party hunting legal (mean = 1.8/3.0).
3. Move the deer season out of the rut (mean = 1.8/3.0).
4. Antler-point restriction regulation to protect 75% of the yearling buck population and no buck party hunting (mean = 1.9/3.0).
5. Buck license lottery, party hunting legal, fewer buck licenses (mean = 2.1/3.0).
6. Buck license lottery, party hunting not legal, more buck licenses (mean = 2.2/3.0).
7. Would not hunt the area if the regulations were changed (mean = 2.6/3.0).

Management Scenario 2: Population is 25% Above Goal—Hunters generally ranked their choices from least intrusive (early antlerless-only season) to the most restrictive (buck license lottery). The option of changing hunting location again ranked low and the motivational trends appeared similar to scenario 1 in that hunters want the option of pursuing bucks every year.

Overall, the following regulatory options were ranked as follows:

1. Early antlerless-only season (mean = 1.7/3.0).
2. Antler-point restriction regulation to protect 50% of the yearling buck population and no buck party hunting (mean = 1.8/3.0).
3. Antler-point restriction regulation to protect 75% of the yearling buck population and buck party hunting legal (mean = 1.8/3.0).
4. Earn-a-buck regulation (mean = 1.8/3.0).
5. Buck license lottery, party hunting not legal, more buck licenses (mean = 2.1/3.0).
6. Buck license lottery, party hunting legal, fewer buck licenses (mean = 2.2/3.0).
7. Would not hunt the area if the regulations were changed (mean = 2.6/3.0).

Management Scenario 3: Population is 50% Above Goal—Hunters again ranked the early antlerless-only season highest. Mean values under this scenario were comparable to scenario 2 except that hunters ranked earn-a-buck regulations slightly higher than antler-point restriction regulations.

Overall, the following regulatory options were ranked as follows:

1. Early antlerless-only season (mean = 1.6/3.0).
2. Earn-a-buck regulation (mean = 1.8/3.0).

3. Antler-point restriction regulation to protect 75% of the yearling buck population and party hunting legal (mean = 1.8/3.0).
4. Antler-point restriction regulation to protect 50% of the yearling buck population and no buck party hunting (mean = 1.8/3.0).
5. Buck license lottery, party hunting not legal, more buck licenses (mean = 2.2/3.0)
6. Buck license lottery, party hunting legal, fewer buck licenses (mean = 2.2/3.0).
7. Would not hunt the area if the regulations were changed (mean = 2.7/3.0).

Management Scenario 4: Population at Goal, High Buck Harvest Rates, Limited Antlerless Harvest—Choices in this scenario ranged from moving the deer season out of the rut to limiting the number of buck licenses that would be allocated. Earn-a-buck and early antlerless seasons were not offered as choices because this management scenario did not relate to the need to increase antlerless deer harvests in order to lower deer densities. Overall, hunters displayed a clear interest in having buck hunting opportunity every year as the lottery option ranked lowest again.

Overall, the following regulatory options were ranked as follows:

1. Antler-point restriction regulation to protect 75% of the yearling buck population, party hunting legal, youth can take any buck (mean = 1.7/3.0).

2. Antler-point restriction regulation to protect 75% of the yearling buck population, party hunting legal, youth must abide by regulation (mean = 1.7/3.0).
3. Antler-point restriction regulation to protect 50% of the yearling buck population, no buck party hunting, youth must abide by regulation (mean = 1.8/3.0).
4. Move the deer season out of the rut (mean = 1.8/3.0).
5. All licenses lottery (buck and antlerless), party hunting legal (mean = 2.2/3.0).
6. All licenses lottery (buck and antlerless), party hunting not legal (mean = 2.3/3.0).
7. Would not hunt the area if the regulations were changed (mean = 2.5/3.0).

Management Scenario 5: Implementation of Antler-Point Restriction Regulations—Hunters displayed a preference for a regulatory package that allowed youth hunters to shoot any buck, and ranked the antler-point restriction regulation that protected 75% of the yearling buck population highest but still allowed party hunting. In general, regulations that were increasingly restrictive and did not provide for youth to harvest any deer were ranked lower. The choice of 'not liking antler point regulations but would hunt anyway' ranked higher than the most restrictive antler point regulation (protect 75%, no party hunting, youth abide). As in the other 4 scenarios, the option of changing hunt location if regulations were adopted ranked lowest.

Overall, the following antler point restriction regulation options were ranked as follows:

1. Protect 75% of the yearling buck population, party hunting legal, youth can take any deer (mean = 1.7/3.0).
2. Protect 50% of the yearling buck population, buck party hunting not legal, youth can take any deer (mean = 1.9/3.0).
3. Protect 50% of the yearling buck population, buck party hunting not legal, youth must abide by the regulation (mean = 1.9/3.0).
4. Protect 75% of the yearling buck population, party hunting legal, youth must abide by the regulation (mean = 1.9/3.0).
5. Opposed to antler point restriction regulations but would still hunt the area (mean = 2.0/3.0).
6. Protect 75% of the yearling buck population, buck party hunting not legal, youth must abide by the regulation (mean = 2.0/3.0).
7 Would not hunt the area if the regulations were changed (mean = 2.7/3.0).

MANAGEMENT IMPLICATIONS

Perhaps the most important finding from this survey is that hunters indicated that they will choose to hunt even if they disagree with a new deer hunting regulation. This finding is critical because the effectiveness associated with a hunting regulation will ultimately depend on the hunter's willingness and ability to harvest deer under the new regulation. Thus, it is imperative that hunters are willing to hunt even though they may not support a particular regulation.

This study found that a very high percentage of Minnesota deer hunters are not interested in harvesting an antlerless deer early during the hunting season. Current statewide regulations allow any deer hunter to hunt for a buck without linking that opportunity to harvesting an antlerless deer. Regulations that require or encourage harvest of antlerless deer during the early part of the season may be very effective at increasing the antlerless harvest since most (67%) of the deer harvest occurs during that time frame. An example of a regulation that would require antlerless deer to be harvested early in the season would be an earn-a-buck regulation. There are many ways to encourage antlerless deer harvest. Two examples of regulations that may encourage hunters to harvest antlerless deer may be to allow the taking of a second buck or providing a free late-season hunting license if an antlerless deer is registered during the opening weekend. We are not recommending any of the aforementioned regulations be implemented in Minnesota. We provide these regulations as examples of strategies that may encourage or require the harvest of antlerless deer during the early part of the hunting season.

We also found that the majority of hunters were satisfied with the current number of antlerless deer in deer populations. Based on previous studies that demonstrate hunter satisfaction is related to the number of deer a hunter observes (Thomas et al. 1973), it is reasonable to expect that hunter satisfaction will decline as a result of implementing an alternative hunting regulation that causes deer population reduction. However, this study also found that hunters strongly supported regulations that would increase the proportion of mature bucks in the population. We cannot discern if hunter satisfaction would remain at higher levels if a greater proportion of the population is comprised of mature bucks after deer densities are reduced. However, the management strategy of maintaining a higher proportion of mature bucks in the population, which should increase hunter satisfaction (while deer densities that are managed at lower densities should reduce hunter satisfaction), may be the most logical long-term management strategy to maintain hunter satisfaction given our findings from this survey. Thus, regulations that reduce harvest vulnerability of antlered deer may be a necessary component to deer management if the long-term goal is to maintain deer densities at substantially lower levels. Further research is needed to evaluate this theory.

When the management scenario involved population reduction, hunters ranked early antlerless-only seasons over other regulatory options that we presented. Further, hunters supported early antlerless-only seasons over other proposed regulations as well. However, from a practical management perspective, ranked means associated with antler-point restriction regulations and earn-a-buck regulations were not substantially different than early antlerless-only seasons. Even though hunters generally opposed earn-a-buck regulations (Table 1), it appears that hunters recognized the need for the regulation when population reduction was necessary and a suite of hunting regulation alternatives were presented to them in management scenarios 2 and 3. This might suggest that the Agency ought to invest substantial efforts into educating hunters so that hunters understand the proposed population reduction management scenario as well as management alternatives to manipulate the deer population toward the population goal.

We suggest a reasonable management approach for population reduction would be to first implement early antlerless-only seasons, then implement antler-point restriction or earn-a-buck regulations (with outreach efforts preceding the implementation of the regulations) if the early antlerless-only season did not provide an adequate antlerless harvest for population reduction. Whether antler-point restriction or earn-a-buck regulations are implemented would depend on the harvest efficiency associated with each regulation relative to the numerical antlerless deer harvest required for population reduction. In order to maintain long-term hunter satisfaction, implementation of a regulation that would maintain higher proportions of mature bucks in deer populations may be warranted once deer populations reach goal levels. Our survey indicates that implementing antler-point restriction regulations as part of this long-term population maintenance phase would be the most acceptable regulation.

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Table 1. Percentages of hunters agreeing/disagreeing with survey questions related to population management and alternative deer hunting regulations (from Fulton et al. 2006).

Question	Hunters agree (%)	Hunters disagree (%)	Agree:Disagree
Satisfied with buck quality in area that you hunt?	43	43	1.0:1.0
Satisfied with number of mature bucks in area that you hunt?	39	50	0.8:1.0
Satisfied with number of antlerless deer in area that you hunt?	77	15	5.1:1.0
Satisfied with total number of deer in area that you hunt?	67	24	2.8:1.0
Support regulation to increase proportion of mature bucks?	66	13	5.1:1.0
Support early antlerless-only season?	50	32	1.6:1.0
Support antler-point restriction regulation?	47	43	1.1:1.0
Support regulation that would prohibit cross-tagging of bucks?	46	42	1.1:1.0
Support earn-a-buck regulation?	37	48	0.8:1.0
Support limiting the number of buck license?	29	59	0.5:1.0
Support moving season outside the rut?	29	55	0.5:1.0
Support regulation that would prohibit cross-tagging of all deer?	28	61	0.5:1.0

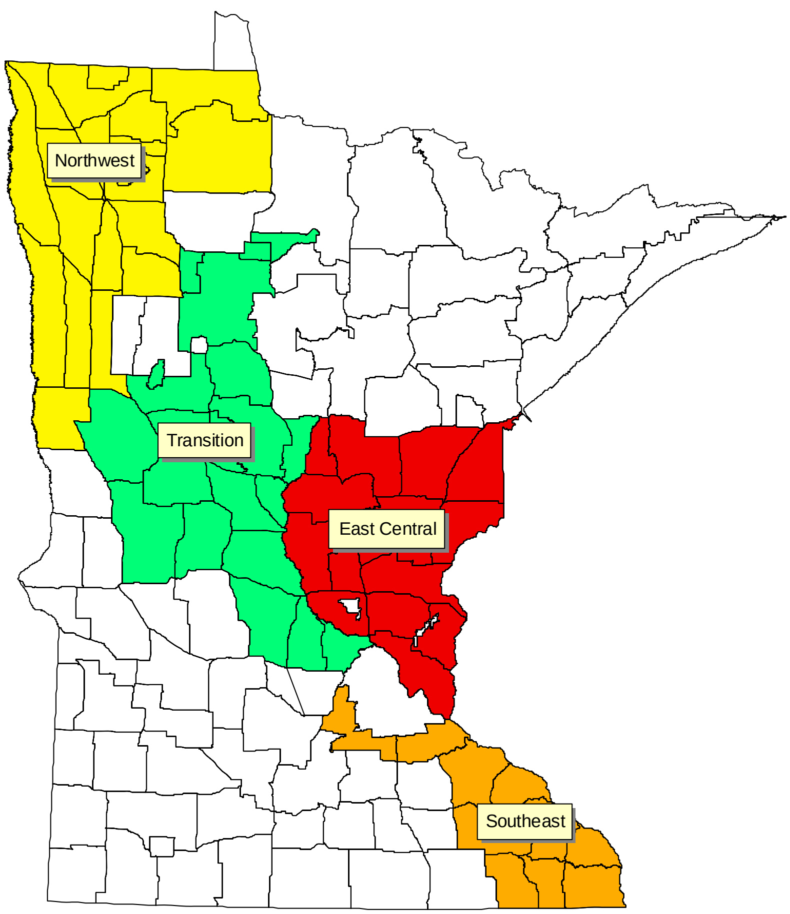

Figure 1. Deer permit areas in Minnesota with choice survey regions shaded, 2004.

ESTIMATING WHITE-TAILED DEER ABUNDANCE USING AERIAL QUADRAT SURVEYS

Brian S. Haroldson and John H. Giudice

SUMMARY OF FINDINGS

We estimated white-tailed deer (*Odocoileus virginianus)* abundance in select permit areas using stratified random and 2-dimensional systematic 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. Beginning in 2008, we will begin to develop a sightability estimator to adjust estimates for animals missed during surveys.

INTRODUCTION

Management goals for animal populations are frequently expressed in terms of population size (Lancia et al. 1994). Accurate and precise 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 each of the 125 permit areas (PA). Traditionally, these goals were established by wildlife managers, largely without public input. MNDNR is currently revising deer population goals within each PA using a consensus-based, round–table approach consisting of 15-20 citizens representing varied interest groups (e.g. deer hunters, farmers, foresters, environmental groups, etc.; Stout et al. 1996). Once goals are established, they 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 has begun 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 and precise estimates of deer abundance are needed to evaluate these regulations.

The objective of this study is to provide independent estimates of deer abundance in select PAs. These 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

We estimated deer populations in the 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. We employed a stratified, random sampling design, with quadrats stratified into 2 abundance classes (low, high) based on relative deer densities, in PAs where the local wildlife manager had prior knowledge about deer abundance and distribution. In other areas, we used a 2-dimensional systematic sampling design (Cressie 1993, D'Orazio 2003). Systematic designs are typically easier to implement, maximize sample distribution, and are often more efficient than simple or stratified random sampling designs (Cressie 1993, D'Orazio 2003).

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 effort was needed to provide 90% confidence interval population estimates that were within 20% of the true population size. We excluded quadrats containing navigation hazards or high human development, and selected replacement quadrats in stratified PAs. Replacement quadrats were unavailable in the systematic PAs because of the rigid, 2-dimensional design. We used OH-58 helicopters during most surveys. However, a Cessna 182 airplane was used in 3 PAs dominated by intensive row-crop agriculture. To increase visibility, we completed surveys after leaf-drop by deciduous vegetation and when snow cover measured at least 15 cm. The pilot and 2 observers searched for deer along transects, generally spaced at 270-m intervals within each quadrat, until they were confident no more animals would be observed. We 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. We estimated deer abundance from stratified surveys using SAS Proc SURVEYMEANS (SAS 1999) and from systematic surveys using formulas developed by D'Orazio (2003).

RESULTS AND DISCUSSION

We completed 5 surveys during January-February 2005, 8 surveys during January-March 2006, and 7 surveys during January-March 2007 (Table 1). Stratified fixed-wing surveys were conducted in PAs 421 and 423. Based on long-term deer harvest metrics, population estimates in these areas were biased low. Several possibilities may explain this result: 1) quadrats were stratified incorrectly; 2) deer were clustered in unsampled quadrats; 3) deer were wintering outside PA boundaries; 4) sightability was biased using fixed-wing aircraft; and/or 5) kill locations from hunter-killed deer were reported incorrectly. Land cover in these PAs was dominated by intensive row crop agriculture. After crops are 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, it was not a common practice in these PAs. Thus, we had no evidence to support poor stratification (1) or non-traditional deer distribution (3) in these units. We also had no reason to believe hunter registration errors had greater bias in these units than in other PAs (5). Although it was possible that deer occupied unsampled quadrats by chance (2), our use of optimal allocation to increase sampling effort in high strata plots because of expected higher deer densities should minimize this possibility. Furthermore, we surveyed 100% of the high-strata plots in PA 421, resulting in no unsampled quadrats. Sightability bias (4), 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 we observed in these PAs. Beginning in 2007, all surveys were conducted using a helicopter.

With the exception of PAs 421, 423, and 201, precision (CV, relative error) of our population estimates was similar among PAs (Table 1). High precision in PA 421 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 quadrants. We observed few deer in these low strata quadrats, resulting in low sampling variance and, therefore, 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 423 and 201 was poor. We observed few deer during either survey (n=144 and 56, respectively). In addition, most quadrats contained no deer, and nearly all observations occurred within 1 or 2 quadrats. Resulting confidence intervals were only within 60% of the true population size (Table 1). Kufeld et al. (1980) described similar issues with precision due to nonuniformity of mule deer distribution within strata in Colorado.

We did not correct population estimates for sightability. Thus, estimates represent minimum counts and are biased low. Although sightability correction factors for deer are available in the literature (Rice and Harder 1977, Ludwig 1981, Stoll et al. 1991, Beringer et al. 1998), we believe it would be inappropriate to apply them to our survey areas because of differences in sampling design and habitat characteristics. Beginning in 2008, we will attempt to develop a sightability estimator to adjust for animals missed during surveys. This estimator will improve our population estimates by reducing visibility bias. 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 prevalence of winter feeding by landowners, and its impact on deer distribution, will also 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, 2005-2007.

Sampling design	Year	Permit area	Sampling rate (%)	Population estimate *N*	Population estimate 90% CI	CV (%)	Error (%)[1]	Density estimate ($deer/mi^2$) Mean	Density estimate ($deer/mi^2$) 90% CI	Model estimate ($deer/mi^2$)
Systematic	2005	252	16	2,999	2,034 – 3,969	19.5	32.2	2.9	2.0 – 3.9	2
		257	16	2,575	1,851 – 3,299	16.9	28.1	6.2	4.4 – 7.9	7
	2006	204	16	3,432	2,464 – 4,401	17.0	28.2	4.6	3.3 – 5.9	5
		209	17	6,205	5,033 – 7,383	11.4	18.9	9.7	7.9 – 11.5	5
		210	17	3,976	3,150 – 4,803	12.5	20.8	6.3	5.0 – 7.6	7
		256	17	4,670	3,441 – 5,899	15.9	26.3	7.1	5.3 – 9.0	5
		236	16	6,774	5,406 – 8,140	12.1	20.2	16.8	13.4 – 20.2	37
	2007	225	17	5,341	4,038 – 6,645	14.7	24.4	8.0	6.0 – 9.9	24
		227	17	5,101	4,245 – 5,960	10.1	16.8	9.8	8.2 – 11.5	13
		346	16	7,896	5,736 – 10,062	16.4	27.4	22.7	16.5 – 29.0	31
Stratified	2005	206	20	2,486	1,921 – 3,051	13.7	22.5	5.2	4.0 – 6.4	5
		342	20	3,322	2,726 – 3,918	10.8	17.7	9.1	7.5 –10.7	10
		421	20	631	599 – 663	3.0	5.0	0.8	0.8 – 0.9	5
	2006	201	20	274	100 – 449	37.6	61.9	1.6	0.6 – 2.7	6
		420	20	1,740	1,301 – 2,180	15.2	25.1	2.6	2.0 – 3.3	3
		423	20	472	179 – 764	37.4	61.5	0.9	0.3 – 1.4	5
	2007	343	20	6,982	5,957 – 8,006	8.9	14.6	10.1	8.6 – 11.6	29
		344[2]	25	4,116	3,375 – 4,857	10.7	17.7	19.7	16.1 – 23.2	49
		347[2]	21	5,482	4,472 – 6,492	11.1	18.2	12.6	10.3 – 14.9	13
		349[2]	23	10,103	8,573 – 11,633	9.1	15.0	20.4	17.3 – 23.5	35

[1]Relative precision of population estimate (goal: 90% CI that is within +/- 20% of the true pop'n size). Calculated as 90% CI bound ⁄ *N*.

[2]Survey area included State Park property within the permit area.

EFFECTS OF ALTERNATIVE DEER HUNTING REGULATIONS ON HUNTER HARVESTS IN MINNESOTA

Marrett D. Grund

SUMMARY OF FINDINGS

I examined white-tailed deer (*Odocoileus virginianus*) harvest data associated with traditional and alternative hunting regulations being tested in Minnesota. Hunters in early antlerless-only seasons and under earn-a-buck regulations were more willing and able to harvest multiple antlerless deer. Antlerless harvest projections associated with these 2 hunting regulations were highest of those that I investigated. However, a previous study indicated that only 57% of Minnesota hunters are willing to participate in an early antlerless-only season. Accounting for this level of hunting effort, antlerless harvest projections under early antlerless-only seasons remained 14-20% higher than intensive management regulations, which was the most aggressive traditional hunting regulation. However, since an earn-a-buck regulation is not voluntary like the early antlerless-only season, earn-a-buck antlerless harvest projections were 60-86% higher than 5-deer bag limit regulations. I also found that antlerless harvest projections associated with 2-deer bag limits were substantially lower than 5-deer bag limit regulations due to a lower percentage of hunters willing or able to harvest a second antlerless deer in managed permit areas. These preliminary results indicate that early antlerless-only seasons may slightly increase the antlerless harvest in comparison to 5-deer bag limit regulations, but earn-a-buck regulations will markedly increase antlerless harvests.

INTRODUCTION

In 1972, Minnesota Department of Natural Resources (MN DNR) closed the deer hunting season due to the scarcity of white-tailed deer. In 1973, the MN DNR adopted a new seasonal framework that allowed deer hunting to occur each year but also allowed populations to grow. Essentially, hunters were allowed to hunt antlered deer, but antlerless deer could not be harvested unless the hunter was awarded an antlerless permit through a lottery. Since then, annual deer harvests have increased almost 4-fold in Minnesota. Clearly, deer populations successfully recovered due to these regulation changes.

With the exception of the southwest and south-central regions of Minnesota where little woody habitat exists, some wildlife managers are more concerned about controlling increasing population growth rates of deer rather than restricting antlerless harvests. For almost a decade, managers have suggested liberalizing regulations associated with harvesting antlerless deer in attempt to reduce population growth rates. In 2005, the MN DNR adopted an Alternative Deer Management (ADM) research project to examine biological and social ramifications associated with regulations that were traditionally used in Minnesota. The ADM regulations were designed to increase antlerless deer harvests. In this paper, I provide preliminary results associated with comparing traditional deer hunting regulations to those used in the ADM study.

OBJECTIVES

- Analyze harvest patterns of hunters hunting under alternative deer hunting regulations and current statewide deer hunting regulations; and
- Estimate numerical harvests associated with alternative deer hunting regulations and current statewide deer hunting regulations.

METHODS

There were 4 hunting zones in Minnesota that are used to determine season timing and length in Minnesota during 2005 and 2006 (see 2005 and 2006 Minnesota Hunting and Trapping Regulations Handbooks). In addition, there were approximately 130 permit areas in Minnesota where deer hunting regulations were applied. At the statewide level, there were 3 basic hunting regulations used for deer management in permit areas: 1) Lottery (LOT) – hunters could hunt only antlered deer unless an individual was awarded an antlerless permit through a lottery of a limited number of antlerless permits; 2) Managed (MAN) – hunters were provided with an either-sex hunting license and an additional antlerless-only hunting license; and 3) Intensive (INT) – hunters were provided with an either-sex hunting license and an antlerless-only hunting license, and hunters could purchase up to 3 additional antlerless-only hunting license at a reduced license fee.

As part of this study, early antlerless-only hunting seasons were offered in 5 permit areas in northwestern Minnesota and 3 permit areas in east-central Minnesota. Hunters were required to purchase an early antlerless-only hunting license at a reduced cost to hunt during this voluntary season, which was held during the second weekend of October. I evaluated antler-point restriction and earn-a-buck regulations in 7 state parks in Minnesota. Deer bag limits were identical to INT permit areas for all study areas associated with this study.

Hunters participating in each hunting regulation in 2005 and 2006 were identified in the Electronic Licensing System (ELS) database and were categorized according to each hunting zone. I then conducted simple frequency analyses to determine the number of hunters intending to hunt under each regulation. Similarly, I used the ELS deer harvest database to identify the number of deer individual hunters harvested under each hunting regulation. I then conducted a simple frequency analysis to estimate the percentage of hunters that were unsuccessful, and the number of hunters that harvested 1, 2, and >2 deer for each hunting regulation.

In order to compare harvest efficiency associated with each hunting regulation, I projected numerical harvests by standardizing the number of hunters (effort) for each regulation. To standardize effort, I assumed that there were 1,000 hunters hunting under each regulation and projected the number of antlerless deer harvested based on the proportion of hunters who harvested 0, 1, 2, and >2 deer.

RESULTS AND DISCUSSION

Harvest Patterns Among Regulations

The number of permit areas associated with each hunting regulation differed between 2005 and 2006 due to changes in hunting regulations needed to manage deer populations according to population goals (Tables 1 and 2). However, from a practical harvest management perspective, there were very similar trends in the proportion of hunters taking 0, 1, 2, or >2 deer under each hunting regulation between years. An exception was the earn-a-buck regulation. The percentage of hunters not harvesting an antlerless deer under earn-a-buck regulations increased from 54% in 2005 to 63% in 2006. Hunter success rates are usually inversely related to deer density in a linear fashion (Roseberry and Woolf 1991). Thus, the increased proportion of hunters not harvesting an antlerless deer in 2006 may be attributable to a reduction in deer densities between years.

