SUMMARIES OF WILDLIFE RESEARCH FINDINGS 2005

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

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ECOLOGICAL CONSIDERATIONS FOR LANDSCAPE-LEVEL MANAGEMENT OF BATS[1]

Joseph E. Duchamp[2], Edward B. Arnett[3], Michael A. Larson, and Robert K. Swihart[2]

Abstract: Bats exhibit a high degree of temporal and spatial mobility across a variety of habitats. This characteristic dictates using a landscape approach for their study. To effectively protect and conserve populations, it is important to acknowledge that bats interact with their environment over broad spatial scales composed of heterogeneous mixtures of habitats. Our goal in this chapter is to facilitate further consideration of landscape attributes in both research of and management for bat populations by reviewing basic concepts in landscape ecology and summarizing current literature that incorporates a landscape approach. Major sections of the chapter include fundamentals of landscape ecology, selecting the appropriate landscape elements for analysis of bat habitat, managing habitat for bats across broad spatial scales, and using habitat models (e.g., habitat suitability index, resource selection functions) to predict effects of land management on bats.

[1] Manuscript will be published as Chapter 9 in Lacki, M.J., J. P. Hayes, and A. Kurta, Editors. 2006. Conservation and management of bats in forests. Johns Hopkins University Press, Baltimore, Maryland
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ECOLOGY AND POPULATION DYNAMICS OF BLACK BEARS IN MINNESOTA

David L. Garshelis, Pamela L. Coy and Karen V. Noyce

SUMMARY OF FINDINGS

During April 2005 – March 2006, 37 radiocollared black bears *(Ursus americanus)* were monitored at 3 Minnesota study sites: Chippewa National Forest (CNF; central study site), Camp Ripley (southern) and Voyageurs National Park (northern). Prior to this year's monitoring, 827 individual bears were handled at these 3 sites, beginning in 1981 in the CNF. Mortality data were obtained through collars turned in by hunters or collars tracked to carcasses. Hunting remains the largest source of mortality of collared bears, even though hunters were asked not to shoot bears with radiocollars. Reproductive output varied among the 3 study sites in response to food conditions. All sites exhibited largely synchronous reproduction by adult females, with high cub production occurring in odd-numbered years. This matches data from the statewide harvest age structure. The harvest age structure also shows evidence of an increasing proportion of yearling bears, indicative of population-wide changes in either reproduction or mortality.

INTRODUCTION

A paucity of knowledge about black bear *(Ursus americanus)* 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 (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 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.

By comparing results from 3 study sites over a long term, we have gained insights into both spatial and temporal variation in bear life history parameters that are directly related to bear management. We tested and deployed a tetracycline-based mark–recapture program, and have since obtained 3 statewide population estimates over a span of 12 years (Garshelis and Visser 1997, Garshelis and Noyce 2006). However, confounding variables, related mainly to capture heterogeneity (e.g., Noyce et al. 2001) have necessitated further study for refinement of the technique. We developed a means of ascertaining reproductive histories from the spacing of cementum annulations in teeth (Coy and Garshelis 1992), which was used to investigate variation in reproductive output across the state (Coy 1999). We also developed a method for obtaining unbiased estimates of age of first reproduction and interval between litters (Garshelis et al. 1998, Garshelis et al. 2005). These data are needed for continued statewide population modeling. For many years, we have focused our efforts on measuring and monitoring physical condition of bears (Noyce and Garshelis 1994, Noyce et al. 2002) and their food supply (Noyce and Garshelis 1997). Results of this work have been instrumental in explaining variations in harvest numbers and sex-age structure (Garshelis 2006). All of these represent areas of continued research and monitoring.

OBJECTIVES

- Monitor temporal and spatial variation in cub production and survival;
- Monitor rates and sources of mortality; and
- Obtain additional, improved, measurements of body condition, effects of hibernation, and wound healing abilities.

METHODS

Radiocollars (with breakaway and/or expandable devices: Garshelis and McLaughlin 1998, Coy unpublished data) 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 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 (these data are not presented in this report). 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, weighed, and ear tagged. 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

From 1981