Projected Antlerless Harvests

In general, ADM regulations were more effective at increasing projected antlerless harvests than non-ADM regulations (Tables 3 and 4). Early antlerless-only seasons produced the highest antlerless harvest projections of the 3 ADM regulations. This result may be misleading, however, because early antlerless-only seasons are voluntary and only 57% of hunters indicated they would participate in an early antlerless-only season if that season was available in their area (Fulton et al. 2006). The early antlerless-only season harvest projections adjusted for this level of hunter effort remained 14-20% higher than harvest projections associated with comparable INT regulations. In contrast to the voluntary antlerless-only seasons, all hunters would need to participate under antler-point restriction and earn-a-buck regulations because those regulations would be applied during the regular firearms season. In 2005, projected antlerless harvests under antler-point restriction regulations were similar to projected antlerless harvests under INT regulations. However, the projected antlerless harvest under antler-point restriction regulations increased during 2006. This might suggest that hunters adapted to the antler-restriction regulation and were more willing to harvest antlerless deer during their second year of experience under the regulation. Harvest projections associated with earn-a-buck regulations were 60-86% higher than comparable INT regulations. At this point, my study suggests that earn-a-buck regulations will produce the highest antlerless harvests of the 3 ADM regulations when applied during the regular firearms season.

Projected antlerless harvests were highest under INT regulations followed by MAN and LOT regulations. It has been suggested that there is little practical difference in the number of antlerless deer that would be harvested under MAN or INT regulations. However, antlerless harvest projection comparisons between INT and MAN regulations suggest otherwise (Tables 3 and 4). Deer density may explain some of this occurrence since a MAN regulation would more likely be used under lower deer densities than INT regulations. However, deer densities are markedly different in permit areas located in Zones 2 and 3 that used MAN regulations compared to permit areas located in Zones 1 and 4 that used MAN regulations (Grund 2005, Lenarz 2005); yet the projected antlerless harvests under MAN regulations were comparable in all zones.

Observed differences between MAN and INT regulations have little to do with the hunter's ability to harvest up to 5 antlerless deer under INT regulations. Rather, it appears that hunters are more willing to harvest a second antlerless deer under INT regulations than under MAN regulations. There are 2 likely reasons for hunters to be more willing to harvest a second antlerless deer under INT regulations. First, a hunter who harvested an antlerless deer during the early part of the hunting season under MAN regulations may be less willing to harvest another antlerless deer because that would prevent the hunter from harvesting a buck. The other scenario is a hunter who harvested an antlered deer during the early portion of the hunting season under MAN regulations is only able to harvest 1 additional antlerless deer. Regardless, managers should use caution when considering MAN regulations when high numerical antlerless harvests are needed or desired.

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Table 1. Number of management areas, number of hunters, and number of hunters registering antlerless deer under different management strategies employed throughout Minnesota during the 2005 deer hunting season.

	Management areas	Hunters[a]	Number of Hunters Registering (%) No antlerless[b]	1 antlerless	2 antlerless	>2 antlerless
Zone 1						
Lottery	9	25,099	22,204 (89)	2,828 (11)	62 (0)	5 (0)
Managed	14	90,310	68,904 (76)	19,107 (21)	2,213 (2)	86 (0)
Intensive	7	55,205	39,537 (72)	12,401 (22)	2,495 (5)	772 (1)
Zone 2[c]						
Lottery	0	0	n/a	n/a	n/a	n/a
Managed	9	14,694	10,960 (75)	3,363 (23)	353 (2)	18 (0)
Intensive	29	105,571	75,016 (71)	24,225 (23)	4,909 (5)	1,421 (1)
Zone 3[de]						
Lottery/Lottery	1	2,539	2,222 (88)	309 (12)	8 (0)	2 (0)
Lottery/Managed	3	5,888	4,813 (82)	982 (17)	86 (1)	7 (0)
Lottery/Intensive	3	12,369	9,343 (76)	2,431 (20)	474 (4)	121 (0)
Managed/Intensive	4	15,730	10,757 (69)	3,823 (24)	836 (5)	314 (2)
Zone 4						
Lottery	24	35,254	30,421 (86)	4,772 (14)	58 (0)	2 (0)
Managed	12	26,831	21,195 (79)	5,636 (21)	274 (1)	10 (0)
Intensive	11	36,725	25,279 (69)	9,619 (26)	1,468 (4)	357 (1)
Alternative Regulations						
Early Antlerless Season	8	4,848	2,676 (55)	1,338 (28)	561 (12)	273 (6)
Antler-Point Restriction	3	765	540 (71)	174 (23)	39 (5)	12 (1)
Earn-a-Buck	4	900	484 (54)	279 (31)	104 (12)	33 (4)

[a] Hunters who declared where they intended to hunt
[b] Estimated based on the number of hunters registering deer versus the number of hunters declaring where they intended to hunt
[c] Excluding Permit Area 228
[d] Split Season: A Season Strategy/B Season Strategy
[e] Excluding Permit Area 337

Table 2. Number of management areas, number of hunters, and number of hunters registering antlerless deer under different management strategies employed throughout Minnesota during the 2006 deer hunting season.

	Permit areas	Hunters[a]	Number of Hunters Registering (%) No antlerless[b]	1 antlerless	2 antlerless	>2 antlerless
Zone 1						
Lottery	4	2,448	2,221 (91)	222 (9)	2 (0)	3 (0)
Managed	15	92,131	72,231 (79)	17,880 (19)	1,991 (2)	29 (0)
Intensive	11	71,635	54611 (76)	13,994 (20)	2,416 (3)	614 (1)
Zone 2[c]						
Lottery	1	247	214 (87)	32 (13)	1 (0)	0 (0)
Managed	12	31,363	25,393 (81)	5,370 (17)	570 (2)	30 (0)
Intensive	34	120,089	90,518 (75)	23,831 (20)	4,531 (4)	1,209 (1)
Zone 3[de]						
Lottery/Lottery	1	2,600	2,172 (84)	416 (16)	12 (0)	0 (0)
Lottery/Managed	2	3,277	2,896 (88)	354 (11)	24 (1)	3 (0)
Lottery/Intensive	2	8,210	6730 (82)	1,257 (15)	178 (2)	45 (1)
Managed/Intensive	4	14,308	11,260 (79)	2,556 (18)	391 (3)	101 (0)
Intensive/Intensive	2	9,118	6,682 (73)	1,881 (21)	424 (5)	131 (1)
Zone 4						
Lottery	28	47,622	42,207 (89)	5,369 (11)	42 (0)	4 (0)
Managed	10	16,553	13,989 (85)	2,419 (15)	136 (0)	9 (0)
Intensive	1	1,836	1,448 (79)	343 (19)	39 (2)	6 (0)
Alternative Regulations						
Early Antlerless Season	8	6,041	3,248 (54)	1865 (31)	680 (11)	248 (4)
Antler-Point Restriction		745	516 (69)	165 (22)	42 (6)	22 (3)
Earn-a-Buck	4	783	497 (63)	210 (27)	57 (7)	19 (3)

[a] Hunters who declared where they intended to hunt
[b] Estimated based on the number of hunters registering deer versus the number of hunters declaring where they intended to hunt
[c] Excluding Permit Area 228
[d] Split Season: A Season Strategy/B Season Strategy
[e] Excluding Permit Area 337

Table 3. Projected antlerless harvests based on a hypothetical scenario of 1,000 hunters in each management area. Numerical harvests were derived based on proportional harvest patterns for each management strategy used in the 2005 Minnesota deer hunting season (see Table 1).

		Numerical Antlerless Harvest Based on Hunters Registering:				
	Hunters	No antlerless	1 antlerless	2 antlerless	>2 antlerless[a]	Total
Zone 1						
Lottery	1,000	0	110	0	0	110
Managed	1,000	0	210	40	0	250
Intensive	1,000	0	220	100	35 (3.5)	355
Zone 2						
Lottery	n/a	n/a	n/a	n/a	n/a	n/a
Managed	1,000	0	230	40	0	270
Intensive	1,000	0	230	100	35 (3.5)	365
Zone 3						
Lottery/Lottery	1,000	0	120	0	0	120
Lottery/Managed	1,000	0	170	20	0	190
Lottery/Intensive	1,000	0	200	80	0 (3.3)	280
Managed/Intensive	1,000	0	240	100	66 (3.3)	406
Zone 4						
Lottery	1,000	0	140	0	0	140
Managed	1,000	0	210	20	0	230
Intensive	1,000	0	260	80	32 (3.2)	372
Alternative Regulations						
Early Antlerless Season	1,000	0	280	240	210 (3.5)	730
Antler-Point Restriction	1,000	0	230	100	32 (3.2)	362
Earn-a-Buck	1,000	0	310	240	128 (3.2)	678

[a] Average number of deer registered by hunters registering >2 antlerless deer is in parentheses after the projected numerical harvest.

Table 4. Projected antlerless harvests based on a hypothetical scenario of 1,000 hunters in each management area. Numerical harvests were derived based on proportional harvest patterns for each management strategy used in the 2006 Minnesota deer hunting season (see Table 2).

		Numerical Antlerless Harvest Based on Hunters Registering:				
	Hunters	No antlerless	1 antlerless	2 antlerless	>2 antlerless[a]	Total
Zone 1						
Lottery	1,000	0	90	0	0	90
Managed	1,000	0	190	40	0	230
Intensive	1,000	0	200	60	36 (3.6)	296
Zone 2						
Lottery	1,000	0	130	0	0	130
Managed	1,000	0	170	40	0	210
Intensive	1,000	0	200	80	37 (3.7)	317
Zone 3						
Lottery/Lottery	1,000	0	160	0	0	160
Lottery/Managed	1,000	0	110	20	0	130
Lottery/Intensive	1,000	0	150	40	34 (3.4)	224
Managed/Intensive	1,000	0	180	60	0 (3.5)	240
Intensive/Intensive	1,000	0	210	100	37 (3.7)	347
Zone 4						
Lottery	1,000	0	110	0	0	110
Managed	1,000	0	150	0	0	150
Intensive	1,000	0	190	40	0 (3.2)	230
Alternative Regulations						
Early Antlerless Season	1,000	0	310	220	140 (3.5)	670
Antler-Point Restriction	1,000	0	220	120	102 (3.4)	442
Earn-a-Buck	1,000	0	270	140	96 (3.2)	506

[a] Average number of deer registered by hunters registering >2 antlerless deer is in parentheses after the projected numerical harvest.

PREY OF WOLVES IN THE GREAT LAKES REGION

Glenn D. DelGiudice, Keith R. McCaffery, Dean E. Beyer, and Michael E. Nelson

Wolves (*Canis lupus*) were abundant in the Great Lakes region just prior to early European settlement (early to mid-1800s). The subsequent extirpation of wolves and most of their large prey is just one of the many threats humans have posed to the existence of North American wildlife, both by exploitation and by indifference. To fully understand (and learn from) the recent ongoing recovery of wolves in the Great Lakes region (Minnesota, Wisconsin, and Michigan), and the historic declining trend that preceded it, requires consideration of their food or prey, both historically and today. Ungulate prey species are often at the center of the wolf-human conflict, where and when it occurs. What history, recent management, and research have taught us is that it was not wolf predation that diminished the diversity and richness of ungulate species in the Great Lakes region, but rather the human "appetite," and unfortunately for wolves and their prey, the unprecedented drive to satisfy it.

On the other hand, humans have a great capacity for conservation when that is their true intention. But the success of such efforts also relies largely on species-specific biology, in this case, not just of wolves, but of their existing prey as well. Wolves are adaptable, opportunistic predators when it comes to their foraging behavior, but what animal species become their prey has depended largely on the potential prey's size, abundance, and vulnerability. Consequently, the relative contributions of primary and secondary prey to the diets of Great Lakes wolves, to their individual health and welfare, and long-term population persistence, have changed historically, and today continue to vary seasonally, annually, and across the landscape.

This chapter begins with a brief description of the historic trends in distribution and relative abundance of the large ungulates that were likely most important in the multi-prey system of the Great Lakes wolf. Our major focus, however, is the more recent trend of white-tailed deer (Odocoileus virginianus), the wolves' primary prey in a single ungulate prey system that has persisted throughout the 20th century and during their recent ongoing recovery. We concentrate our discussion on specific aspects of the deer's ecology that have enabled its populations to thrive despite relatively heavy human exploitation, increasing numbers of wolves, and a concomitant expansion of their range. This discussion is based upon management efforts and an unparalleled amount of data generated from the study of coexisting white-tailed deer and wolves in the Great Lakes region. We devote similar, but more limited attention to moose, which are primary or secondary prey for wolves on Isle Royale and in various portions of northern Minnesota.

From Chapter 10 in "Recovery of Gray Wolves in the Great Lakes Region of the United States: an Endangered Species Success Story." Submitted.

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, Barry A. Sampson, and David W. Kuehn

SUMMARY OF FINDINGS

During January to March 1991 to 2005, a total of 452 female white-tailed deer (*Odocoileus virginianus*), including 43 female newborns, were captured, radiocollared, and recruited into this study assessing the effects of varying winter severities and diminishing conifer cover on numerous aspects of white-tailed deer ecology. The wide-ranging severity of winter weather conditions (winter severity index of 38 in winter 2003-2004 to 195 in winter 1995-1996) during the past 15 years, and the diverse data we have collected, will continue to provide a more comprehensive understanding of white-tailed deer ecology in much of Minnesota's forest zone as we continue our data analyses. During the past year, we've been concentrating our efforts on several tasks, including: 1) organizing the diverse, 15-year data sets in preparation for analyses specifically related to the experimental design of this study; 2) completing data analyses and manuscript preparations; and 3) updating the habitat composition layers for the 4 study sites relative to vegetative succession, natural habitat destruction (e.g., by flooding), and timber harvests. We describe how we addressed vegetative succession and the types of changes we observed, particularly relative to conifer cover, and 4) continued monitoring of the survival of wolves (*Canis lupus*) radiocollared in the region of our study sites and reporting cause-specific mortality during 2006.

INTRODUCTION

The goal of this long-term investigation is to assess the value of conifer stands as winter thermal cover/snow shelter for white-tailed deer at the population level. Historically, conifer stands have declined markedly relative to the increasing number 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 harvests of conifers (to varying degrees) compared to hardwoods, because of evidence, at least at the individual animal 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

The null hypotheses in this study are that conifer stands have no effect on the survival, movement, or distribution of female white-tailed deer during winters of varying severities. Relative to varying winter severities, the specific objectives of the comprehensive, quasi-experimental approach of this study have been to:

- monitor deer movements between seasonal ranges by aerial radio-telemetry, and more importantly, within winter ranges, for determination of home range size;
- determine habitat composition of winter home ranges and deer use of specific vegetation types;

- monitor winter food habits;
- monitor winter nutritional restriction and condition via serial examination of deer body mass and composition, blood and bladder-urine profiles, and urine specimens suspended in snow (snow-urine);
- monitor age-specific survival and cause-specific mortality of all study deer; and
- collect detailed weather data in conifer, hardwood, and open habitat types to determine the functional relationship between the severity of winter conditions, deer behavior (e.g., use of habitat) and their survival.

STUDY DESIGN AND PROGRESS

This study employed a replicated manipulative approach, which is a modification of the Before-After-Control-Environmental Impact design (BACI; Stewart-Oaten et al. 1986; see DelGiudice and Riggs 1996). The study involves 2 control sites (Willow and Dirty Nose Lakes) and 2 treatment sites (Inguadona and Shingle Mill Lakes), a 5-year pre-treatment (pre-impact) phase, a 4-year treatment phase (conifer harvest serves 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 Lakes sites during winter 1990-1991. The Shingle Mill and Dirty Nose Lakes sites were included beginning in winter 1992-1993. The objective of the experimental treatment (impact) was to reduce moderate (40-69% canopy closure) and optimum (≥70% canopy closure) conifer thermal cover/snow shelter to what is considered a poor cover class (< 40% canopy closure).

Data collected on all 4 study sites included the following: 1) descriptive quantification of deer habitat by color infrared air photo interpretation, 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; 4) age determination by last incisor extraction and cementum annuli analysis; 5) physiological samples collected during captures and recaptures of radiocollared female deer and data generated from laboratory analyses, 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 visual ultrasound; 6) morphological measurements; 7) assessment of winter nutritional restriction by chemical analysis of urine in snow; 8) seasonal migrations and other movements via very high frequency (VHF) and Global Positioning System (GPS) radiocollars; 9) habitat use; 10) annual and seasonal cause-specific mortality; 11) age-specific survival rates; 12) winter food habits; and 13) movements, territory size, survival, and cause-specific mortality of radiocollared wolves.

We completed the 15th and final year of data collection. Ultimately, we radiocollared and monitored a total of 452 female deer, including 43 female newborn fawns. During 1991 to 2006, in annual issues of the Minnesota Department of Natural Resources' "Summaries of Wildlife Research Findings" we've presented summary data describing the winter weather conditions (e.g., weekly snow depths, monthly mean daily minimum and maximum ambient temperatures, winter severity index); live-capture success; and age distribution, pregnancy, and fecundity (fetuses:doe) rates of the female cohort recruited for this study. Additionally, in those summaries we've addressed winter and annual mortality rates (and their relations to the varying severities of winter weather conditions), specific causes of mortality (e.g., hunting, wolf predation, "miscellaneous"), and how the underlying age-specific hazard function (instantaneous probability of death) drove age-specific, seasonal, and annual survival rates of these females from birth to old age (up to 17.5 years old). To varying degrees we've presented preliminary descriptions of seasonal migration patterns of the collared deer; margins of understanding of safe capture, chemical immobilization, and handling; food habits; assessments of winter nutritional restriction and condition; as well as the territory sizes, survival, and specific fates of wolves ranging over the study sites.

Additionally, during the past 16 years, we've published a number of scientific and popular articles that have delved into many of the aforementioned topics in much greater detail than appropriate for the annual research summaries. Importantly, often as a result of collaborations with our Research Unit's biometricians (M. Riggs, J. Fieberg), these scientific articles and their associated in-depth analyses have allowed us to explore new, more scientifically rigorous, and illuminating analytical approaches to viewing the diverse data sets we were accumulating during this long-term study. These large data sets, analyses, and articles facilitated not only an increased understanding of numerous aspects of white-tailed deer ecology that we've been able to share with the scientific and management communities, but ultimately served as preparation for our most important upcoming data analyses relative to the long-term study's BACI design, primary goals, and objectives (described above). The many popular articles and presentations also allowed us to share current, interesting information synthesized from the data with numerous, diverse special interest groups, academic (K-12 and college-level) audiences, and the general public over the years.

During the past year, we've been concentrating our efforts on several tasks, including 1) completing data analyses and manuscript preparations of the type discussed above; 2) organizing the diverse, 15-year data sets in preparation for analyses specifically related to the BACI experimental design, and 3) updating the habitat composition layers for the 4 study sites relative specifically to vegetation succession and any other changes over the 15-year period. We describe below how we addressed vegetative succession and the types of changes we observed.

HABITAT ANALYSES AND UPDATES

Detailed baseline habitat analyses using mirror stereoscope interpretation of color infrared air photos (1:15,840) and geographic information systems (GIS, Arc/Info, ArcView) were completed early in the study (Figures 1 to 4). Forest stand types were classified according to their dominant 2-3 tree species, height, and winter canopy closure classes. Open habitat types, water sources, and roads were also delineated. The classification system was developed with the specific intent that it would facilitate an examination of potential relations between use of habitat types by white-tailed deer and their winter biological requirements. Page and space constraints herein would not allow us to present near the level of detail in Figures 1 through 4 of the actual habitat analyses, but the coverages depicted in these figures provide a general representation of the vegetative mosaics (highlighting conifer canopy closure class) that comprise the winter range of white-tailed deer in north-central Minnesota.

During the 15-year study period there was potential for natural and human-induced changes of the vegetation/habitat to occur. Because we are examining habitat use by study deer (via radio-telemetry) during each year, it was important to update the classification of the habitat layers of the 4 study sites to account for vegetative succession, as well as habitat destruction (e.g., by flooding). This was particularly important for types that were openings when the study began, as well as for conifer types with canopies that may have succeeded from a less dense closure class (A [< 40%] or a B [40-69%]) to a more dense class (B or a C [≥70%]).

We had current air photo coverage taken and rectified by fall 2006 at a scale of 1:15,840. We then were able to compare specific habitat types from the initial interpretation with the current coverage and determine whether significant change, particularly in conifer canopy closure classes, had occurred. Overall, on the Willow Lake control site, conifers increased 22.6% due to succession, with increases specifically in canopy closure classes A, B, and C of 29.7, 26.9, and 16.5%, respectively, from 1991 to 2005 (Table 1). Conversely, on the Dirty Nose Lake control site, conifers declined 22.7%, with specific changes of 20.5, 30.8, and 23.7% in canopy closure classes A, B, and C, respectively, from 1993 to 2005 (Table 1). At the Inguadona Lake treatment site, conifers were reduced by 18.2%, primarily associated with the mid-study treatment harvests, with 19.0 and 65.5% decreases in the A and C classes, respectively. However, canopy closure class B showed an overall 39.7% net increase. Finally, at the Shingle Mill Lake site, decreases in all classes (A, 8.2%; B, 27.5%; and C, 7.5%) accounted for an overall decrease in area of conifers of 12.9% (Table 1). Net changes in conifer canopy closure classes were attributable primarily to a combination of natural and human-induced sources, including: 1) destruction of stands by natural seasonal flooding; 2) planned, mid-study, experimental treatment conifer harvests; 3) non-study, planned timber harvests committed to by cooperators (primarily U. S. Forest Service) prior to initiation of the study; and 4) gradual natural succession during the 13-15 years each site was part of the long-term study. The specific effects of all of these sources of change will be quantified using GIS technology, air photo coverage, and documentation available from the different landownership cooperators. Furthermore, a number of mitigating circumstances will be considered as part of the statistical analyses ultimately conducted relative to the *a priori* objectives and goals of the study. For example, many of the non-study timber harvests conducted at the Dirty Nose Lake site occurred at the periphery of the site or in areas of the site not being used regularly by radio-monitored study deer. Consequently, it is presumed their impact on deer behavior and habitat use will not be as significant as it potentially could have been. Also, a number of the cuts were made in the last year or 2 of the post-treatment phase; if examination of the data suggests that these cuts were a source of disturbance or bias relative to study deer distribution or habitat use, data of the potentially affected deer will be excluded from certain analyses.

Detailed spatial and temporal analyses of annual deer use of habitat types on the study sites relative to specific winter weather conditions and overall winter severity will begin during the current year. A preliminary analysis has shown that during phases of the study associated with mild to average winter conditions, deer distribution over the study sites was more dispersed and use of vegetative cover was more variable, whereas when influenced by severe winter conditions, deer locations were more concentrated in dense conifer cover. Location data sets from about 35 GPS-radiocollared deer (programmed to collect data at 1 to 6-hour intervals over 24-hour daily periods) during 2000-2006, will be used to augment analyses of data collected from VHF-radiocollared deer and to enhance our understanding of deer use of winter cover types relative to varying weather conditions.

MONITORING ACTIVITY AND CAUSE-SPECIFIC MORTALITY OF WOLVES

Wolves were extirpated from the region of our study sites during the 1950s-1960s, but as their population recovered, they naturally expanded their range and became re-established in this region just 5 years prior to the initiation of our study in 1991. With this, data from our long-term study show that wolves are the primary source of natural mortality for female deer at least 0.5 years old. Presently, our study sites are near the leading southern edge of wolf range expansion. Since spring 1993, we have captured and radiocollared 57 (31 females, 26 males) wolves from 7 to 9 packs that range over the 4 study sites (Table 2). We radio-located these wolves from fixed-wing, year-around, in order to monitor their survival and investigate causes of mortality. Fates of these wolves include being killed by a variety of human-related and natural causes. During 2006, 1 radiocollared wolf was shot, 1 was snared, and 1 had its collar chewed off by other wolves.

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. Approximately 145 enthusiastic, competent, and dedicated interns have made collection of winter field data possible, and we thank them for their efforts. 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.

Table 1. The change in area (hectares) of conifer canopy closure classes (A [< 40%], B [40-69%], and C [≥70%]) on the 4 study sites of the white-tailed deer/conifer winter cover study, Grand Rapids-Remer-Longville, Minnesota, 1991 to 2005.

Site	1991/1993[a]	2005	Change
Willow Lake			
A	273	354	81
B	108	137	29
C	399	465	66
Total	780	956	176
Dirty Nose Lake			
A	493	392	-101
B	120	83	-37
C	97	74	-23
Total	710	549	-161
Inguadona Lake			
A	788	638	-150
B	239	334	95
C	278	96	-182
Total	1,305	1,068	-237
Shingle Mill Lake			
A	389	357	-32
B	273	198	-75
C	398	368	-30
Total	1,060	923	-137

[a]The Willow Lake and Inguadona Lake sites entered the study in 1991 as a control and treatment site, respectively, whereas the Dirty Nose Lake and Shingle Mill Lake sites entered the study in 1993 as a replicate control and treatment site, respectively.

Table 2. History of radiocollared gray wolves, north-central Minnesota, 1993 to 2006 (AD=adult, JUV=juvenile).

Wolf number	Pack	Capture date	Sex	Age class	Fate	Date
2093	WILLOW	MAY 1994	F	AD	SHOT	MAR 1996
2094	WILLOW	MAY 1994	M	AD	SHOT	NOV 1997
2056	WILLOW	MAY 1996	M	AD	NOT COLLARED	
2058	WILLOW	MAY 1996	F	AD	PROB. SHOT	AUG 1996
2052	NORTH INGY	MAY 1993	M	AD	UNKNOWN	DEC 1996
2087	SOUTH INGY	MAY 1993	F	AD	DIED FROM NATURAL CAUSES (EMACIATED, MANGEY)	AUG 2, 1998
2062	SOUTH INGY	AUG 1997	F	AD	SHOT	FEB 1998
2089	SHINGLE MILL	MAY 1993	F	AD	KILLED BY WOLVES	SEP 1994
2050	SHINGLE MILL	MAY 1993	M	AD	COLLAR CHEWED OFF	AUG 1993
2095	SHINGLE MILL	MAY 1995	F	AD	LOST SIGNAL	NOV 1995
2064	SHINGLE MILL	AUG 1996 MAY 2004	F	JUV	ON THE AIR	
2060	SHINGLE MILL	AUG 1996 JUL 1998 – RECAPTURED	F	JUV	LOST SIGNAL	FEB 1, 2000
2059	SHINGLE MILL	AUG 1996	M	JUV	LOST SIGNAL	OCT 1996
2085	DIRTY NOSE	MAY 1993	M	AD	DISPERSED	OCT 1993
2054	DIRTY NOSE	MAY 1993	M	AD	DISPERSED	SEP 1993
2091	DIRTY NOSE	APR 1994	F	AD	RADIO FAILED	MAY 27, 1998
2092	DIRTY NOSE	APR 1994	F	AD	RADIO FAILED	MAY 27, 1998
2096	MORRISON	MAY 1995	F	AD	DROPPED TRANSMITTER	NOV 22, 1996
2252	WILLOW	APR 1998	M	AD	ROAD-KILL	JUN 1998
2253	DIRTY NOSE	APR 1998	F	AD	UNKNOWN MORTALITY	AUG 3, 1998
2254	SHINGLE MILL	JUL 1998	M	AD	DROPPED TRANSMITTER	JUL 17, 2001
2066	MORRISON	JUL 1998	M	AD	KILLED BY WOLVES	JUN 4, 1999
2067	SHINGLE MILL	JUL 1998	M	JUV	COLLAR CHEWED OFF	JUL 1998
2068	HOLY WATER	JUL 1998	M	AD	LOST SIGNAL	AUG 27, 1999
2069	SOUTH INGY	JUL 1998	M	AD	LOST SIGNAL	DEC 4, 1998
2070	SOUTH INGY	JUL 1998	F	AD	LOST SIGNAL	JUL 3, 2002
2255	SOUTH INGY	JUL 1998	F	AD	DISPERSED	MAR 22, 1999
2256	DIRTY NOSE	AUG 1999	M	AD	DROPPED TRANSMITTER	JUL 6, 2001
2257	E. DIRTY NOSE	MAY 1999	M	AD	LOST SIGNAL	JAN 14, 2001
2258	WILLOW	AUG 1999	M	AD	DISPERSED	MAR 16, 2000
2259	DIRTY NOSE	JUL 2000	M	AD	DISPERSED	JUL 2001
2261	SHINGLE MILL	AUG 2000	M	AD	DROPPED TRANSMITTER	APR 10, 2002
2074	SOUTH INGY	AUG 2001	F	AD	SHOT BY FARMER	OCT 23, 2002
2073	SHINGLE MILL	AUG 2001	F	JUV	DROPPED TRANSMITTER	AUG 28, 2001
2071	SHINGLE MILL	SEP 2000	F	AD	SNARED	JAN 13, 2001
2139	SHINGLE MILL	AUG 2002 RECAPTURED JUN 2003	F	AD	SNARED	DEC 24, 2006
2141	INGUADONA	SEP 2002	F	JUV	DROPPED TRANSMITTER	SEP 22, 2002
2149	INGUADONA	MAY 2003	M	AD	SHOT	NOV 2003
2143	WILLOW	MAY 2003	M	AD	KILLED BY WOLVES	JUN 20, 2004
2144	MORRISON BROOK	JUN 2003	F	AD	SHOT	NOV 12, 2004
2145	INGUADONA	JUL 2003	F	AD	DIED, MANGE	JAN 3, 2004
2148	WILLOW	AUG 2003	F	AD	DISPERSED	DEC 2, 2003
2291	SMITH CREEK	AUG 2003	F	AD	LOST SIGNAL	MAR 28, 2005
2146	WILLOW	AUG 2003	F	JUV	DISPERSED	MAR 15, 2005
2262	DIRTY NOSE	SEP 2003	F	AD	SHOT	NOV 14, 2003
2263	SHINGLE MILL	MAY 2004	F	AD	SHOT	NOV 24, 2006
2264	DIRTY NOSE	MAY 2004	F	AD	ON THE AIR	
2266	WILLOW	MAY 2004	F	AD	ROAD-KILL	NOV 6, 2004
2267	INGUADONA	MAY 2004	M	AD	KILLED BY WOLVES	MAR 3, 2005
2268	INGUADONA	MAY 2004	M	AD	UNKNOWN MORTALITY	JAN 19, 2005
2269	WILLOW	MAY 2004	M	AD	DISPERSED	JUN 2004
2270	WILLOW	MAY 2005	M	AD	COLLAR CHEWED OFF	NOV 7, 2006
2271	SHINGLE MILL	MAY 2005	F	AD	ON THE AIR	
2272	UNAFFILIATED	MAY 2005	M	AD	ON THE AIR	
2273	INGUADONA	JUN 2005	F	AD	ROAD-KILL	FEB 8, 2006
2289	UNAFFILIATED	JUL 2005	M	AD	KILLED BY WOLVES	AUG 13, 2005
2290	SHINGLE MILL	AUG 2005	F	JUV	SLIPPED COLLAR	AUG 2005
2292	SHINGLE MILL	AUG 2005	M	JUV	SLIPPED COLLAR	AUG 2005

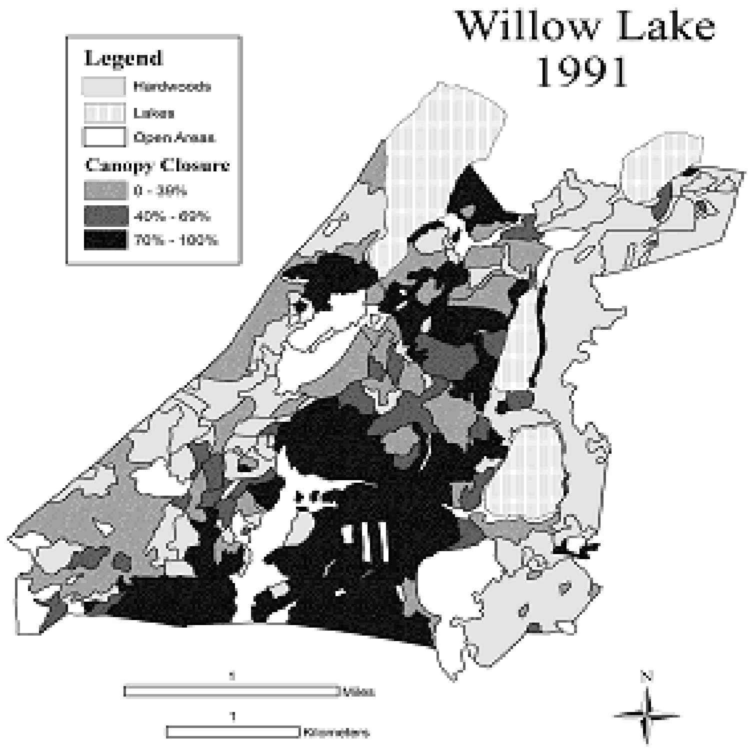

Figure 1. Habitat mosaic of Willow Lake control site, Grand Rapids-Remer-Longville, Minnesota, 1991–2005.

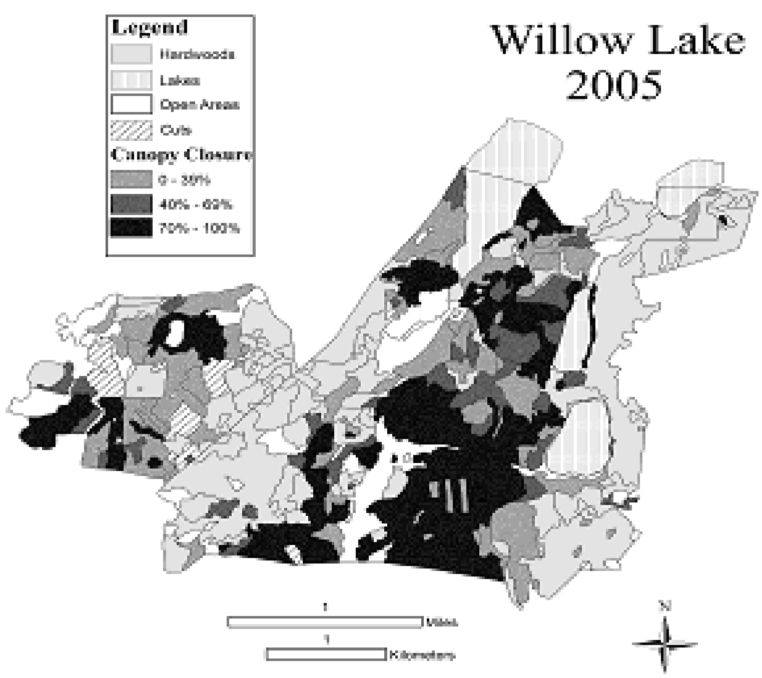

Figure 1. Continued.

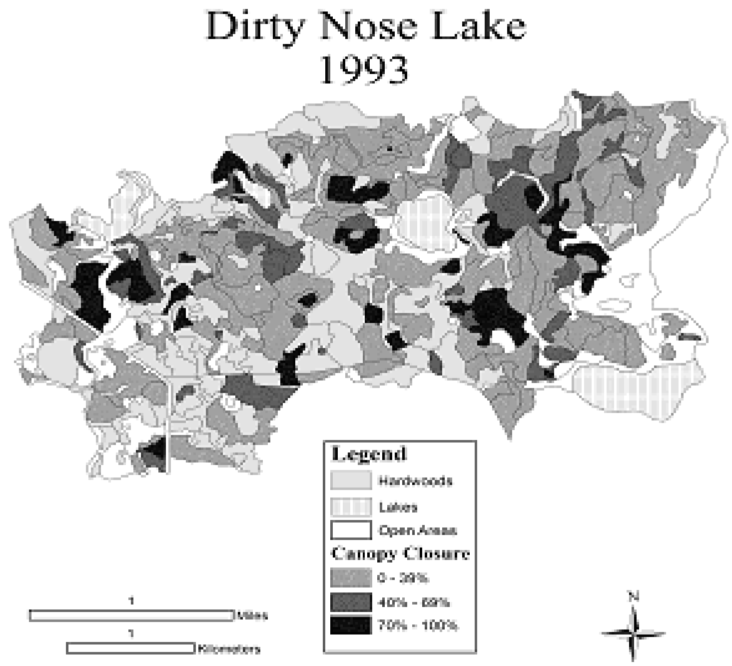

Figure 2. Habitat mosaic of Dirty Nose Lake control site, Grand Rapids-Remer-Longville, Minnesota, 1993–2005.

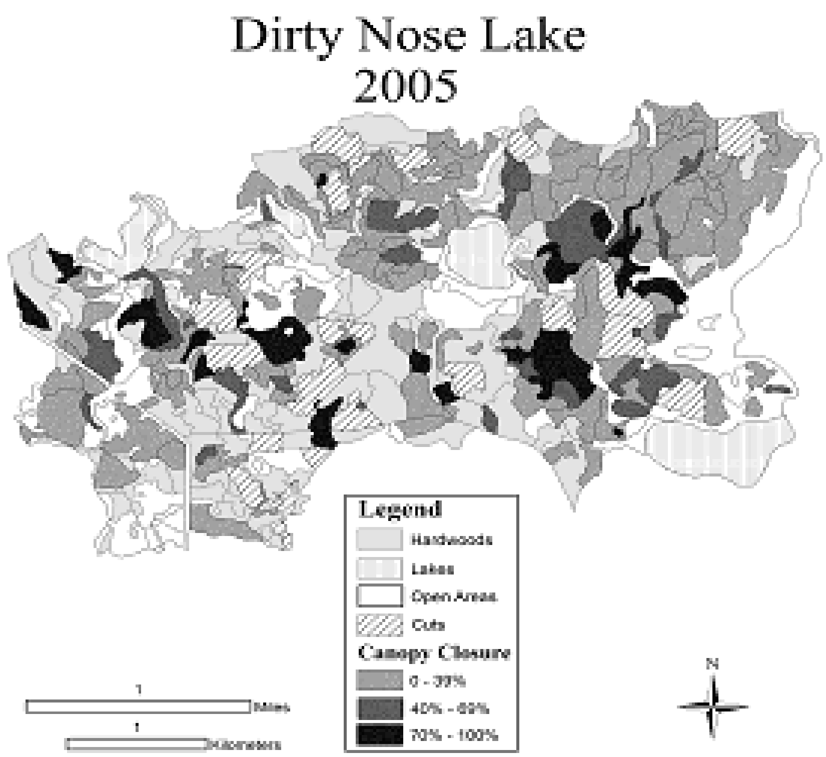

Figure 2. Continued.

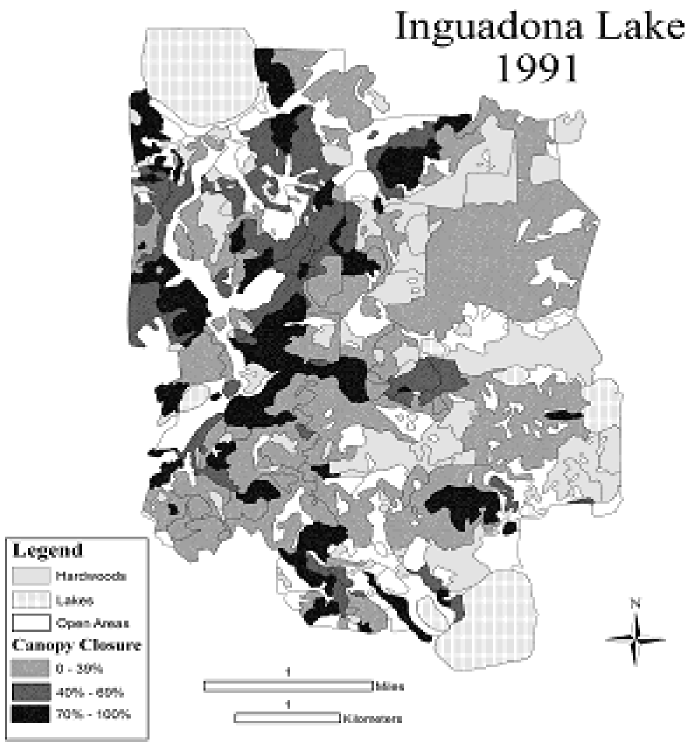

Figure 3. Habitat mosaic of Inguadona Lake treatment site, Grand Rapids-Remer-Longville, Minnesota, 1991–2005.

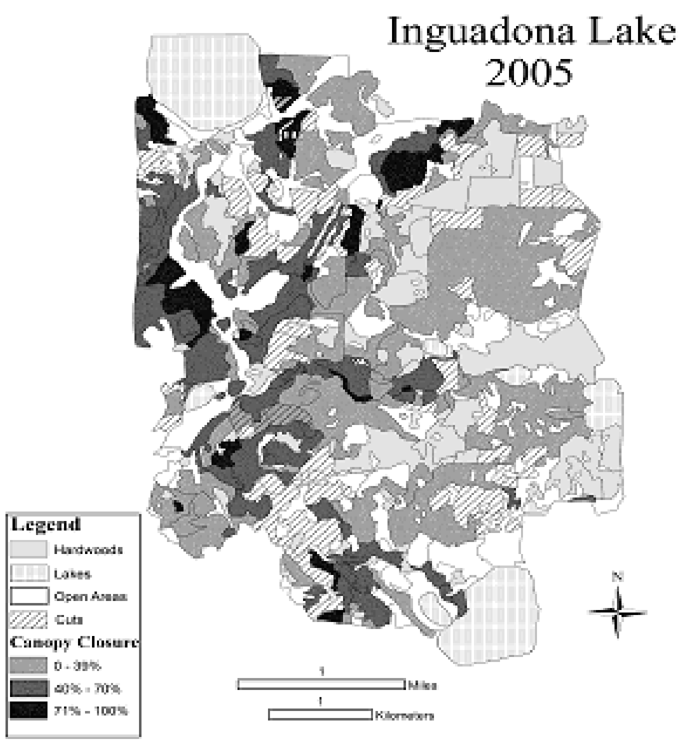

Figure 3. Continued.

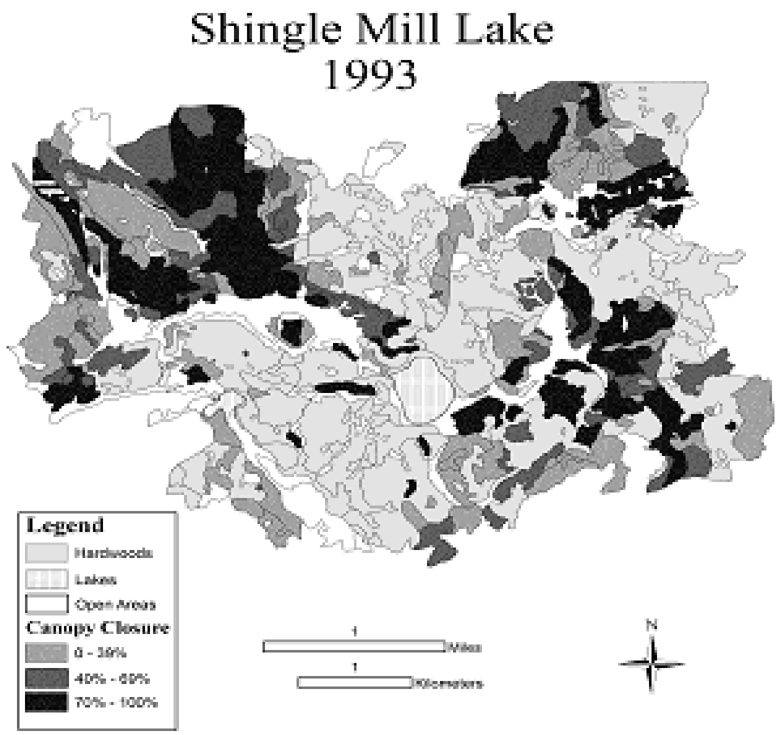

Figure 4. Habitat mosaic of Shingle Mill Lake treatment site, Grand Rapids-Remer-Longville, Minnesota, 1993–2005.

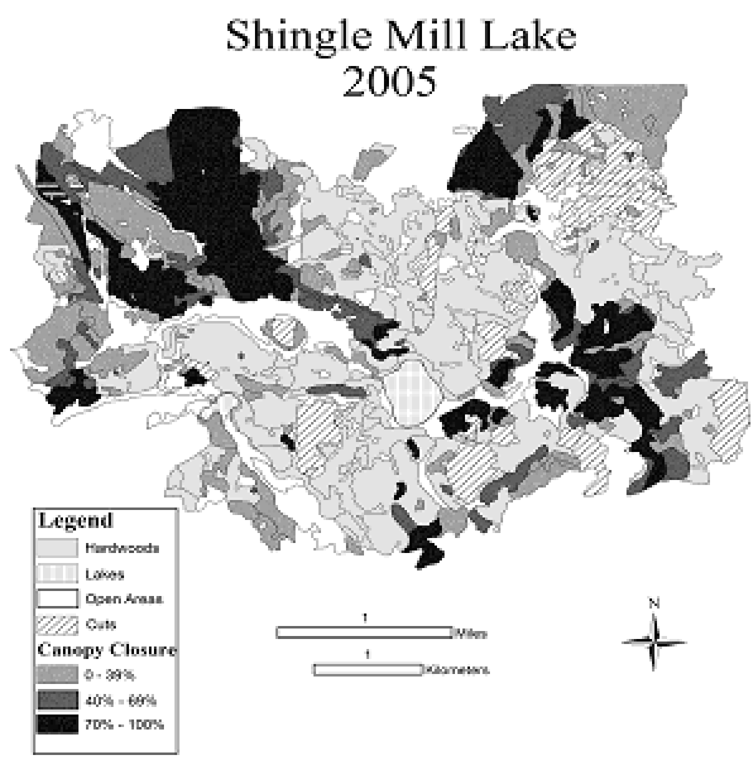

Figure 4. Continued.

MOOSE POPULATION DYNAMICS IN NORTHEASTERN MINNESOTA

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SUMMARY OF FINDINGS

A total of 114 moose (54 bulls and 60 cows) have been captured and collared since beginning the study in 2002. As of 31 March 2007, 77 collared moose (41 bulls and 38 cows) have died. Annual mortality rates varied between sexes and among years, and generally were higher than found elsewhere in North America. Pregnancy rates of captured cows were variable, but higher than found in northwestern Minnesota. Radio collared moose were used to develop a "sightability model" to correct observations during the annual aerial moose survey. This model will likely improve the accuracy and precision of the aerial survey.

INTRODUCTION

Moose (*Alces alces)* formerly occurred throughout much of the forested zone of northern Minnesota, but today, most occur within 2 disjunct ranges in the northeastern and northwestern portions of the state. The present day northeastern moose range includes all of Lake and Cook counties, and most of northern St. Louis County. In recent years, population estimates based on aerial surveys suggest that moose numbers are relatively stable. That moose numbers in northeast Minnesota have not increased in recent years is an enigma. Research in Alaska and Canada has indicated that adult non-hunting mortality in moose populations is relatively low. When these rates are used in computer models to simulate change in Minnesota's northeastern moose population, moose numbers increase dramatically, counter to the trend indicated by aerial surveys. Several non-exclusive hypotheses can be proposed to explain this result: 1) average non-hunting mortality rate for moose in northeastern Minnesota is considerably higher and/or more variable than measured in previous studies; 2) recruitment rates estimated from the aerial surveys and used in the model are biased high; and/or 3) moose numbers estimated by the aerial survey are biased low.

OBJECTIVES

- Determine annual rates of non-hunting mortality for adult and calf moose in northeastern Minnesota;
- Determine annual rates of reproduction in northeastern moose; and
- Determine the proportion of moose observed during aerial surveys and the factors that influence observability.

METHODS

Moose were captured in southern Lake County and southwestern Cook County, an area within the Laurentian Upland and North Shore Highland subsections of Minnesota's Ecological Classification System.

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In 2002, moose were captured by netgunning from a helicopter. We found this to be an inefficient method in our chosen study area. Thus in 2003–2005, moose were immobilized with a combination of carfentanil and xylazine delivered by a dart gun from a helicopter. A radio-collar was attached, and blood, hair, and fecal samples were collected from each moose. Beginning in 2003, a canine tooth also was extracted for aging.

Mortality was determined by monitoring a sample of up to 78 radiocollared moose. The transmitter in each radio-collar contained a mortality sensor that increased the pulse rate (mortality mode) if it remained stationary for more than 6 hours. When a transmitter was detected in mortality mode, we located the moose and conducted a necropsy to determine, if possible, the cause of death. Mortality rates were calculated using Kaplan-Meier survival functions (Pollock et al. 1989). During the first year of the study, the GPS location of each moose was determined weekly from the air. Beginning in March 2003, GPS locations were determined for one-half of the moose each week, and a mortality check was conducted on the remaining moose. After moose were located on 30 or more occasions, only mortality checks were conducted.

Pregnancy was determined from serum and fecal progesterone levels (Haigh et al. 1981, Monfort et al. 1993). Beginning in 2004, all collared cows were located in late May to determine the number of calves born, and the following April to determine calf survival. In addition, the presence/absence of a calf with a collared cow was determined, when possible during the telemetry flights.

A sightability model (Anderson and Lindzey 1996, Quayle et al. 2001) was developed using observations of the radiocollared moose during the 2004-2007 aerial moose surveys. During the survey, test plots were identified that contained 1 or more radiocollared moose. Each test plot was surveyed using procedures identical to those used in the operational survey. If the collared moose was observed within the plot, a suite of covariates including environmental conditions, group size, and visual obstruction were recorded. If the collared moose were not observed, they were located using telemetry, and the same set of covariates were recorded. Logistic regression was used to determine which covariates should be included in the sightability model.

RESULTS

No additional moose were captured in 2007. A total of 114 moose (60 cows and 54 bulls) have been captured and radiocollared in northeastern Minnesota between February 2002 and February 2005 (Figure 1).

As of 31 March 2007, 79 collared moose (41 bulls and 38 cows) have died. The cause of death in 33 cases could be identified (15 hunter kill, 2 poached, 7 train/ car/truck collision, 7 wolf predation, 1 natural accident, and 1 bacterial meningitis). Three additional deaths were censored from the study because they occurred within 2 weeks of their capture (1 wolf predation and 2 unknown). We were unable to examine the remains of 5 moose. Two died within the Boundary Waters Canoe Area Wilderness and in 3 cases, we only found the radio-collar. Thirty-eight collared moose appear to have died from unknown, non-traumatic causes. In 16 cases, scavengers had consumed the carcasses, but evidence suggested that predators might not have killed them. In the remaining 22 cases, most had little or no body fat (rump, kidney, abdominal, or heart), and were often emaciated. Moose dying of unknown causes died throughout year (Figure 2). To date, samples from unknown cases have tested negative for Chronic Wasting Disease, Rabies, Eastern Equine Encephalitis, and West Nile Virus. Sera from captured moose were tested for Bovine Viral Diarrhea, Borreliosis (Lyme's disease), Leptospirosis, Malignant Catarrhal Fever, Respiratory Syncytial Virus, Parainfluenza 3, Infectious Bovine Rhinotracheitis, Epizootic Hemorrhagic Disease, and Blue Tongue. All test results were negative except for Borreliosis (21 of 64 serum samples had positive titers 1:320 or greater). Follow up tests on tissues of moose harvested by hunters did not reveal any evidence that moose were infected with Lyme's disease.

Annual non-hunting and total mortality varied considerably among years and between sexes (Table 1). It should be noted that only 7 bulls were collared during 2002. In both sexes, 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).

Serum samples from 30 additional collared moose were tested for the presence of P. tenuis-specific antibodies using an enzyme-linked immunosorbent assay procedure (ELISA) (Ogunremi et al. 1999). Eighteen (15 cows and 3 bulls) of the 109 collared moose tested were sero-positive for antibodies against P. tenuis. Subsequently, 5 died of unknown causes, 3 were likely killed by wolves, 1 was killed by a hunter, and 1 is listed as capture related because it died within 2 weeks of capture. Only 3 skulls were examined for the presence of P. tenuis; results were positive in one case, negative in the other 2.

Pregnancy rate between 2002 and 2005 was 84% (*n*=56). In 3 of the 4 years, pregnancy rate ranged between 92 and 100%; in 2003, pregnancy rate was only 57%. This contrasts with a pregnancy rate of only 48% between 1996 and 1999 in northwestern Minnesota (Murray et al. 2006).

Survival of calves born to radiocollared cows remained constant since 2004 when calf surveys were initiated. In late April 2005, 12 of the 26 calves born to radiocollared cows in May 2004 were still alive (46% survival). In late April 2006, 41% of calves born the previous year were still alive. As of late January 2007, 56% of calves born in May 2006 were still alive. Annual calf survival in northwestern Minnesota averaged 66% (53-81%, Murray et al. 2006).

In January 2007, radio collared moose were located 49 times on test plots in the process of developing a sightability model. In 19 cases, the collared moose was observed using the standard survey protocol. In 30 cases, the collared moose was not observed, and telemetry had to be used to locate the collared moose. Since 2004, telemetry was required to locate 50% of the moose in test plots which suggests that, on average, only half of the moose are observed from the transects during the operational survey. Analyses of the data are ongoing and a final sightability model will be available for use during the 2008 moose survey.

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Table 1. Annual non-hunting and total mortality of collared moose. Number of collared moose in sample at beginning of calendar year is listed in parentheses.

Non-Hunting Mortality				Total Mortality			
Year	Bulls	Cows	Combined	Year	Bulls	Cows	Combined
2002	0% (7)	29% (17)	21% (24)	2002	14% (7)	29% (17)	25% (24)
2003	27% (27)	23% (33)	24% (60)	2003	33% (27)	23% (33)	28% (60)
2004	14% (23)	6% (35)	9% (59)	2004	35% (23)	6% (35)	17% (59)
2005	16%(35)	19%(43)	17%(78)	2005	24%(35)	19%(43)	23%(78)
2006	30%(25)	37%(35)	34%(60)	2006	41%(25)	37%(35)	39%(60)

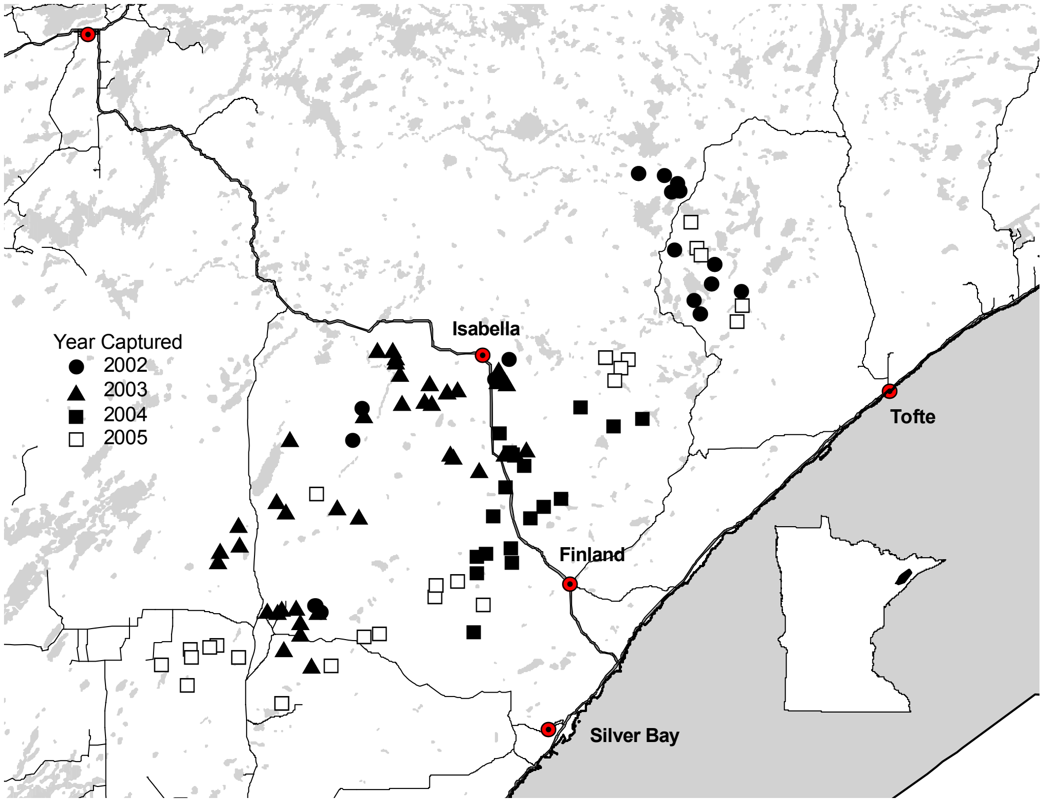

Figure 1. Capture locations of moose radio collared, 2002-2005.

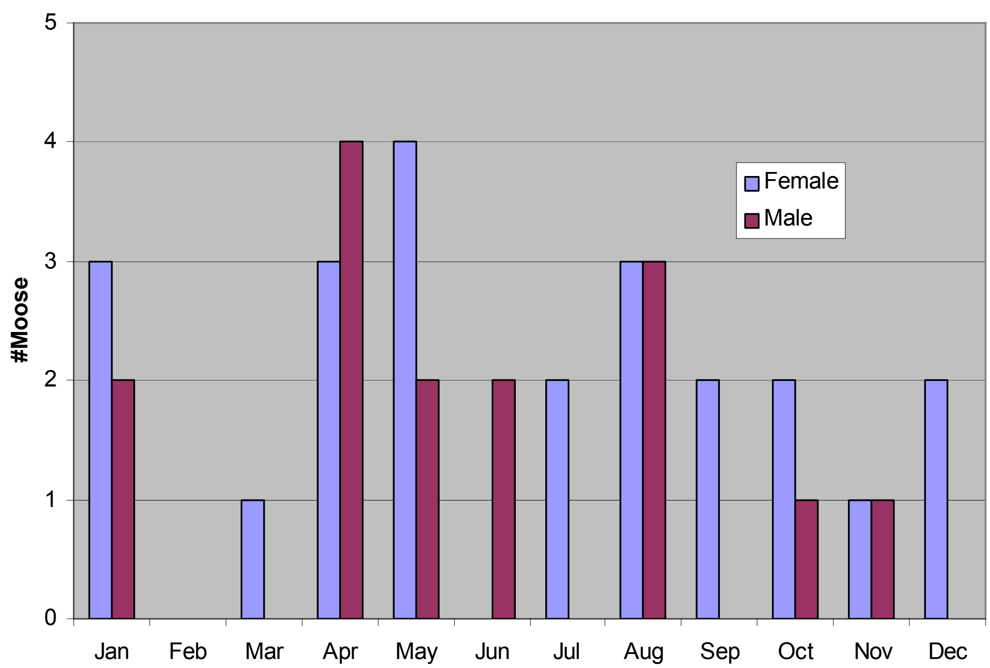

Figure 2. Timing of unknown mortality by sex in radiocollared moose in northeastern Minnesota study area.

ECOLOGY AND POPULATION DYNAMICS OF BLACK BEARS IN MINNESOTA

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SUMMARY OF FINDINGS

During April 2006–March 2007, we monitored 35 radiocollared black bears (*Ursus americanus*) at 3 study sites spanning the black bear's geographic range, north to south in Minnesota: Voyageurs National Park (VNP, northern), Chippewa National Forest (CNF; central), Camp Ripley (southern). Mortality data were obtained through collars turned in by hunters or collars tracked to carcasses. Hunting continues to be the largest source of mortality of collared bears, even though hunters were asked not to shoot bears with bright orange radiocollars. In fact, the hunting mortality rate of collared bears was higher this year (62% of collared bears killed in the CNF) than in any year since our study began in 1981. Reproductive output was highest in the southern study site and declined northward in response to diminishing food. All sites have exhibited largely synchronous reproduction, with high cub production occurring in odd-numbered years. This pattern continued in 2007.

INTRODUCTION

A paucity of knowledge about bear ecology and effects of harvest on bear populations spurred the initiation of a long-term telemetry-based bear research project by the Minnesota Department of Natural Resources (MN DNR) 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 within the state in sizes, growth rates, and productivity of bears, apparently related to varying food supplies, we started other satellite bear projects in different study sites. Each of these began as graduate student projects, supported in part by the MN DNR. 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 of varying ages. Camp Ripley is unhunted, but bears may be hunted when they range outside, which they often do in the fall, as the reserve is only 6–10 km wide. Oaks are far more plentiful here than in the 2 study sites further north. VNP, being a national park, is also unhunted, but again bears may be hunted when they range outside. Soils are shallow and rocky in the park, and foods are generally least plentiful of the 3 sites.

OBJECTIVES

- Monitor temporal and spatial variation in cub production and survival;
- Monitor rates and sources of mortality; and
- Compare body condition indices across sites and years (not covered in this report).

METHODS

Radiocollars with breakaway and/or expandable devices were attached to bears either when they were captured in barrel traps during the summer or when they were handled as yearlings in the den of their radiocollared mother. Limited trapping has been conducted in recent years. However, during December–March, all radio-instrumented bears were visited once or twice a year at their den site. Bears in dens were immobilized 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, or 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, with the cooperation of investigators from the University of Minnesota (Dr. Paul Iaizzo) and Medtronic (Dr. Tim Laske), heart condition was measured with a 12-lead EKG and ultrasound on a select sample of bears. Bears were returned to their den after processing. Reproduction was assessed by observing cubs in dens of radiocollared mothers. Cubs were not immobilized, but were removed from the den after the mother was drugged, then sexed, and weighed. We evaluated cub mortality by examining dens of these same 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 approximately once each month. 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.

RESULTS AND DISCUSSION

Since 1981 we have handled >800 individual bears and collared nearly 500. As of April 2006, the start of the current year's work, we were monitoring 41 collared bears: 17 in the CNF, 16 at Camp Ripley, and 8 in VNP. However, 6 bears dropped their collars so were lost from the sample.

Mortality

Legal hunting has been the predominant cause of mortality among radiocollared bears from all 3 study sites (Table 1). In previous years, 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 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 in dim light conditions. Nevertheless, 8 of 13 bears (62%) with functional collars in the CNF were killed during this year's hunt (Sep-Oct, 2006). This includes 1 bear that was wounded and lost. All were females, aged 1–7 years. This is the highest rate of hunter-caused mortality observed in this study over the past 26 years, leaving only 5 collared bears at this long-term study site. Additionally 1 of 8 collared bears was killed by a hunter outside Camp Ripley, and 3 other Camp Ripley bears could not be found after the hunt; either these bears were killed and not reported or their collar failed (wide-ranging unsuccessful aerial searches for them seemed to exclude the possibility that they simply moved far out of the area). No VNP bears were killed by hunters this year.

Other human-related mortalities included 1 bear whose cut-off collar was found in a river, and 2 male bears from Camp Ripley that were hit by vehicles on roads outside the reserve. A much higher proportion of the deaths at Camp Ripley (30% of those with known cause) were a result of collisions with vehicles than at the other 2 sites (4–6%).

Only 1 natural mortality was observed, a yearling in VNP. We do not know why this bear died (starvation seemed unlikely). However, it is interesting that of only 12 bears tracked in VNP over the past 2 years, 3 died of natural mortality, whereas no natural mortalities occurred at the other 2 study sites during this time (and few natural mortalities occurred earlier in the study; Table 1).

Reproduction

For the past decade, collared bears on all of our study sites had strong reproductive synchrony, with low cub production in even-numbered years and high production in odd-numbered years (Figure 1). This synchrony matches that exhibited in the age structure of the statewide bear harvest. It appears to have stemmed from a very poor food year in 1995, causing low cub production in 1996, followed by a good food year in 1996, yielding high cub production in 1997. Since then, all years have had average or above-average summer and fall foods, so the synchronous reproduction has persisted because nearly all bears have maintained a 2-year reproductive cycle. Of 13 mature bears checked in dens in March, 2007, 10 (77%) had cubs, 2 had yearlings, and 1 that was due to have cubs (because she had cubs in 2005) failed to produce. Reproductive synchrony appears to be strongest in VNP and least in Camp Ripley (Figure 1). In part this is because a large proportion of Camp Ripley bears produced their first cubs at 3 years old, which is out of synch with their mother.

Bears at Camp Ripley, where hard mast (especially oak) is abundant, grow faster and thus have an earlier age of first reproduction than at the other 2 study sites. This is reflected in the reproductive rates (cubs born/female) of 4–6 year-old females, which was twice as high at Camp Ripley as at VNP (where no bears produced cubs at 4 years old), and intermediate at CNF (Table 2). This north-south gradient was also apparent in the reproductive rates of older bears, due to fewer missed reproductive opportunities in Camp Ripley (the first bear that did not produce cubs on a 2-year cycle was observed this year) and 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. The proportion of females with cubs was lowest in VNP and highest in Camp Ripley (where it exceeded 50% as an artifact of sampling; Table 2).

Mean litter size was somewhat higher in the central CNF site (2.6 cubs/litter; Table 3) than at the other sites (2.3 cubs/litter; Tables 4–5). However, counting only litters where at least 1 cub survived 1 year, litter sizes were remarkably similar across areas for 7+ year-old bears (mainly multiparous mothers; Table 2). In all areas, litter size was smaller for younger females, nearly all of which were first-time mothers (Table 2). Notably, 2 collared bears produced litters of 5 cubs this year; of the 222 litters that we examined previously, only 2 other 5-cub litters were observed, both by the same female (in 1982 and 1984).

Only 1 bear was monitored through its age of senescence. She had her last cubs at age 25 (in 1999). This bear is still being monitored, now aged 33. This year, 1 bear in CNF had cubs at 22 years of age and 1 at VNP had cubs at 21.

Average sex ratio of cubs shortly after birth was slightly male-biased (52%) and virtually identical among all sites. Observed year-to-year variation in cub sex ratios (Tables 3–5) was likely attributable to sampling error. In all areas, the mortality rate of male cubs was higher than (1.5-2x) that of females. Overall, cub mortality appeared to be lower in CNF (18%; Table 3) than in the other 2 sites (26–28%; Tables 4–5). The difference, though, was not statistically significant (CNF vs. VNP and Camp Ripley combined, $P = 0.08$).

Cub production and cub mortality did not show an upward or downward trend during our 26 years of monitoring at CNF (or since 1999 at the other 2 sites). However, statewide bear harvests have shown an increasing proportion of yearlings, suggesting a changing statewide age structure, or possibly changing selectivity by hunters (with varying numbers of hunters).

FUTURE DIRECTION

We plan to continue monitoring bears on these 3 study sites, although sample sizes have been greatly diminished by the exceedingly high harvest of collared bears in the CNF this year. We are also initiating a new study site at the edge of the range in northwestern

Minnesota. This study will be led by a PhD student from the University of Minnesota. Our goal there is to assess the factors that may limit range expansion, including highly fragmented forested habitat, lack of agricultural crops that bears can eat, and human-related mortality. Bears will be outfitted with Global Positioning System (GPS) collars to document their fine-scale habitat use. Three bears whose dens were found by local people were collared this March. Comparisons will be made between these GPS-collared bears at the northwestern edge of the range and bears with GPS collars that have been monitored for the past several years at Camp Ripley, along the southern edge of the range.

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Table 1. Causes of mortality of radiocollared black bears ≥1 year old from the Chippewa National Forest (CNF), Camp Ripley, and Voyageurs National Park (VNP), Minnesota, 1981–2006. 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
Shot by hunter	219	10	10
Likely shot by hunter[a]	8	1	0
Shot as nuisance	22	2	1
Vehicle collision	12	7	1
Other human-caused death	9	0	0
Natural mortality	7	3	4
Died from unknown causes	3	1	0
Total deaths	280	24	16

[a] Lost track of during the hunting season. Does not include 3 bears lost at Camp Ripley in 2006 (see text).

Table 2. Reproductive rates (cubs/female), mean litter size, and proportion of females with cubs (for all measures, counting only litters in which at least 1 cub survived 1 year) in winter dens (March) in VNP (1997–2007), CNF (1981–2007) and Camp Ripley (1991–2007) (*n* = 4+ year-old female-years of observation). Reproduction increased from north (VNP) to south (Camp Ripley).

	VNP (*n* = 56)			**CNF** (*n* = 402)			**Camp Ripley** (*n* = 39)		
Age of female	Repro rate	Litter size	Prop w/ cubs	Repro rate	Litter size	Prop w/ cubs	Repro rate	Litter size	Prop w/ cubs
4–6 yrs	0.59	2.0	29	0.84	2.3	37	1.28	2.3	56
7–25 yrs	1.15	2.7	44	1.33	2.8	48	1.52	2.7	57
4–25 yrs	0.98	2.6	39	1.15	2.6	44	1.41	2.5	56

Table 3. Black bear cubs examined in dens of radiocollared mothers in or near the Chippewa National Forest during March, 1982–2007.

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%	—
Overall	172	449	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, 1992–2007.

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%	—
Overall	23	54	2.3	52%	26%

[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.

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.

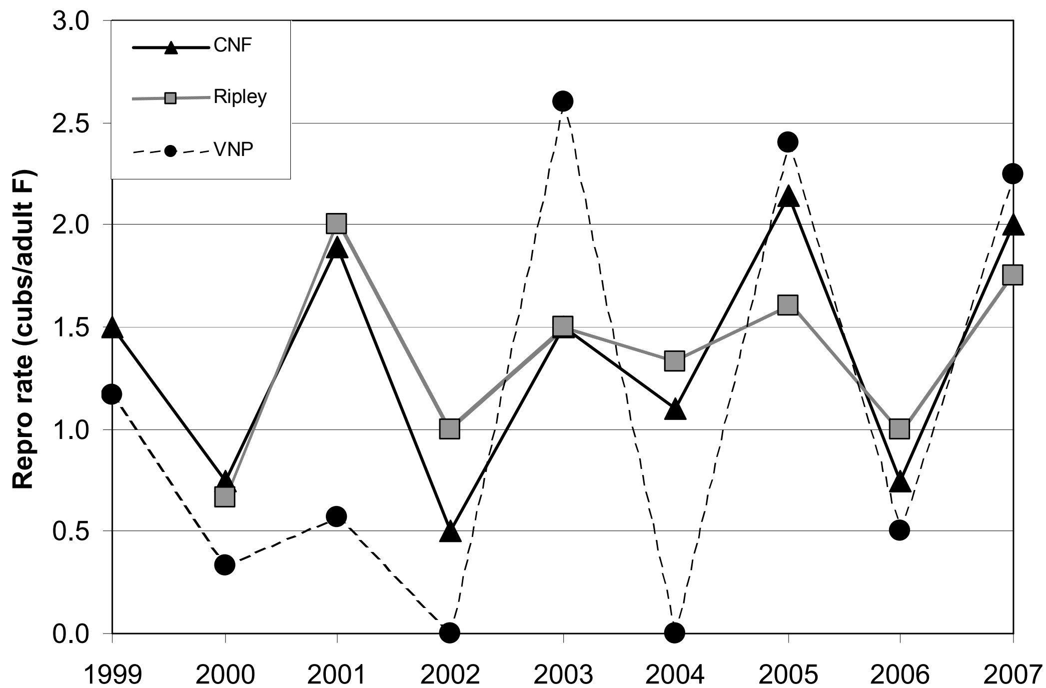

Figure 1. Reproductive rates (cubs per 4+ year-old female; counting only litters where at least 1 cub survived 1 year) of bears on the Chippewa National Forest (CNF), Camp Ripley, and Voyageurs National Park (VNP). All areas exhibited the same reproductive synchrony, although the pattern was most dramatic in VNP, at the northern extreme of the bear range, and weakest at Camp Ripley, at the southern edge of the range.

IDENTIFYING PLOTS FOR SURVEYS OF PRAIRIE-CHICKENS IN MINNESOTA

Michael A. Larson

SUMMARY OF FINDINGS

To explore potential improvements in surveys of prairie-chickens in Minnesota, I developed this study to determine landscape-scale characteristics associated with plots of land occupied by prairie-chicken leks and to evaluate potential within-year sources of variation in the probability of detecting a prairie-chicken lek, if one is present. The study area consisted of nearly the entire range of prairie-chickens in northwest Minnesota. Observers visited randomly selected Public Land Survey (PLS) sections (~259 ha) 3 times during April and early May of 2005 to detect leks. Confirmatory analyses indicated that wind speed and cloud cover were negatively correlated with the probability of detecting a lek. Road density was positively correlated with the probability of detection, but it was negatively correlated with the probability of a section being occupied by a lek. Exploratory analyses also revealed positive correlations between occupancy and both grass cover as a proportion of area and the area of all cover types considered as habitat and a negative correlation between occupancy and distance to the nearest known lek from the previous year. Approximately 13% of sections in the study area were occupied by a lek, but the precision of the estimated abundance of occupied sections was low ($\hat{Y}$ = 420, SD = 270).

INTRODUCTION

Nearly all methods for monitoring populations of greater prairie-chickens (*Tympanuchus cupido pinnatus*), including those currently employed by the Minnesota Department of Natural Resources (MN DNR), depend upon locating leks, or concentrations of the birds at their arenas for breeding displays (i.e., booming grounds), during spring. Surveying a statistically valid sample of leks requires identifying all areas where leks may occur and then sampling to find a number of plots occupied by active leks. The range of prairie-chickens in Minnesota covers approximately 10,000 km^2, so a major limitation to monitoring leks of prairie-chickens is determining where to survey within that range.

The availability of Geographic Information System (GIS) technology and databases of spatially explicit land cover have made it feasible to use landscape-scale habitat criteria to identify areas where leks may occur. Although land cover associated with prairie-chicken leks in Minnesota and Wisconsin have been quantified during previous studies (Merrill et al. 1999, Niemuth 2000, 2003), interpretation and application of those data are problematic. In particular, the previous studies were based on a case–control sampling design, which does not allow inferences about relative probabilities of occurrence (Keating and Cherry 2004). In addition, they did not select active leks randomly or verify nonuse at the randomly selected control locations.

Inferences about trends in the abundance of grouse throughout the state require statistically valid samples of survey locations from defined areas in which the species may occur. This study builds upon existing knowledge of landscape-scale habitat criteria that may be useful for identifying plots where prairie-chicken leks may occur, thereby dramatically reducing the area needed to be included in monitoring programs. It also serves as a pilot project for a new survey design that may prove to be more efficient than current survey methods for detecting changes in the abundance of prairie-chickens. Results of this study may benefit management programs for prairie-chickens by improving the quality of inferences drawn from spring surveys and developing resource selection functions for using landscape characteristics to estimate the relative probability of an area being occupied by a lek.

OBJECTIVES

- To determine landscape-scale characteristics associated with plots of land occupied by prairie-chicken leks in Minnesota; and
- To evaluate potential within-year sources of variation in the probability of detecting prairie-chicken leks in Minnesota.

METHODS

Study Area

Prairie-chickens occur in 3 distinct ranges in Minnesota. A study area was established in the northwest prairie-chicken range because the northwest range contained the largest population of prairie-chickens, was where the hunting permit areas were, and was the focus of all recent prairie-chicken monitoring efforts by the MN DNR. The study area included the northern 96% of the northwest range as defined by Giudice (2004) based upon land type associations of the Ecological Classification System (Figure 1). The size of the study area was limited only by a maximum distance of 90 km to the southeast of Moorhead, where the southernmost field technicians resided.

Notation

Methods for this study were based on analytical techniques for estimating the probability of site occupancy (MacKenzie et al. 2002). Throughout this report notation follows that of MacKenzie et al. (2002): ψ, probability that a sample plot is occupied by a lek; p, probability of detecting a lek within a sample plot, given that the plot is occupied; N, number of sample plots in a study area; T, number of surveys, or distinct sampling intervals during which all plots are visited once; the "hat" character (e.g., $\hat{\psi}$) denotes the estimated value of a quantity; and c, the probability of detecting a lek during visits that occur after a lek already has been detected within a plot (i.e., recapture).

Sampling Design

A sampling unit, or plot, was defined as a PLS section, most of which were 1.6- $\times$ 1.6-km squares (i.e., 259 ha = 1 mi^2). In portions of the prairie-chicken range in Minnesota some PLS sections were rectangular and much smaller than 259 ha. Variability in the size of plots was accounted for by the possible inclusion of habitat area within a plot as a covariate for ψ. The size of plots roughly corresponded to home range sizes of prairie-chickens during spring (<400 ha; Robel et al. 1970).

I applied a dual frame sampling design in which samples were drawn from a list frame consisting of plots known to have been occupied by a lek during 2004, and a much larger area frame consisting of the statistical population of plots to which the estimate of occupancy can be inferred (Haines and Pollock 1998). The area frame completely overlapped the list frame, so inferences were based upon the mutually exclusive overlap and nonoverlap domains. Dual frame sampling was appropriate for this study because an area frame was necessary for sample plots to be representative of other plots in the population, and the list frame was useful for focusing adequate sampling effort in plots where leks were known to have occurred recently. The locations of leks, especially those attended by more than a few males, are relatively consistent among years (Schroeder and Braun 1992), which makes them amenable to the use of a list frame.

An observer visited each sample plot once during each of T=3 consecutive biweekly periods from 4 April 2005 until 15 May 2005 (Svedarsky 1983). A visit consisted of a 20-minute interval between 0.5 hours before and 2 hours after sunrise (Cartwright 2000) during which a plot was surveyed with the purpose of detecting the presence of a lek (i.e., ≥2 male prairie-chickens) by sight or sound. The value of some time-dependent covariates of p were recorded during each visit, whereas the value of other covariates that vary only spatially were recorded only once for each plot. Observers also compared maps of land cover from the GAP level 4 database with actual land cover in sample plots and marked corrections on the maps. Most of the covariates of ψ were measured using a GIS, but some were verified by observers in the field.

Occupancy models often require an assumption that p is homogeneous (i.e., does not vary among plots). Using covariates of p in the model may ameliorate the negative effects of potential heterogeneity in p, but to prevent the sampling design from introducing heterogeneity, each observer visited a different set of plots during each biweekly survey period. Differences among observers in their ability to detect leks, therefore, would not be correlated with specific plots.

Data Analysis

I transformed the value of the covariates of ψ and p so they were within the interval [-9.9, 9.9], which precluded problems with numerical optimization that occur occasionally when using a logit link function. I developed sets of 8 and 14 *a priori* models to represent hypotheses about which covariates contributed to variation in p and ψ , respectively. Included in the set of models for ψ were 2 supported by previous studies (Table 1; Merrill et al. 1999, Niemuth 2003). I used Program MARK to fit occupancy models to the detection-nondetection survey data (MacKenzie et al. 2002). I used Akaike's Information Criterion adjusted for sample size (AIC_c) to calculate the Akaike weight (w), which is a relative weight of evidence for a model, given the data. I based all inferences on parameter estimates averaged over the best models that accounted for ≥95% of the Akaike weights (Burnham and Anderson 2002:150, 162). To estimate uncertainty in $\hat{p}$ and $\hat{\psi}$ given specific values of covariates I calculated limits of 95% confidence intervals on the logit scale then transformed them to the real scale (Neter et al. 1996:603). I combined estimates of $\hat{\psi}$ across sampling domains to estimate the number of plots occupied by prairie-chicken leks in the northwest range of Minnesota (Haines and Pollock 1998). Finally, I conducted an exploratory analysis by fitting models that were not specified *a priori*.

RESULTS AND DISCUSSION

I randomly selected n_{Area}=135 plots from the area frame (N_{Area}=3,137 plots), but 2 were excluded because they were not accessible by passable public roads and were not visited by observers (Figure 1). Inferences, therefore, were limited to portions of the study area that were accessible by public roads during spring. I randomly selected n_{List}=135 plots from the list frame (N_{List}=181 plots), 1 of which was excluded due to inaccessibility. Six of the plots selected from the area frame were also on the list frame, so $n_{nonoverlap}$=127 plots were in the nonoverlap domain (i.e., 127=135–2–6), and $n_{overlap}$=140 plots were in the overlap domain (i.e., 140=135–1+6).

The AIC-best *a priori* model for p was the "global" model, which contained all 16 covariates (i.e., 5 for observers, recapture, day of the study, time of day, temperature, wind speed, presence of precipitation, proportion of the sky obscured by clouds, road density, density of interior roads, proportion of suitable land cover types that were visible from roads, and proportion of suitable land cover types that were under snow or temporary water). It accounted for 97% of the AIC weight in the model set. The second-best model for p, labeled the "weather-1" model, had an AIC weight of 3% and contained 5 covariates (i.e., time of day, temperature, wind speed, precipitation, and cloud cover).

The 4 best occupancy models, which accounted for 98% of the AIC weight, included the global model for p (Table 2). Although they contained 21–25 parameters, only 6 model-averaged parameter estimates had confidence intervals that did not include 0 (Table 3). Wind speed, cloud cover, road density, and an observer effect were correlated with p (Figure 2; $\hat{\bar{p}}$ = 0.45, 95% CI=0.34–0.56). Road density was also correlated with occupancy (Figure 3). No land cover covariates, however, were correlated with occupancy within each sampling frame.

The probability of occupancy was 0.83 (95% CI=0.31–0.98) for plots in the overlap domain (i.e., from the list frame) and 0.09 (95% CI=0.01–0.46) for plots in the nonoverlap domain (i.e., from the area frame but not the list frame). Therefore, $\hat{\psi}$ = 420 (SD=270) plots in the study area were occupied by a lek. The lack of precision of $\hat{\psi}$ was acceptable, given the objectives of the study. The results, however, will be useful for evaluating the level of sampling effort necessary to estimate $\hat{\psi}$ with adequate precision at range-wide scales in the future.

I started the exploratory analysis by simplifying the model for p to include only the dominant 4 covariates rather than all 16 and by using combinations of covariates for ψ that may not have been included in the *a priori* set of models. The AIC-best occupancy model then included domain, habitat area, density of all roads, and density of paved roads as covariates for ψ. There was still much model-selection uncertainty, and the combined-1 and disturbance-1 models for ψ were only 2.0 and 3.1 AIC-units away from the best model.

I further refined the exploratory analysis by removing the domain covariate because it appeared to be an excellent discriminator between occupied and unoccupied plots and therefore potentially masking relationships between ψ and more informative landscape characteristics. Using a reduced model for p (K=5) and no domain covariate for ψ resulted in 3 models that accounted for >99% of the AIC-weight in the new model set. The model-averaged parameter estimates whose confidence intervals did not include 0 were those for the proportion of the plot covered in grass, distance to the next nearest lek observed the previous year, area of habitat in the plot, and density of roads (Figure 4).

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I thank N. G. Emery, T. E. Flatt, N. W. Gruber, R. J. Kleinschmidt, E. D. Peterson, and J. B. Winter for collecting data in the field; R. Wright and B. Sampson for GIS assistance; J. Fieberg and D. Pierce for helping me improve the project proposal; and L. Gilbert for map interpretation and data entry.

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Table 1. *A priori* models for explaining variation in the probability (ψ) of a sample plot being occupied by a prairie-chicken lek in Minnesota during spring of 2005.

Name	Covariates included
Habitat-1	Grass[a], Prairie[a], Sedge[a], Forest[a], Crop[a], Edge[b], Tree[c], Lek distance[d]
Habitat-2	Grass, Prairie, Forest, Edge, Lek distance
Habitat-3	Grass, Forest, Lek distance
Habitat-4	Grass
Disturbance-1	Homes[e], Road density, Density of interior roads, Density of paved roads
Disturbance-2	Homes, Road density
Combined-1	Grass, Forest, Lek distance, Habitat area, Homes, Road density
Combined-2	Grass, Forest, Lek distance, Homes, Road density
Combined-3	Grass, Forest, Lek distance, Habitat area
Lek distance	Lek distance
Forest	Forest
Habitat area	Habitat area
Niemuth	Grass, Sedge, Forest, Lek distance
Merrill	Forest, Homes

[a] Proportion of area of a plot in this cover type.
[b] Edge between forest and nonforest cover types.
[c] Presence of trees within suitable cover types.
[d] Distance from the nearest known lek during the 2004.
[e] Number of occupied human residences within the plot.

Table 2. Ranking of *a priori* models of occupancy of PLS sections by leks of greater prairie-chickens in northwest Minnesota during spring of 2005 (models with AIC-weight <0.001 not included).

Model[a]	K[b]	AIC_c	AIC-weight
p(global) ψ (disturbance-1)	22	608.9	0.677
p(global) ψ (combined-1)	25	612.0	0.143
p(global) ψ (disturbance-2)	21	612.6	0.107
p(global) ψ (combined-2)	24	613.9	0.056
p(weather-1) ψ (combined-1)	14	619.1	0.004
p(global) ψ (combined-3)	23	619.2	0.004
p(global) ψ (habitat-2)	24	619.7	0.003
p(global) ψ (lek distance)	20	620.4	0.002
p(weather-1) ψ (disturbance-1)	11	621.9	0.001
p(global) ψ (habitat-1)	27	622.5	0.001
p(global) ψ (habitat-4)	20	622.7	0.001
p(global) ψ (habitat-3)	22	622.8	0.001
p(global) ψ (domain)	19	622.9	0.001

[a] Models for p, the probability of detection, are described in the text; models for ψ, the probability of occupancy, are explained in Table 1.
[b] K = number of parameters, which includes 2 intercept terms—1 for the p portion of the model and 1 for the ψ portion.

Table 3. Parameter estimates averaged over the best 4 models of the occupancy of sample plots by leks of greater prairie-chickens in Minnesota during spring of 2005 and unconditional confidence intervals on the logit scale.

			95% confidence limits	
Probability	Parameter[a]	Estimated value	Lower	Upper
Detection	Intercept	-2.269	-6.213	1.675
	Observer 1	-0.474	-1.310	0.362
	Observer 2	-0.363	-1.183	0.457
	Observer 3	-0.201	-0.925	0.522
	Observer 4	-0.749	-1.563	0.065
	Observer 5	1.187	0.359	2.015
	Recapture	0.211	-0.562	0.984
	Day	-0.150	-0.424	0.124
	Time	-0.081	-0.638	0.476
	Temperature	-0.028	-0.083	0.026
	Wind speed	-0.885	-1.253	-0.516
	Precipitation	0.106	-0.720	0.932
	Cloud cover	-0.768	-1.438	-0.098
	Road density	0.469	0.044	0.894
	Interior roads	-0.114	-1.223	0.995
	Proportion visible	2.705	-1.318	6.728
	Ground cover	0.388	-5.925	6.701
Occupancy	Intercept	0.180	-2.368	2.728
	Overlap domain	3.861	2.420	5.302
	Homes	-0.511	-3.793	2.772
	Road density	-1.373	-2.289	-0.456
	Paved roads	-1.062	-2.848	0.725
	Grass	0.276	-0.722	1.273
	Forest	0.259	-1.681	2.200
	Lek distance	-0.349	-1.577	0.878
	Habitat area	0.221	-0.556	0.998

[a] Parameter names for models for p, the probability of detection, are described in the text; parameter names for models for ψ, the probability of occupancy, are explained in Table 1.

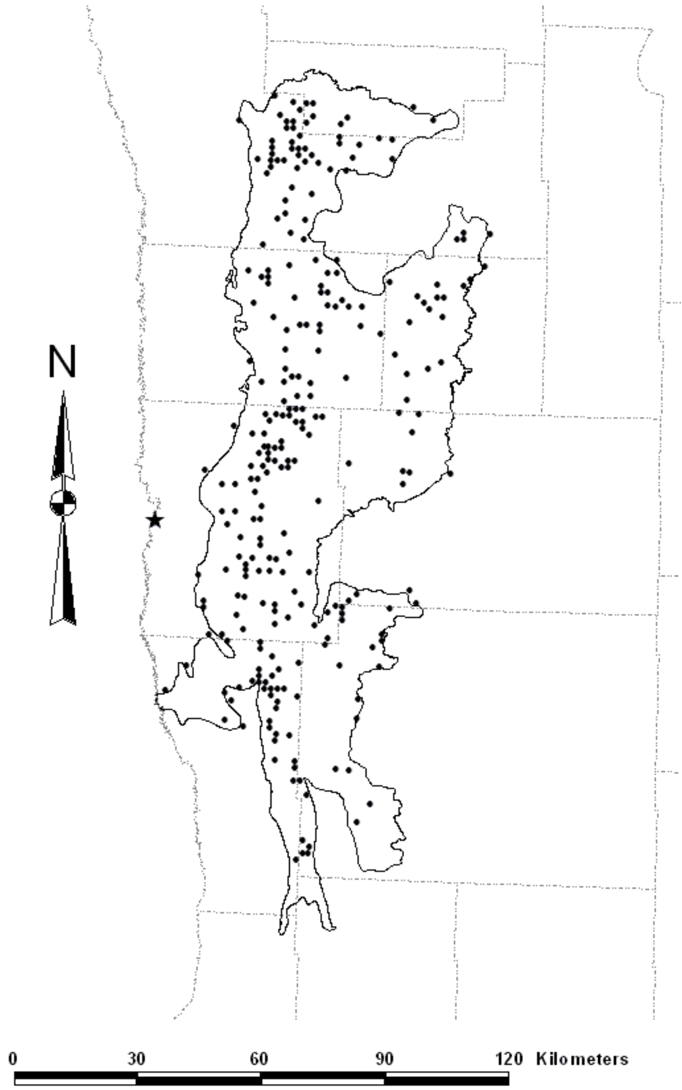

Figure 1. The northwest prairie-chicken range based on land type associations of the Ecological Classification System (solid line) relative to county boundaries (dashed lines) in western Minnesota. Sample plots (dots) were not selected from areas >90 km southeast of Moorhead (star).

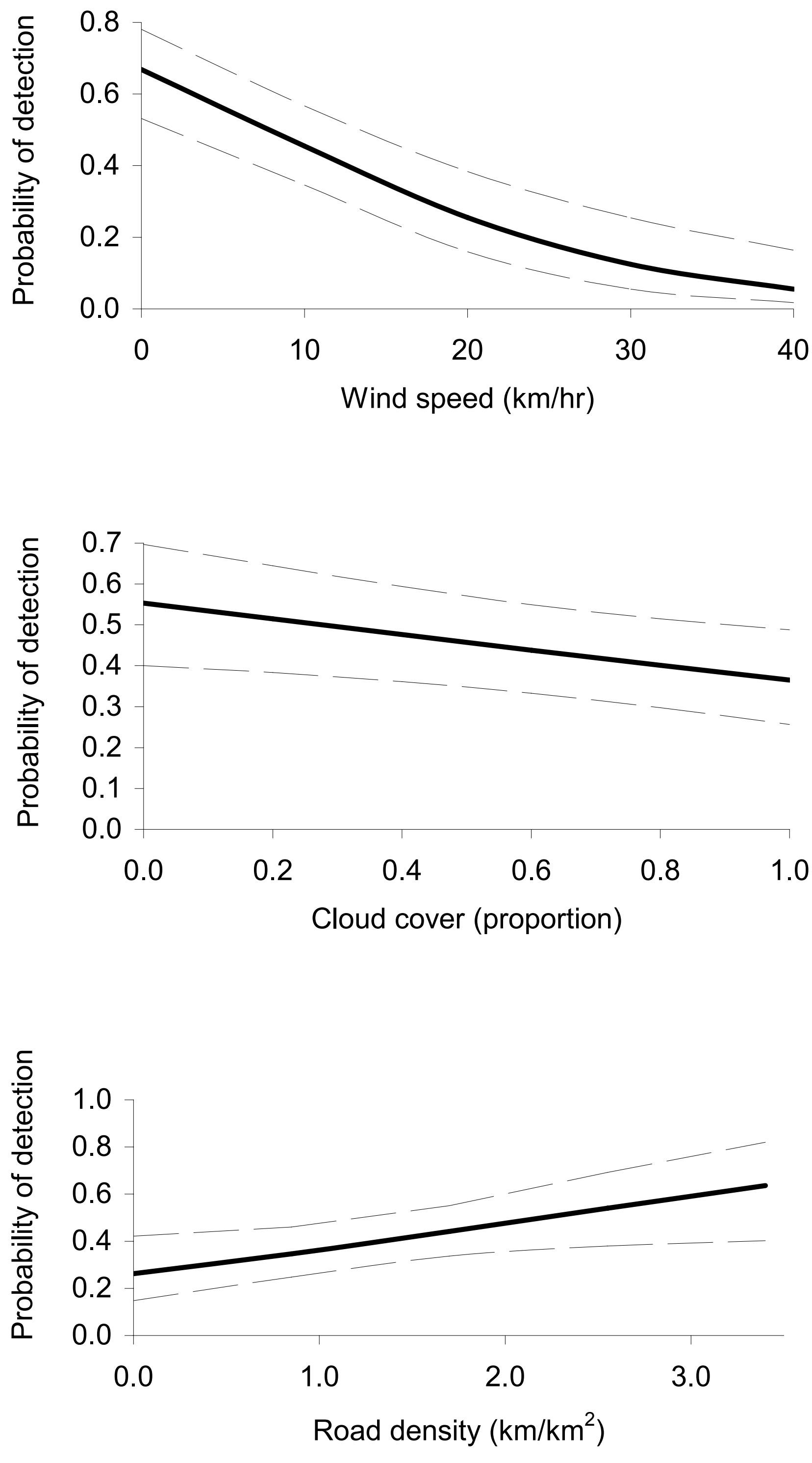

Figure 2. Model-averaged probabilities (and 95% confidence intervals) of detecting a prairie-chicken lek in sample plots in Minnesota during spring of 2005 over the range of observed values of 3 selected model parameters based on *a priori* models.

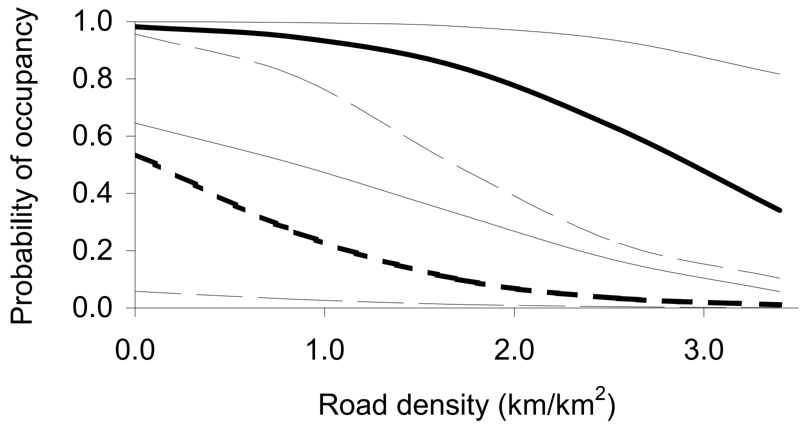

Figure 3. Model-averaged probabilities (heavy lines) and 95% confidence intervals (light lines) of a sample plot in Minnesota being occupied by a prairie-chicken lek during spring of 2005 over the observed range of road densities in the overlap domain (i.e., plots known to have contained a lek during 2004; solid lines) and nonoverlap domain (i.e., all other plots in the study area; dashed lines) based on *a priori* models.

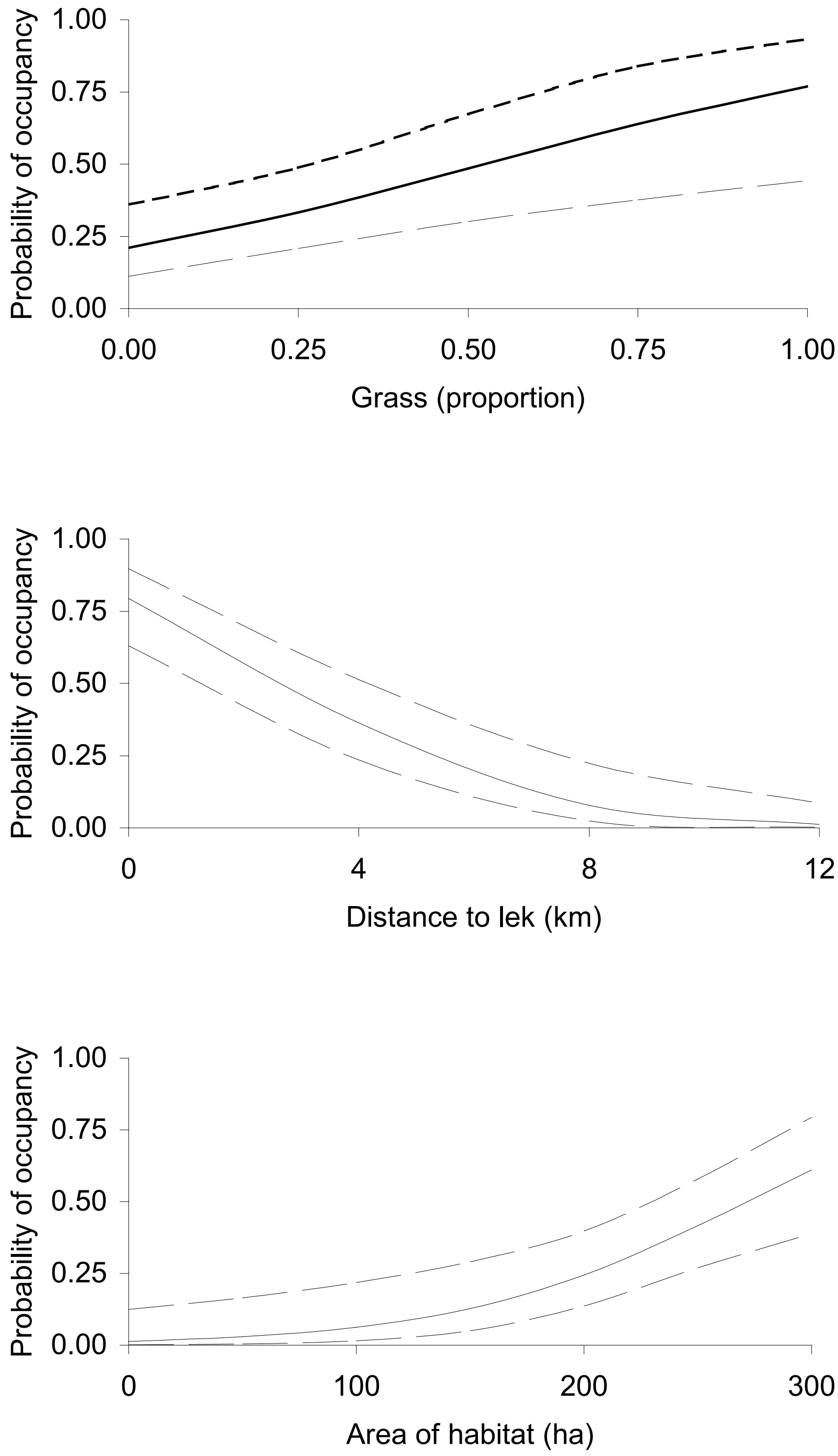

Figure 4. Model-averaged probabilities (and 95% confidence intervals) of detecting a prairie-chicken lek in sample plots in Minnesota during spring of 2005 over the range of observed values of 3 selected model parameters based on an exploratory analysis.

IDENTIFYING PLOTS FOR SURVEYS OF SHARP-TAILED GROUSE IN MINNESOTA

Michael A. Larson

SUMMARY OF FINDINGS

The justification, objectives, and methods for this project are identical to those for the prairie-chicken project, which is summarized separately. Collection of data for sharp-tailed grouse (Tympanuchus phasianellus campestris) occurred during spring 2006 in the eastern portion of their range in Minnesota, and is occurring during spring 2007 in the northwestern portion of their range in Minnesota (Figure 1). Data from both years will be analyzed together. Therefore, results for sharp-tailed grouse are not available at this time.

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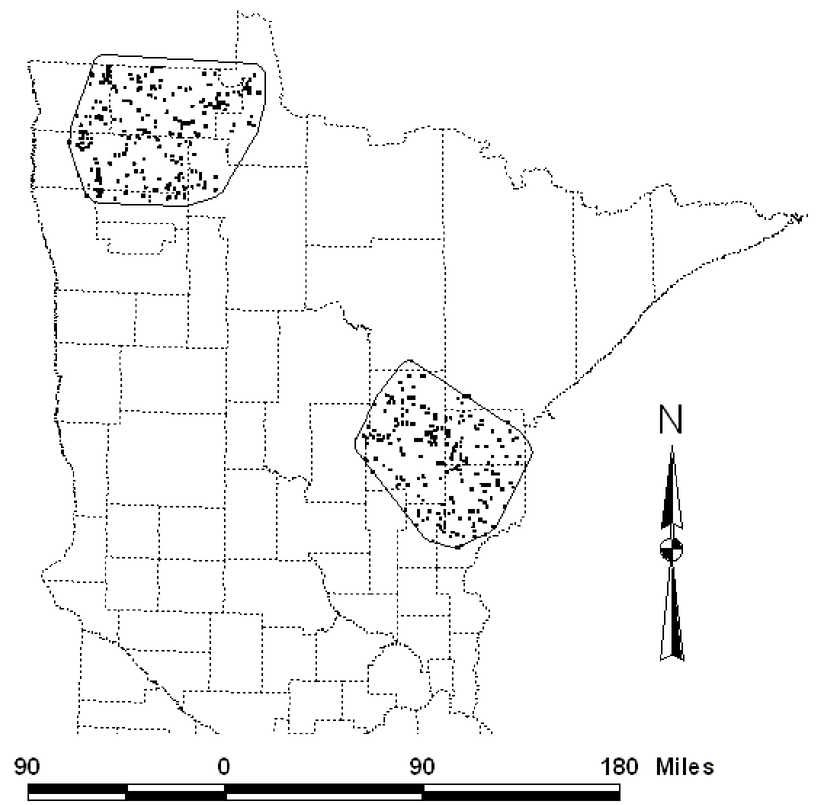

Figure 1. Sharp-tailed grouse study areas (solid lines) relative to county boundaries (dashed lines) in northern Minnesota. Sample plots (dots) were selected randomly after screening for road access and minimum habitat availability.

USE OF STABLE ISOTOPES OF CARBON, NITROGEN, AND OXYGEN IN STUDIES OF DIET AND NUTRITION OF MINNESOTA BLACK BEARS

Karen V. Noyce

SUMMARY OF FINDINGS

Hair from Minnesota black bears (*Ursus americanus*), as well as samples of common plant and animal bear foods, were analyzed for stable isotopes of carbon, nitrogen, and oxygen, to explore their potential utility in studies of bear diet and nutrition. Values for both $\delta^{13}C$ and $\delta^{15}N$ in plant foods were distinct from those in bear hair; values in animal foods (ants, fawns) were intermediate, overlapping somewhat with both bear hair and plant foods. Carbon isotope ratios in bear hair exhibited an increasing north-south gradient (moving from core to peripheral bear range) that likely reflected an increasing proportion of C4 plants (corn, cane sugar) in the diet. Bears known to have been feeding on corn (one denned in a corn field) had the highest $\delta^{13}C$ values. Hair collected from 2 yearling bears and cut into 4 equal lengths representing sequential 2-month periods during their cub year, showed abrupt increases in $\delta^{13}C$ values between mid- and late-summer, likely signaling a shift in diet to corn or other anthropogenic foods. There was no difference in $\delta^{15}N$ in bears from study areas with very different nutritional resources, but $\delta^{15}N$ was typically higher in cubs than in their mothers, consistent with their consumption of their mothers' milk. In contrast, $\delta^{18}O$ did not show the expected elevation in offspring relative to their mothers in 5 paired samples, nor did it provide clear indication of nursing/weaning history in cubs. However, $\delta^{18}O$ values in adults (n=6) all fell between 16.9 and 17.8, whereas values from whole hair or portions of hair collected from yearling bears spanned a much wider range (14.4–19.3). Further work to improve our understanding of $\delta^{18}O$ in bear tissues might yet reveal ways that it can help decipher infant nutritional patterns in bears.

INTRODUCTION

The use of stable isotopes in wildlife studies has expanded rapidly in recent years as new applications for these methods have been developed. Interpreting stable isotope ratios involves comparing the ratio of 2 isotopes of an element (e.g.^{13}C and ^{12}C) in the tissues of an animal to their constant ratio in the inorganic environment in order to deduce information about the animal's feeding behavior (the comparison is expressed as $\delta^{13}C$). Although isotopes of the same element behave in the same manner in metabolic reactions in animal tissues, because of their different masses, heavy and light isotopes react at different rates and are thus assimilated or released into the environment at different rates. For example, plants assimilate ^{13}C and ^{12}C in different proportion to their abundance in the environment, and 2 major groups of plants (C3 and C4), using different photosynthetic pathways, assimilate ^{13}C and ^{12}C in characteristically different ratios from each other. Plants in the C3 group include temperate forest species, such as Minnesota's native forest vegetation, whereas C4 plants are mostly of tropical origin and include the common agricultural crops, corn and sugar cane. Carbon isotope ratios in animal tissues can indicate the relative proportion of C3 and C4 plants in the assimilated diet. This application of stable isotope analysis has recently been used to compare the feeding histories of back-country versus nuisance bears in Japan (Mizukami et al. 2005).

Because of differential uptake of nitrogen isotopes in animal tissues, the $\delta^{15}N$ signature changes by a relatively constant factor with each successive step in the trophic chain; i.e., $\delta^{15}N$ is higher in plants than in the environment, higher in herbivores than in plants, and higher in carnivores than in herbivores. This characteristic has been used to compare the relative importance of plant versus animal protein in the diets of omnivores, such as black bears and grizzly bears in British Columbia (Hobson et al. 2000) and grizzlies in different parts of North America (Mowat and Heard 2006).

Stable isotopes of carbon and nitrogen have also been used to investigate weaning and fasting in mammals (polar bears: Polischuk et al. 2001). Though not previously used in wildlife studies, anthropologists have found $\delta^{18}O$ in tooth enamel and dentine to be helpful in deducing weaning practices in ancient cultures (Wright et al. 1999). Oxygen isotopes show a small trophic effect similar to nitrogen, such that ^{18}O in body fluids is more enriched than in environmental (meteoric) water. Thus, if an animal obtains most of its water from body fluids, such as milk, $\delta^{18}O$ will be higher than if its water is imbibed from the environment. In combination with $\delta^{13}C$ and $\delta^{15}N$, this can be used to surmise the age at which supplemental foods enter the diet of infants and the age of weaning.

I undertook exploratory work to examine the feasibility of using hair samples from denning black bears to investigate several aspects of their diet and nutritional ecology. The composition of hair reflects an animal's body chemistry at the time it is growing, thus a shaft of hair sequentially sectioned, can provide a record through time of changes in isotopic composition. The hair of yearling bears in the den reflects their diet from time of birth the previous winter until the cessation of hair growth in late summer or early fall. Adult hair collected in the den represents a shorter period of time, probably about May–September of the previous year (Mizukami et al. 2005), as hair growth for the year's molt does not start until mid-spring. I collected bear hair from a variety of bears and locations, as well as samples of common plant and animal foods of bears. For this work, sample sizes were small, but the scope of samples broad, in keeping with the intent of the study as preparatory to planning future, more comprehensive investigations.

OBJECTIVES

My objectives were to investigate the feasibility of the following:

- determining the importance of anthropogenic foods in the diet of bears from different regions of Minnesota;
- correlating nutritional condition of individual bears with particular temporal dietary (isotopic) patterns;
- determining dietary composition, using mixing models, based on isotopic signatures of bear hair and major bear foods; and
- comparing weaning histories of cubs exhibiting widely different nutritional status distinguishing hair from cubs from that of older juveniles and adults.

METHODS

I sampled male bears residing at the southern fringe of the bear range and females from 3 study areas in other parts of Minnesota including: Camp Ripley, located in central Minnesota in the southern transition area between forest and agriculture; the Chippewa National Forest (CNF), located 150 km to the north in the center of the bear range; and Voyageurs National Park (VNP), located along the Canadian border, another 130 km to the north (Table 1). I collected 2 mother-daughter pairs of samples from females denned with their yearlings in each study area. Hair from 3 yearlings (2 from CNF, 1 from Camp Ripley) was subsampled by dividing it into 4 equal lengths, each portion representing a 2-month time period between February–September during the bear's previous year. Samples of bear foods were collected from either the CNF or Camp Ripley areas.

Samples of bear foods were air-dried on low heat in a convection drying oven, then ground to a fine powder using mortar and pestle. Vegetation samples comprised a mixture of equal parts clover (*Trifolium* spp), wild calla (*Calla palustris*), jewelweed (*Impatiens biflora*), and new leaves of quaking aspen (*Populus tremuloides*). Fruit samples included strawberries (*Fragaria* spp) and raspberries (*Rubus idaeus*), mixed in equal parts. Samples of bear hair were rinsed 3 times in a 2:1 chloroform-methanol solution to remove oils, and then air-dried, cut into small segments and mixed. All samples were sent to the Colorado Plateau Stable Isotope Laboratory at Northern Arizona University, where they were weighed and analyzed, using a Thermo-Finnigan Delta^{plus} Advantage gas isotope-ratio mass spectrometer, interfaced with a Costech Analytical ECS4010 elemental analyzer. All samples were analyzed for $\delta^{13}C$ and $\delta^{15}N$; mother-daughter pairs were analyzed for $\delta^{18}O$ as well.

RESULTS AND DISCUSSION

There was distinct separation in both $\delta^{13}C$ and $\delta^{15}N$ values between bear hair and plant foods, except for 1 of 2 mixed-berry samples, in which $\delta^{15}N$ was anomalously high (Figure 1). In other plant samples, $\delta^{13}C$ and $\delta^{15}N$ were lower than and non-overlapping with values in bear hair. The 2 samples of mixed green vegetation yielded the lowest values of $\delta^{13}C$ (mean of -28.8 $\delta^{13}C$, versus mean –26.6 $\delta^{13}C$ for berries and nuts). All plant samples were below –25.9. Among plant foods, $\delta^{15}N$ was lowest in 2 hazelnut samples. Overall, $\delta^{15}N$ ranged from –2.67 to 0.88 in plant material, except in 1 berry sample where $\delta^{15}N$ was 3.14, in sharp contrast to the other berry sample.

Bear hair samples spanned a broader range of values; however, in all cases, $\delta^{13}C$ was >–25 and $\delta^{15}N$ was >3. Animal foods of bears were intermediate: ants were lower in both $\delta^{13}C$ and $\delta^{15}N$ than bear hair and higher in $\delta^{13}C$ and $\delta^{15}N$ than most plant foods. Samples from 2 white-tailed deer (*Odocoileus virginianus*) fawns of different ages were very different in $\delta^{15}N$; values in hair and muscle from a 20-day-old fawn (Figure 1, points 1, 2) were considerably higher than in a 12-day-old fawn (Fig 1, points 3,4). For $\delta^{13}C$, hair samples from the 2 fawns were similar, but muscle samples differed. In both fawns, $\delta^{13}C$ was higher in muscle than in hair.

Bear hair showed distinct differences by location (Figure 2); $\delta^{13}C$ in bears living in or south of Camp Ripley was higher than in any bears from the CNF or VNP. The highest values were in 2 adult males living in the southern fringe of the bear range. One was known to have fed on corn extensively before denning (in the cornfield). A third male living at the edge of the bear range, along with Camp Ripley bears, yielded somewhat lower $\delta^{13}C$ values, but all were above –21; one Camp Ripley female was also known to have fed on corn in the late summer. Bears from the CNP all had $\delta^{13}C$ below –21 and VNP bears ≤-23.

Results for mother-daughter pairs were varied (Table 2). There were no consistent relationships between adults and their nursing offspring in any stable isotope ratios, though whole-hair $\delta^{15}N$ tended to be higher in offspring than in their mothers (4 of 5 cases). There were no consistent temporal trends in subsampled yearling hair that appeared indicative of time of weaning. Moreover, we did not see the expected increase in $\delta^{18}O$ in hair of daughters relative to that of mothers. In fact, particularly in one case (Table 2), $\delta^{18}O$ in the daughter (bear #21) was lower than that of her mother (bear #13), even early in the season, when she should have been nursing.

Nevertheless, in 2 of 3 cases where we subsampled yearling hair (1 Camp Ripley and 1 CNF yearling), $\delta^{13}C$ values showed a similar marked and sudden change between the third and fourth time periods, representing approximately June-July and August-September (Table 2, Figure 3). The first 3 sampling periods (the distal portions of the hairs) were similar in $\delta^{13}C$ (-22 to -23), but the proximal section (the last to grow before denning) jumped to –20.16 in the CNF cub and –16.81 in the Camp Ripley cub. A second CNF cub did not show this pattern; this yearling was extremely undernourished, and it may be significant that its $\delta^{18}O$ values throughout the season were lower than those of the other CNF yearling, who was of average body condition and further below those of the Camp Ripley yearling, who was the largest and fattest of the three.

DISCUSSION AND FUTURE DIRECTION

These preliminary results seem to indicate that $\delta^{13}C$ may be a sensitive and useful index to regional and temporal differences in the reliance of Minnesota bears (and perhaps other species) on some types of anthropogenic foods. In this case, they appear to indicate that corn and/or other human-related foods (e.g. corn-based dog food, processed foods containing cane or corn sugar) increasingly enter the food chain of bears from relatively unpopulated parts of the northern bear range (VNP) to the populated and agricultural south. Though consistent with what we know of bear feeding behavior across the state, the consistency of the trend in all the samples was somewhat surprising — every one of the Camp Ripley and more southerly bears appeared to include significantly more such foods in their diet than any of the CNF and VNP bears. It would be prudent to rule out, through further sampling, any other potential sources of this consistent geographic gradation in $\delta^{13}C$.

Neither $\delta^{18}O$ nor $\delta^{15}N$ provided an easy or clear-cut way to infer nutritional condition of bears, interpret the weaning history of cubs, or to distinguish between cubs and adults based on hair samples. There were, nevertheless, some intriguing suggestions in this small exploratory data that may be revealed more fully with further investigation and larger sample sizes from bears with known histories. For example, if $\delta^{18}O$ is consistently more labile in nursing cubs than in adults, this could perhaps provide a first screen for distinguishing between cubs and adults based on hair samples. The counterintuitive occurrence of higher $\delta^{18}O$ values in mothers than in their nursing cubs raises questions about whether the metabolism of body fat, which produces water, in post-denning mothers may influence values in females of different body condition. Though current data are insufficient to attempt diet reconstruction using mixing models due to the variation in stable isotope signatures of similar types of food, more comprehensive sampling and analysis of food items should make this possible. Finally, further sampling of mother-offspring pairs and studies using captive bears can help to better document hair growth patterns, enabling clearer interpretations of isotope data.

ACKNOWLEDGMENTS

Many people helped with the handling of bears and collection of hair samples, especially D. Garshelis and P. Coy, of the Minnesota Department of Natural Resources Forest Wildlife Populations and Research Group. I thank C. Humpal for her help preparing samples for analysis and R. Doucett, of the Colorado Plateau Stable Isotope Laboratory, for advice and performing analyses.

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Table 1. Description of samples collected in Minnesota for isotopic analysis.

Sample ID	Sample material	Location collected	Analyses performed
1a,b	Green vegetation mix	CNF	C,N
2a,b	Acorns	CNF, Garrison	C,N
3a,b	Hazelnuts	CNF	C,N
4a1,4b1	Deer fawns (hair)	12 and 20 days old	C,N,O,
4a2,4b2	Deer fawn (meat)	" " "	C,N,O
5a,b	Mix of berries	Misc.	C,N
6a,b	Ants	Camp Ripley	C,N
7	Bear - ad. M (denned in cornfield)		C,N
8	Bear – ad. M	Long Prairie	C,N
9	Bear – ad. M	Buckman	C,N
10	Bear – ad. F (fed in corn)	Camp Ripley	C,N,O
11	Bear – ad. F	Camp Ripley	C,N,O
12	Bear – ad. F	CNF	C,N,O
13	Bear – ad. F	CNF	C,N,O
14	Bear – ad. F	VNP	C,N,O
15	Bear – ad. F	VNP	C,N,O
16	Bear – ad. F	CNF	C,N
17	Bear – ad. F	CNF	C,N
18	Bear – yrl. F, daughter of #10	Camp Ripley	C,N,O
19a,b,c,d	Bear – yrl. F, daughter of #11	Camp Ripley	4 subsamples C,N,O
20a,b,c,d	Bear – yrl. F, daughter of #12	CNF	" " " " "
21a,b,c,d	Bear – yrl. F, daughter of #13	CNF	" " " " "
22	Bear – yrl. F, daughter of #14	VNP	C,N,O
23	Bear – yrl. F, daughter of #15	VNP	C,N,O

Table 2. Ratios of stable carbon, nitrogen, and oxygen in mother-daughter pairs of black bears in Minnesota.

	$\delta^{13}C$		$\delta^{15}N$		$\delta^{18}O$	
	Mother	Daughter	Mother	Daughter	Mother	Daughter
Bears 11,19						
Feb – March		-21.96		4.90		18
April – May		-22.18		4.73		19.3
June – July		-22.5		3.70		19.1
Aug – Sept		-16.81		3.03		16.8
Whole hair	-20.34	-20.86	3.44	4.09	17.8	18.3
Bears 12,20						
Feb – March		-22.95		5.08		16
April – May		-22.14		5.25		16.8
June – July		-22.02		4.95		16.6
Aug – Sept		-20.16		5.57		17.3
Whole hair	-21.75	-22.32	4.38	5.21	17.3	16.7
Bears 13,21						
Feb – March		-22.67		4.89		14.8
April – May		-23.1		3.02		14.4
June – July		-23.63		3.45		15.4
Aug – Sept		-24.37		5.15		
Whole hair	-22.84	-23.44	4.34	4.12	17.1	
Bears 14,22						
Whole hair	-22.90	-24.44	6.86	7.67	17.2	15.4
Bears 15,23						
Whole hair	-22.67	-22.95	5.11	5.81	16.9	17.3

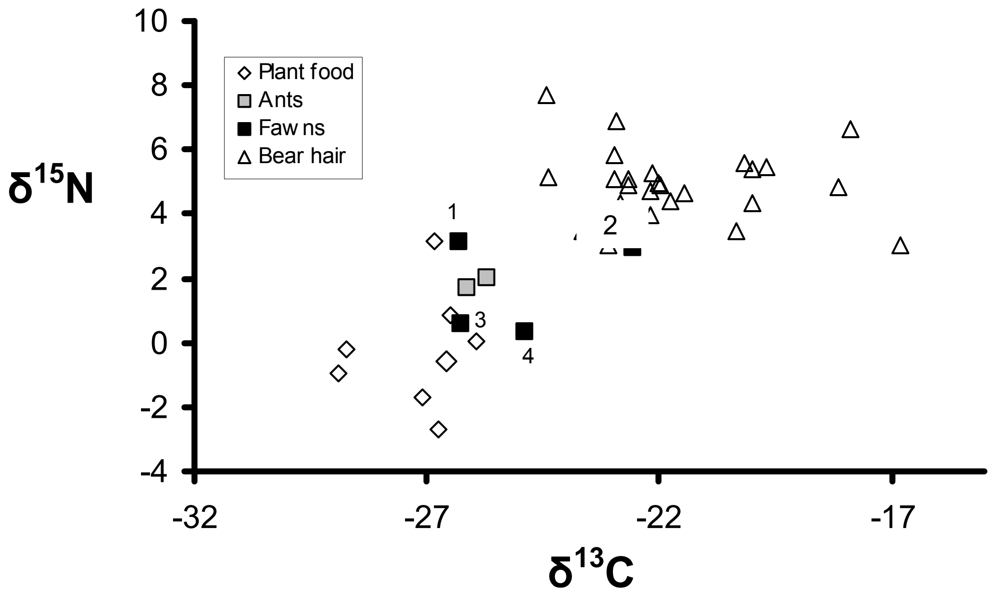

Figure 1. Stable isotope signatures of $\delta^{13}C$ and $\delta^{15}N$ in samples of common wild bear foods and all hair samples from Minnesota black bears. (Numbered samples correspond to: 1,2 – hair and muscle, respectively, from 20-day-old white-tailed deer fawn; 3,4 – hair and muscle from 12-day-old fawn).

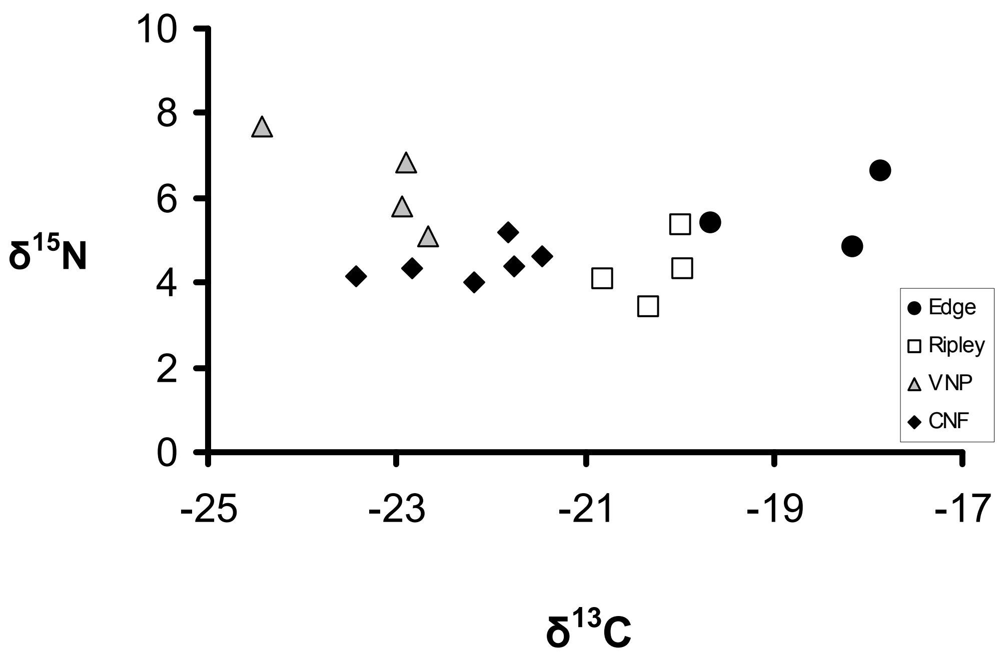

Figure 2. Stable isotope signatures of $\delta^{13}C$ and $\delta^{15}N$ in hairs collected from denning bears that resided in different parts of Minnesota's bear range. (In cases where hairs were subsampled to obtain chronologic results, values on this graph represent the mean of the subsamples.)

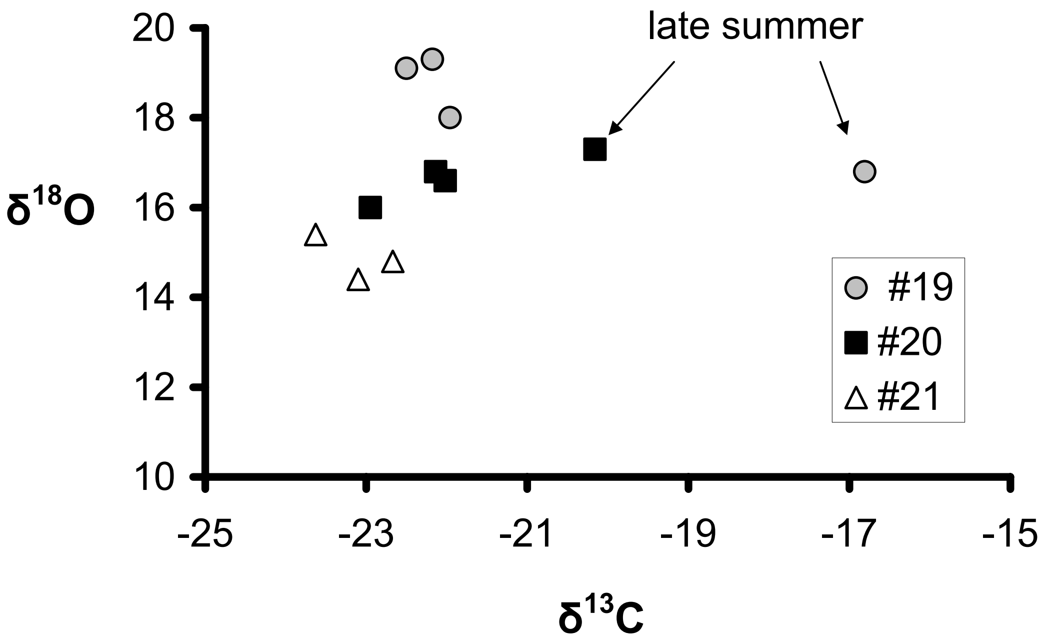

Figure 3. Stable isotope signatures of $\delta^{13}C$ and $\delta^{18}O$ in hairs collected from yearling bears. Hairs were divided into 4 equal portions, representing different 2-month periods in the bear's life from approximately 0 – 8 months old. Due to thinning of the hair shaft at the proximal end, there was insufficient sample to analyze for late summer stable isotope signatures for bear #21.

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

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

SUMMARY OF FINDINGS

Bovine tuberculosis (TB) was discovered in 5 cattle operations in northwestern Minnesota in 2005. Two additional cattle herds were found infected in 2006. To date, all of the infected cattle herds have been depopulated and the Board of Animal Health (BAH) has continued an investigation of herds in the area as well as conducted a statewide surveillance effort. The strain has been identified as one that is consistent with bovine TB found in cattle in the southwestern US and Mexico. In November 2006, the Minnesota Department of Natural Resources (DNR) conducted bovine TB surveillance of hunter-harvested white-tailed deer (*Odocoileous virginianus*) within a 15-mile radius of the infected farms. Results indicated that 5 of the 942 deer tested positive for bovine TB; estimated disease prevalence of 0.5% (SE = 0.2%). All infected deer were harvested within 5 miles of Skime, Minnesota, which is in close proximity to 4 of the infected livestock operations. The United States Department of Agriculture (USDA) also required a statewide assessment of bovine TB in wild deer, thus 4,058 additional samples were collected from hunter-harvested deer outside the surveillance zone and tested for the disease; none of these deer were positive for TB. In response to additional deer found infected with bovine TB since 2005, the DNR created a Bovine TB Management Area in northwestern MN to help focus future disease management efforts. Further, a recreational feeding ban, covering 4,000 mi^2 in northwestern MN, was instituted in November 2006 to help reduce the risk of deer to deer transmission of the disease. Also, the Minnesota Legislature passed an initiative that allocated $54,000 for deer-proof fencing materials for livestock producers within 5 miles of a previously infected herd; DNR is currently managing that program. The DNR will continue to conduct hunter-harvested surveillance in fall 2007 to monitor infection in the local deer population, and consider more aggressive management actions (e.g., sharpshooting deer in key locations) to address concerns of deer becoming a potential disease reservoir.

INTRODUCTION

Bovine tuberculosis (TB) is an infectious disease that is caused by the bacterium *Mycobacterium bovis (M. bovis)*. Bovine TB primarily affects cattle, however, other animals may become infected. Bovine TB was discovered in 5 cattle operations in northwestern Minnesota in 2005, and 2 additional herds in 2006. Entering into the fall 2006, 2 wild deer had 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 (*Odocoileus 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, which artificially congregates deer and increases the likelihood of disease transmission.

Bovine TB is a progressive, chronic disease, that 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

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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, peasized 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 rare. Most human tuberculosis is caused by the bacteria *M. tuberculosis*, which is spread from person to person and rarely infects animals.

METHODS

A surveillance area was developed that encompassed a 15-mile radius around Skime, Salol, and Grygla, Minnesota centering on the locations of the infected livestock operations. A sampling goal was determined to ensure 95% confidence of detecting the disease if prevalent in >1% of the deer population. Given the large geographic area and abundance of deer, the goal was to collect approximately 1,000 samples within the surveillance zone. Additionally, the USDA required a statewide assessment of bovine TB prevalence in 4,000 deer harvested outside of this surveillance zone; thus, registration stations were selected statewide based on deer density and distribution to collect this information (Figure 1). Sampling was conducted during the first 2 weekends of the November 2006 firearms deer hunting season and all samples were voluntarily submitted by hunters.

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 entered into a gun raffle.

Tissue collection procedures included a visual inspection of the chest cavity of the hunter-killed deer. Six cranial LN's (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 Laboratory 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.

RESULTS AND DISCUSSION

In fall 2006, we collected 5,000 samples from hunter-harvested deer; 4,058 outside and 942 inside the bovine TB surveillance area, respectively (Figure 2). This included 13 whole carcasses that were confiscated from hunters due to the presence of suspicious lesions in the chest cavity or lymph nodes; yielding 4 deer positively infected with bovine TB. An additional positive deer was detected that did not have obvious lesions in the chest cavity, but abscesses were found in the lymph nodes. All infected deer were harvested approximately 5 miles from Skime, Minnesota (Figure 3). No deer sampled through the statewide surveillance effort were found positive for bovine TB outside of the bovine TB surveillance area. The strain of bovine TB from the infected deer matched the strain isolated from the infected cattle herds in the surveillance area and was consistent with bovine TB strains commonly found in the southwestern U.S. and Mexico. The proximity of the infected deer to infected cattle herds, the strain type, and the fact that disease prevalence (0.5%) is low, supports our theory that this disease spilled-over from cattle to wild deer in this area of the state. However, the increased number of TB-infected deer found this fall, combined with a wider geographic distribution of these infected animals, led the DNR to create a new Bovine TB Management Zone which encompasses a 10-mile buffer around all infected deer discovered to date (Figure 4). Included in this new zone is a core area, which is a 2-mile buffer around all infected deer. This new management zone and its core will help the DNR focus future management actions to help manage the disease in the local deer population.

In November 2006, a ban on recreational feeding of deer and elk was instituted over a 4,000 mi^2 area to help reduce the risk of disease transmission among deer and between deer and livestock (Figure 5). Enforcement officers are planning to conduct an aerial survey of the bovine TB management zone in February 2007 to ensure compliance with the feeding ban.

Further, the Minnesota Legislature passed a $54,000 funding initiative that increased the amount of deer-proof fencing materials that can be provided by the DNR to cattle producers within 5 miles of a bovine TB-infected herd. The intent of this legislation is to protect stored feed from deer depredation and reduce the risk of deer to deer or deer to cattle transmission of the disease. The program allows for up to $5,000 of deer-proof fencing materials per qualified livestock producer.

The presence of bovine TB in cattle and wild deer in Minnesota has led the USDA to demote the state's bovine TB status from "free" to "modified accredited"; resulting in mandatory testing of cattle and restrictions on cattle movements. As part of the requirements to regain TB-Free accreditation, USDA has required BAH to test 1,500 cattle herds statewide for the disease. The DNR is committed to assisting the BAH in regaining Minnesota's TB-Free status. To accomplish this, the DNR will continue to conduct surveillance in 2007 and beyond.

ACKNOWLEDGEMENTS

We would like to thank the students and faculty from the University of Minnesota, College of Veterinary Medicine, that assisted in our sampling efforts. Also, thanks to DNR staff that worked at registration stations, as well as Steve Benson and Julie Adams for making our surveillance maps.

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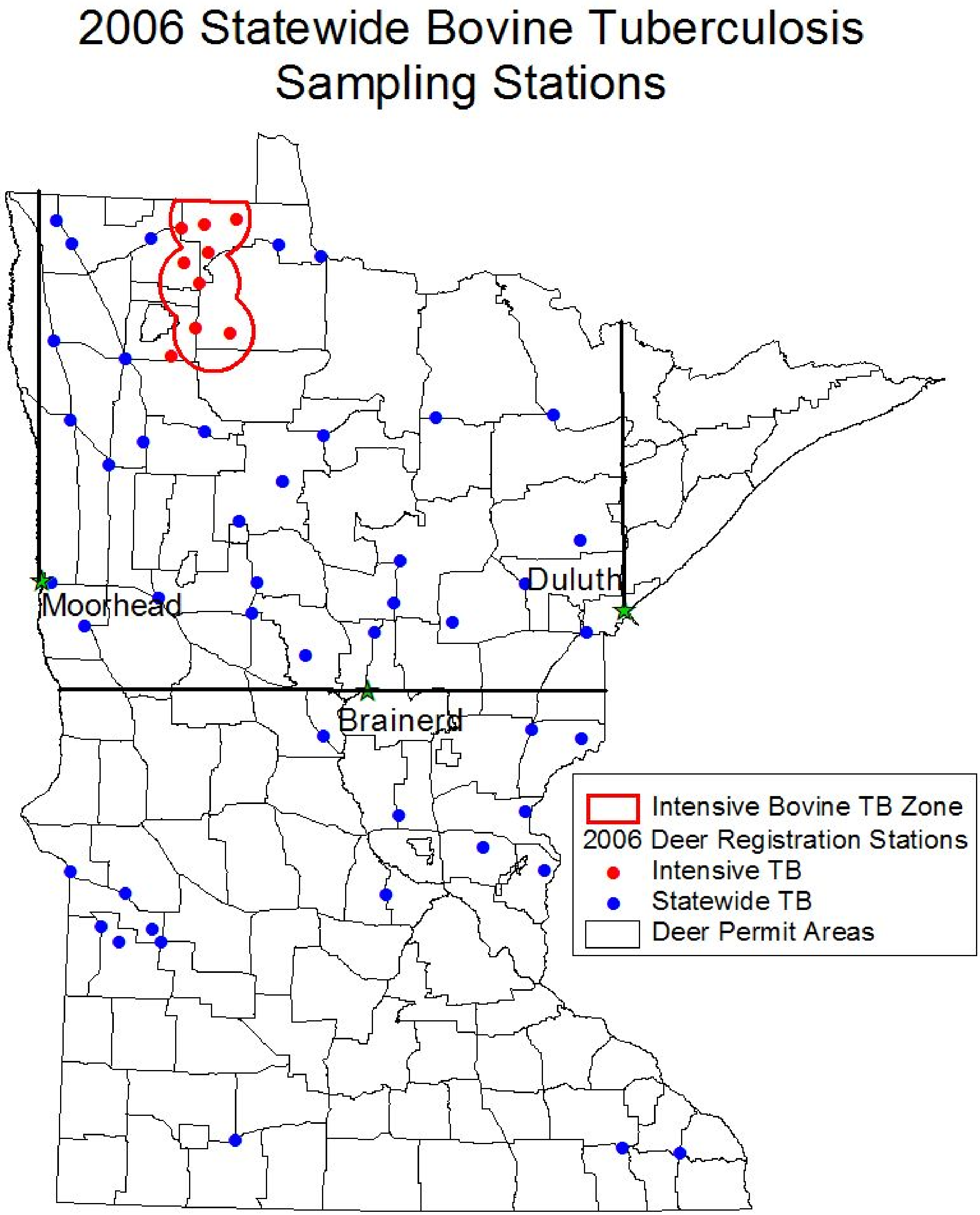


Figure 1. Locations of registration stations to conduct hunter-harvested surveillance of wild deer for bovine tuberculosis in Minnesota, fall 2006.

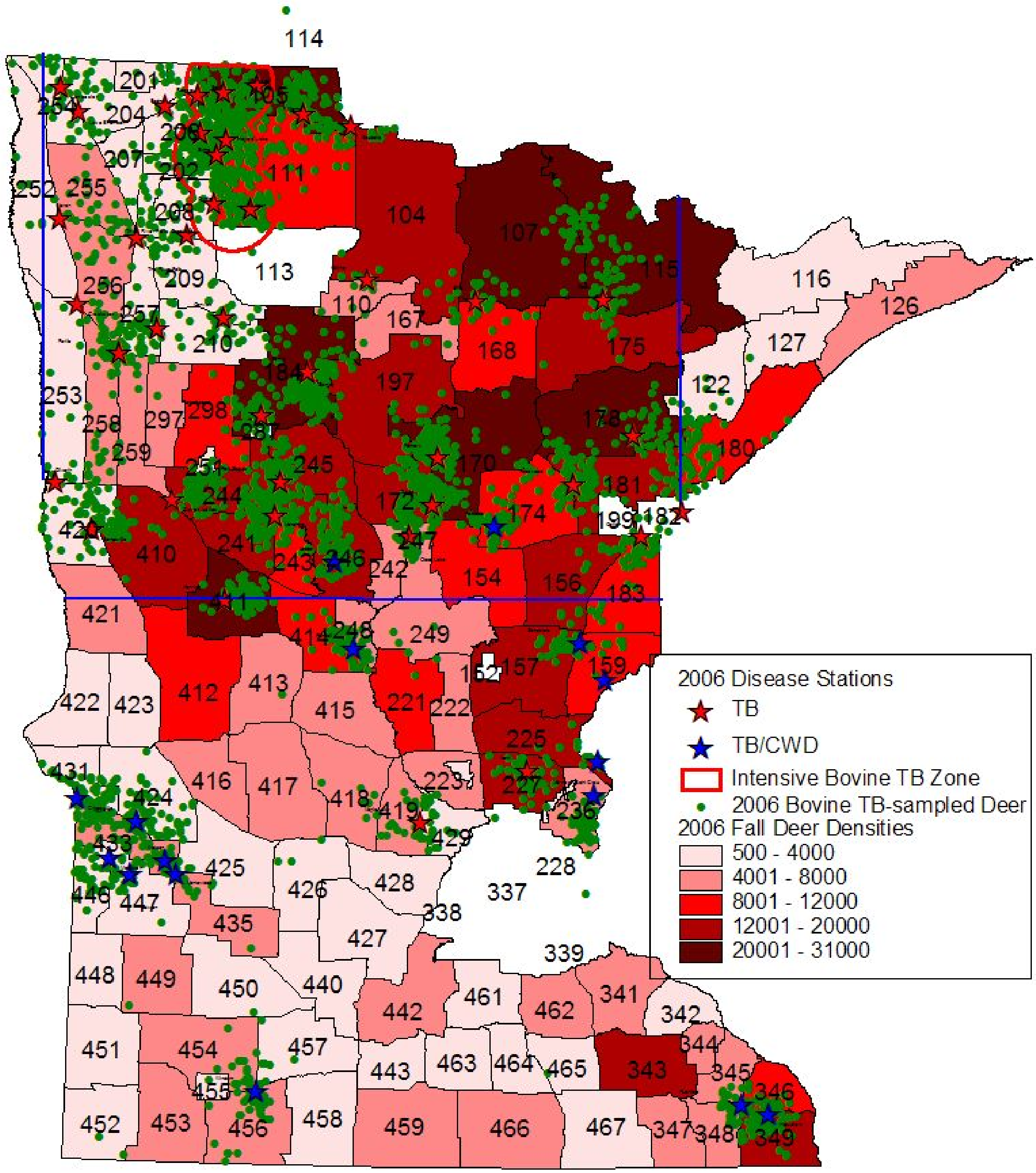


Figure 2. Locations of deer sampled for bovine tuberculosis in Minnesota, fall 2006. Sampling intensity was based on deer densities and distribution within the state.

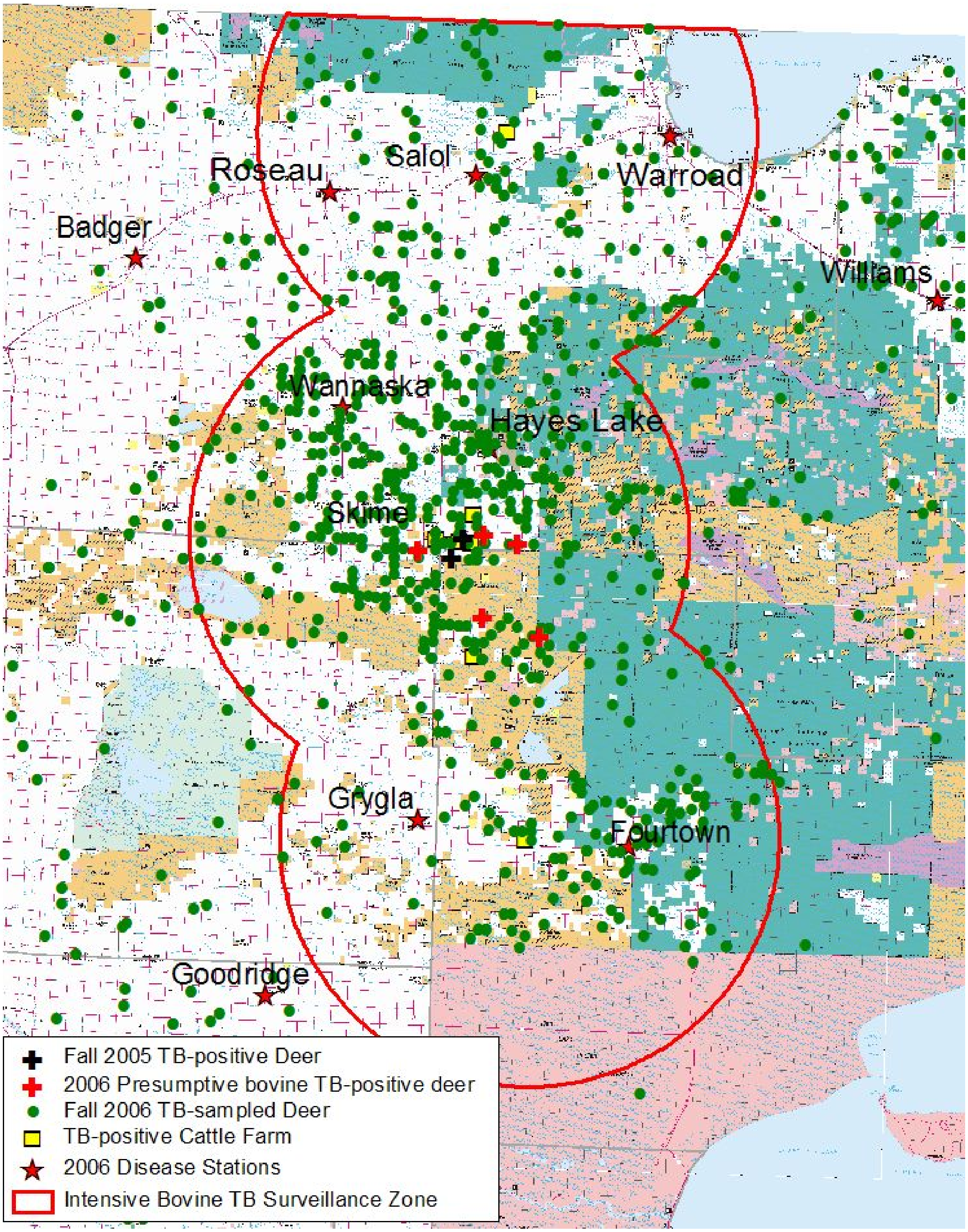


Figure 3. Locations of deer sampled for bovine tuberculosis (TB) in the surveillance zone in northwestern Minnesota, fall 2006. Deer found infected with the disease in 2005 are noted with black crosses, and red crosses correspond to infected deer from 2006.

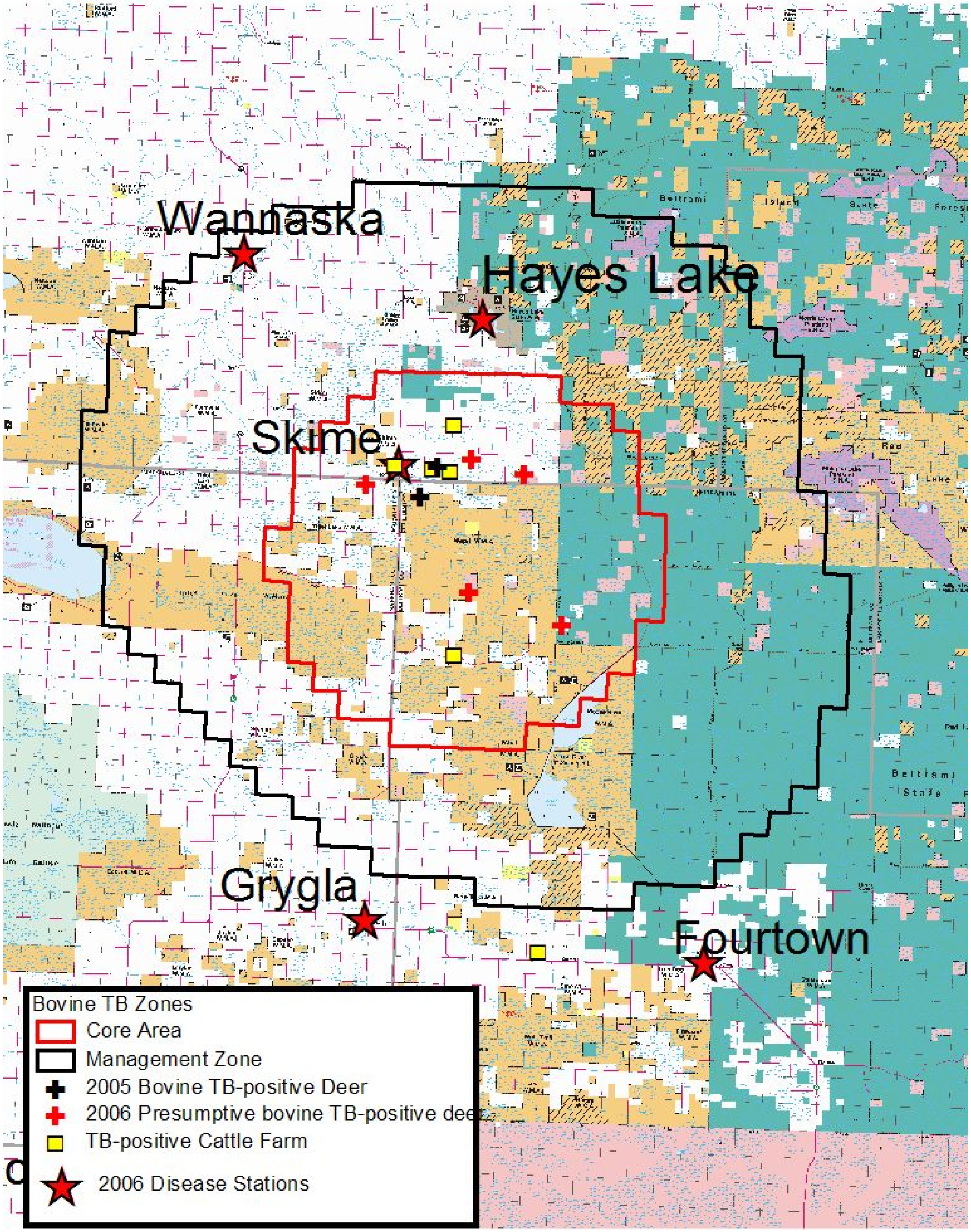
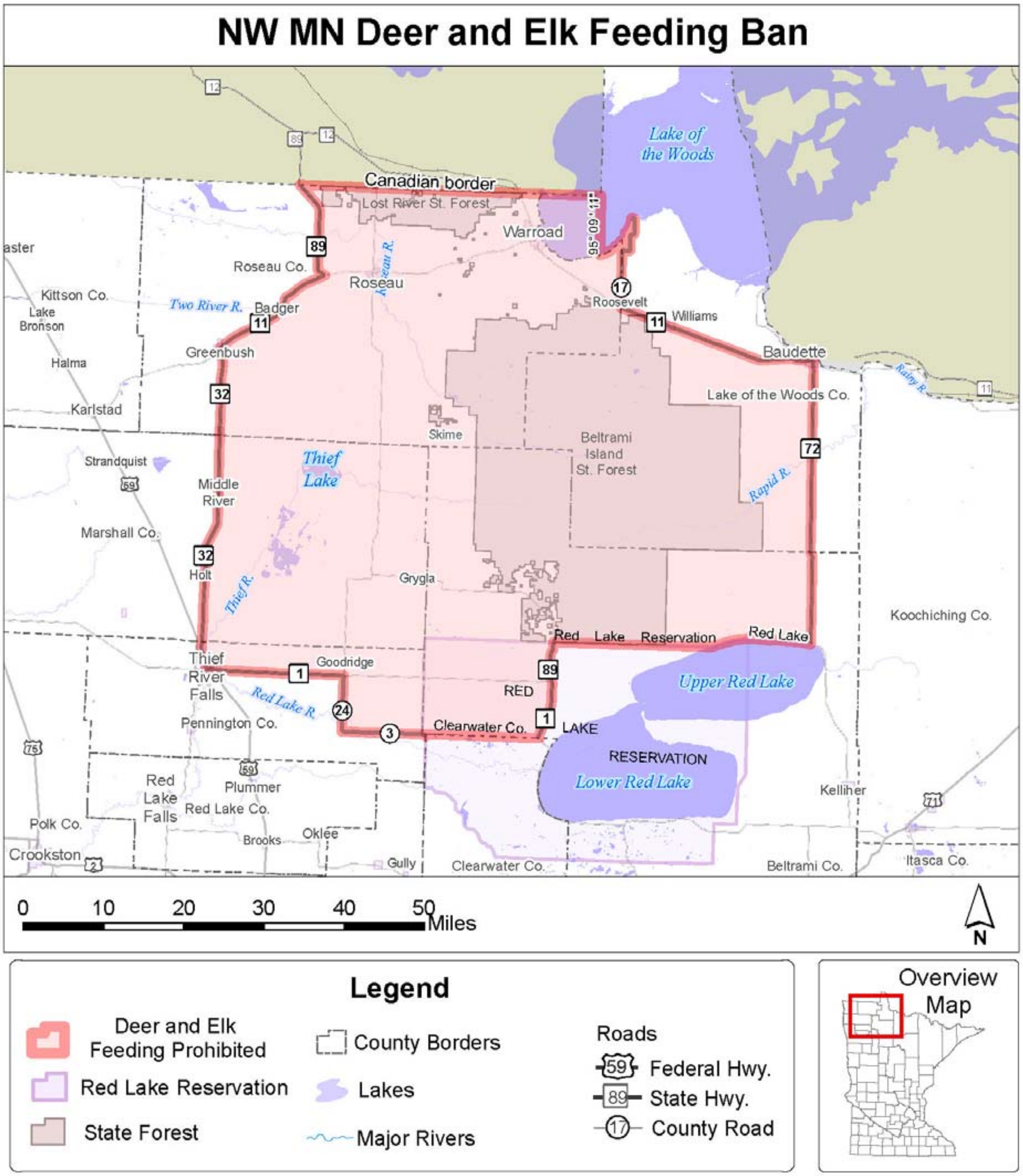


Figure 4. Newly created Bovine Tuberculosis Management Zone (delineated in black), which includes a 10-mile buffer around all deer found positive for the disease and a 2-mile buffered core area (delineated in red). This will allow for increased focus of further disease management efforts for wild deer in northwestern Minnesota.



Deer and Elk Feeding Ban Boundary Description

Beginning at State Trunk Highway (STH) 89 and the Canadian border; thence along the Canadian border to longitudinal point 95 degrees, 9 minutes, 11 seconds; thence due south to the shore of Lake of the Woods; thence along the shore of Lake of the Woods to County Road (CR) 17; thence along CR 17 to STH 11; thence along STH 11 to STH 72; thence along STH 72 to the north shore of Upper Red Lake; thence along the north shore of Upper Red Lake to the boundary of the Red Lake Indian Reservation; thence along the boundary of the Red Lake Indian Reservation to STH 89; thence along STH 89 to STH 1; thence south along STH 1 to the Clearwater County line; thence along the Clearwater County line to the Pennington County line; thence due west to the intersection of CR 27 and CR 3; thence along CR 3 to CR 24; thence along CR 24 to STH 1; thence along STH 1 to STH 32; thence along STH 32 to STH 11; thence along STH 11to STH 89; thence along STH 89 to the point of beginning.

Figure 5. 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.

MINNESOTA DEPARTMENT OF NATURAL RESOURCES CHRONIC WASTING DISEASE SURVEILLANCE PROGRAM 2006

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

SUMMARY OF FINDINGS

As a continuation of Minnesota Department of Natural Resources (DNR) surveillance program for Chronic Wasting Disease (CWD), 1,260 free-ranging white-tailed deer (*Odocoileus virginianus*), including 83 exhibiting clinical signs of illness, were screened for the disease in Minnesota. None of these deer were found positive for CWD.

INTRODUCTION

In February 2006, a captive white-tailed deer was diagnosed with CWD in southwestern Minnesota. Consequently, DNR staff flew the immediate area to assess deer population levels, and formulated plans for 2006 surveillance of wild deer in the area to ensure the disease did not spill into the local wild deer herd. The DNR had recently completed statewide surveillance of hunter-harvested deer from 2002–2004, testing approximately 28,000 animals. None were infected with CWD.

CWD 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 believed to be an infectious protein, called a prion. Precise mechanisms and rates of CWD transmission remain unclear, although animal-to-animal contact and environmental contamination are likely to promote the spread of the disease. Incubation time of the disease, from infection to clinical signs, can range from a few months to nearly 3 years. 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.

METHODS

At the registration 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 Laboratory in 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 entered into a gun raffle.

The DNR continued to sample deer exhibiting clinical symptoms consistent with CWD (targeted surveillance). Information had been disseminated to wildlife staff regarding what to look for regarding symptomatic deer. Staff were given the necessary equipment and training on lymph node removal and data recording. The number of samples expected through targeted surveillance is estimated to be less than 200 animals annually, as few reports of sick deer are received.

During fall 2006, the DNR also collected approximately 4,000 lymph node samples from hunter-harvested deer as part of a one-time, statewide surveillance program for Bovine Tuberculosis. The DNR had planned to screen approximately 1,500 of these samples for CWD

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as well. The registration stations that were selected to screen for both diseases include those along the Minnesota-Wisconsin border and a few central counties where CWD surveillance was conducted in response to CWD-positive captive animals in 2002.

RESULTS AND DISCUSSION

The DNR collected 367 samples from hunter-harvested deer in the vicinity of the positive captive cervid herd and 810 samples from registration stations along the Minnesota-Wisconsin border and central stations (Figure 1). Sampling occurred November 4-12, 2006. All deer tested negative for the disease. Additionally, 83 samples were submitted from suspect deer statewide through targeted surveillance; all deer tested negative for the disease.

Since the agency has now collected approximately 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. Targeted surveillance of suspect deer is expected to continue throughout the State.

ACKNOWLEDGEMENTS

We would like to thank the students and faculty from the University of Minnesota, College of Veterinary Medicine, that assisted in our sampling efforts. Also, thanks to DNR staff that worked at registration stations, as well as Steve Benson and Julie Adams for making our surveillance maps.

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Fall 2006 CWD Surveillance in Hunter-harvested Deer

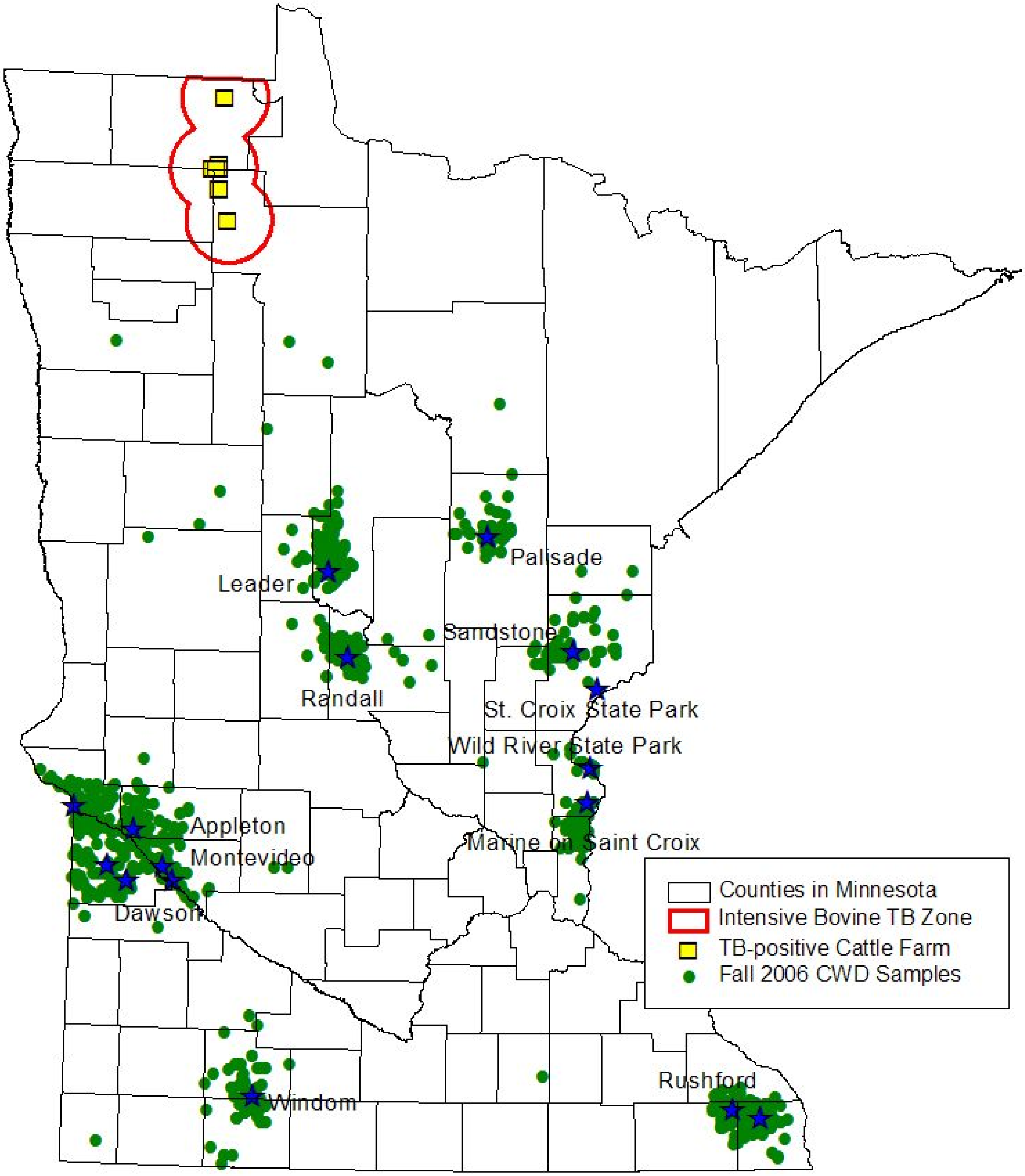

Figure 1. Sampling distribution for hunter-harvested deer tested for chronic wasting disease in Minnesota, fall 2006.

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, the Minnesota Department of Natural Resources (DNR) and the United States Department of Agriculture Wildlife Services (USDA-WS) conducted surveillance for the virus in waterfowl in the State. A combined total of 2,065 birds were sampled for HPAI in Minnesota during 2006. Testing did not result in any positive cases of HPAI, specifically the Asian strain of subtype H5N1; however 1 Northern pintail *(Anas acuta*) and 1 ring-necked duck (*Aythya collaris*) did test positive for a low pathogenic strain of avian influenza (AI) with the subtype H5, and 1 American green-winged teal (*Anas crecca*) tested positive for an N1 subtype. Approximately 164,000 wild birds were sampled throughout the United States in 2006, and no positive cases of HPAI were detected. It is likely that the DNR will continue surveillance for the virus in the state's waterfowl for the next several years, in cooperation with the Mississippi Flyway, Council of the US Fish and Wildlife Service, and the USDA.

INTRODUCTION

Recent worldwide attention on the spread of a highly pathogenic strain of avian influenza (HPAI), 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 birds 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 June 2006, the DNR entered into a $100,000 cooperative agreement with the USDA-WS to sample 1,000 wild birds (either live-caught or hunter-harvested) in Minnesota for HPAI-H5N1 during 2006. In addition to the 1,000 samples to be collected by DNR, 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. Avian influenza 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 200 humans in Eurasia and Africa, resulting in over 100 deaths.

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METHODS

DNR planned to sample 100 common goldeneye (*Bucephala clangula*) and 100 ring-necked ducks during the summer months, primarily in conjunction with planned banding activities, and 100 Northern pintails, 200 mallards (*Anas platyrhynchos*), 200 American green-winged teal, 100 lesser scaup (*Aythya affinis*), and 200 Canada geese (*Branta canadensis*) in the fall through hunter-harvested surveillance. USDA-WS planned to sample 100 mallards, 50 Canada geese, and 100 shorebirds during the summer months, and 100 Northern pintails, 100 lesser scaup, 100 common goldeneyes, 100 ring-necked ducks, 100 American blue-winged teal (*Anas discors*), 100 shorebirds, 100 sharp-shinned hawks (*Accipiter striatus*), and 100 gray-cheeked thrush *(Catharus minimus*) during the fall. If sampling goals per species could not be met, other targeted waterfowl species, such as American wigeon (*Anas americana*) or Northern shovelers (*Anas clypeata*), could be substituted to ensure that total numerical sampling goals were met. Sampling strategies were coordinated between DNR and USDA-WS to maximize access to targeted birds species through existing banding operations and fall hunter-harvested surveillance.

Cloacal 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 Laboratory in Ames, IA for strain-typing.

RESULTS AND DISCUSSION

DNR collected a total of 1,015 samples; 24% in the summer through banding programs and 74% in fall through hunter-harvested surveillance. USDA-WS collected a total of 1,050 samples; 33% in the summer months and 67% in the fall. Thus, a combined total of 2,065 birds were sampled for HPAI-H5N1 in Minnesota in 2006 (Figures 1 and 2, Table 1).

Testing did not result in any positive cases of HPAI-H5N1; however 1 Northern pintail and 1 ring-necked duck did test positive for a low pathogenic strain of avian influenza with the subtype H5, and 1 American green-winged teal tested positive for an N1 subtype (Table 1). The testing protocol was limited to the screening for H5, H7, and N1 subtypes only.

According to the latest numbers on the United States Geologic Survey's website (http://wildlifedisease.nbii.gov/ai/), approximately 164,000 birds have been sampled for HPAI-H5N1 in the U.S. in 2006. This includes over 22,000 samples taken in Alaska, which was considered the most at-risk state for the introduction of HPAI-H5N1 into North America. No positive cases of HPAI-H5N1 have been found anywhere in North American to date. However, the National Veterinary Services Laboratory did report 16 presumptive positive H5N1 cases, of which only 6 were confirmed as a low pathogenic H5N1 subtypes (commonly referred to as the "North American Strain"). These 6 cases included: American green-winged teal (Delaware), mallard (Illinois), mallard (Michigan), mallard (Pennsylvania), mute swan (*Cygnus olor*) (Michigan), and a mallard (Maryland).

Surveillance for HPAI-H5N1 will likely continue in Minnesota, and other parts of the U.S. for the next several years. The USDA has banked all samples taken in 2006 and is currently accepting proposals from state agencies and universities for further avian influenza research. Minnesota remains prepared to assist with future surveillance objective if needed. In addition, the DNR 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

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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 2006.

Species sampled	*n*	***Sex***[a]	***Age class***[b]	***Results of Avian Influenza testing***
Ducks				
American Green-winged Teal	248	39%F, 54%M, 7%U	43%J, 49%A, 8%U	H?N1 (Marsh Lake)
American Wigeon	99	38%F, 56%M, 6%U	63%J, 32%A, 5%U	Negative
American Blue-winged Teal	244	46%F, 32%M, 22%U	50%J, 32%A, 18%U	Negative
Common Goldeneye	86	42%F, 58%M	100%J	Negative
Gadwall	4	25%F, 50%M, 25%U	50%J, 25%A, 25%U	Negative
Lesser Scaup	30	47%F, 37%M, 16%U	37%J, 47%A, 16%U	Negative
Mallard	310	51%F, 46%M, 3%U	82%J, 15%A, 3%U	H4N6 and H?N2 (Thief Lake WMA)
Northern Pintail	111	59%F, 39%M, 2%U	71%J, 25%A, 4%U	H5N? (Thief Lake WMA)
Northern Shoveler	75	49%F, 47%M, 4%U	63%J, 35%A, 2%U	Negative
Redhead	19	26%F, 32%M, 42%U	16%J, 42%A, 42%U	Negative
Ring-necked Duck	330	43%F, 36%M, 21%U	48%J, 30%A, 22%U	H5N? (Upper Rice Lake)
Wood duck	1	Not determined	Not determined	Negative
Canada Geese	151	25%F, 19%M, 56%U	38%J, 56%A, 6%U	Negative
Shorebirds				
Bairds Sandpiper	4	Not determined	Not determined	Negative
Greater Yellowlegs	3	Not determined	Not determined	Negative
Least Sandpiper	98	Not determined	Not determined	Negative
Lesser Yellowlegs	8	Not determined	Not determined	Negative
Pectoral Sandpiper	31	Not determined	Not determined	Negative
Short-billed Dowitcher	3	Not determined	Not determined	Negative
Semipalmated Plover	2	Not determined	Not determined	Negative
Semipalmated Sandpiper	52	Not determined	Not determined	Negative
Spotted Sandpiper	2	Not determined	Not determined	Negative
Upland Sandpiper	2	Not determined	Not determined	Negative
Wilson's Snipe	1	Not determined	Not determined	Negative
Other				
Gray-cheeked Thrush	8	Not determined	Not determined	Negative
Swainson's Thrush	3	Not determined	Not determined	Negative
Sharp-shinned Hawk	140	47%F, 52%M, 1%U	91%J, 8%A, 1%U	Negative
Total	2065			

[a]F=female, M=male, U=unknown.
[b]J=juvenile, A=adult, U=unknown.

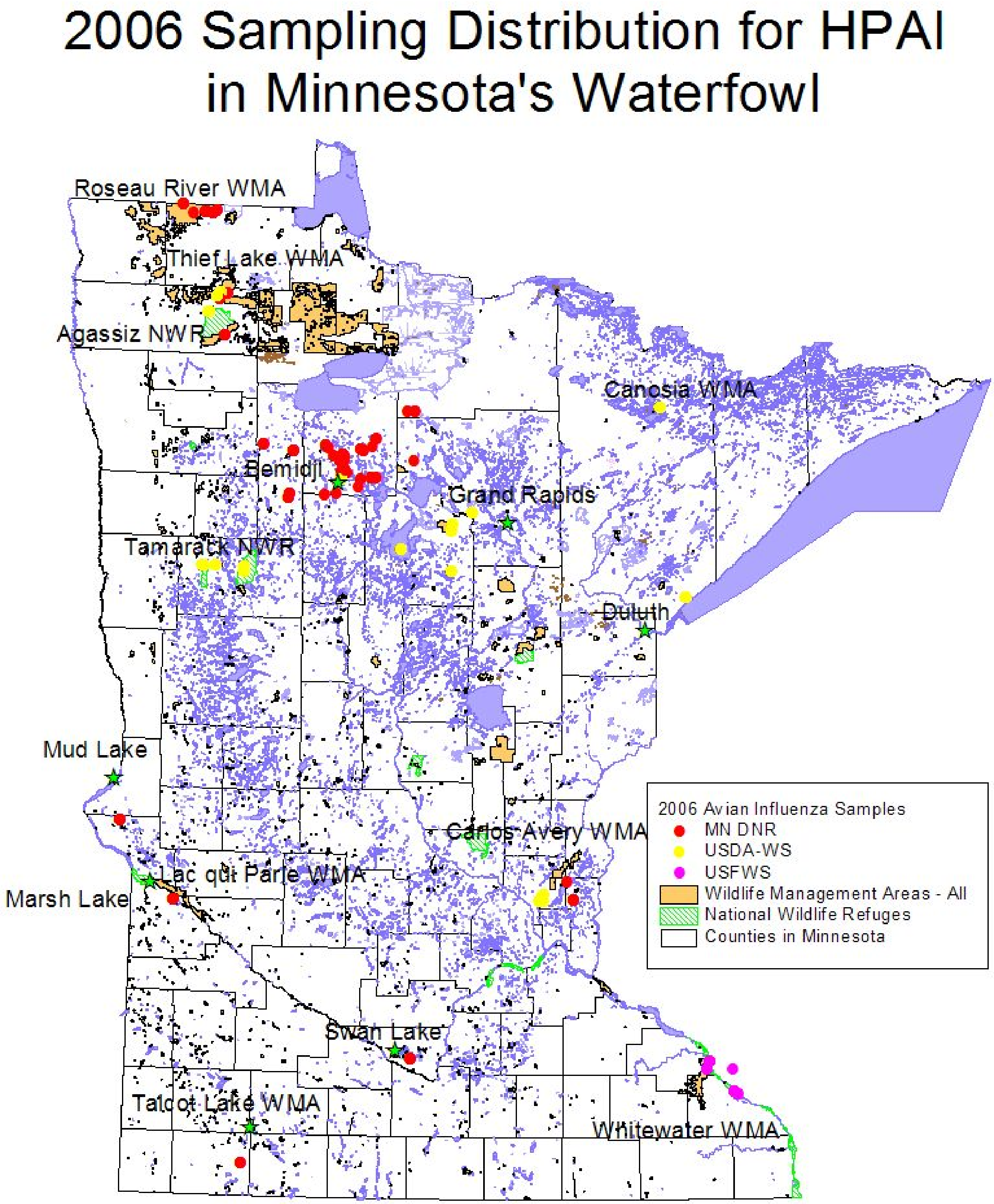

Figure 1. Sampling locations where waterfowl were tested for highly pathogenic avian influenza in Minnesota during 2006.

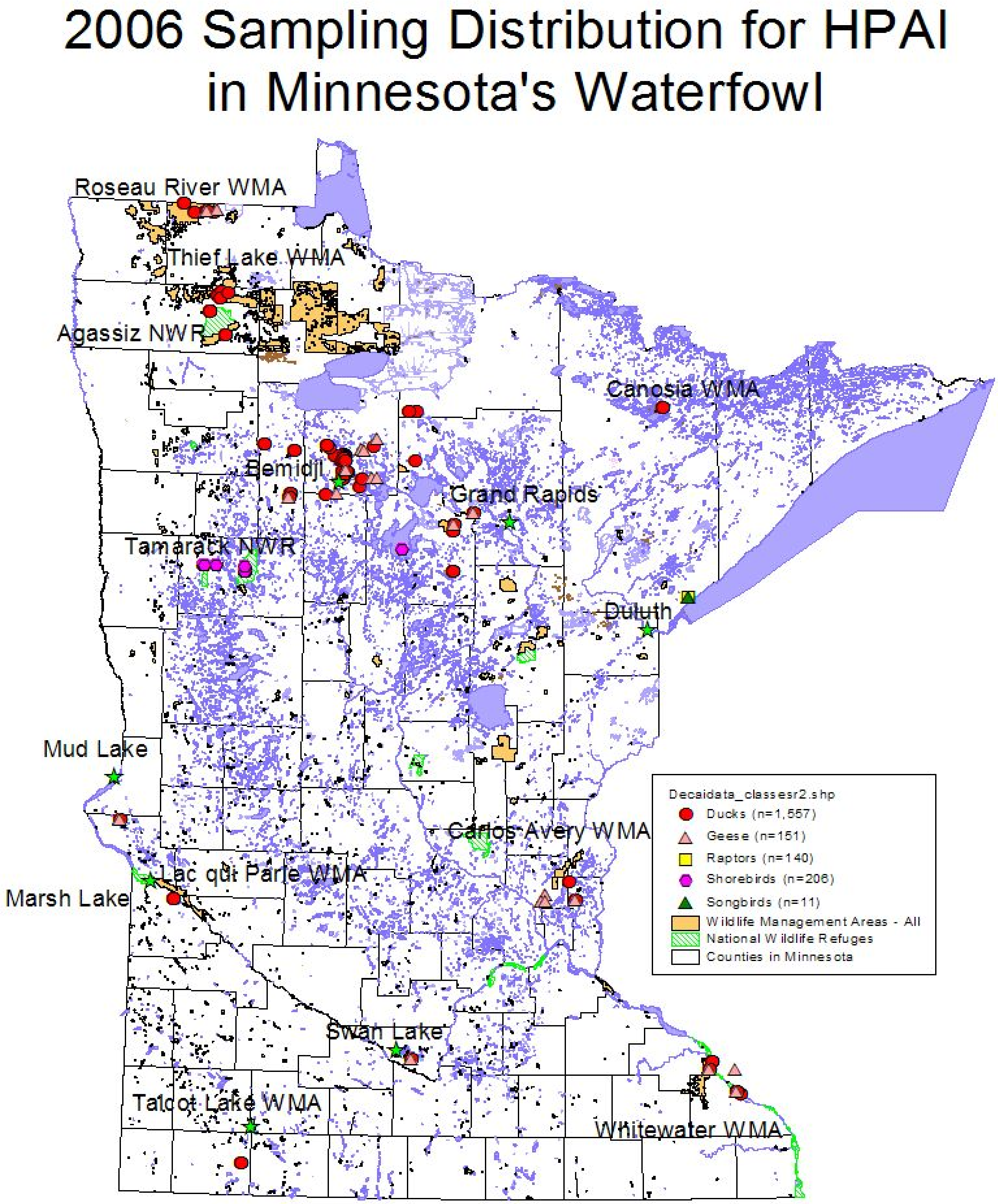

Figure 2. Distribution of waterfowl, including ducks, geese, raptors, shorebirds, and songbirds sampled for highly pathogenic avian influenza in Minnesota during 2006.

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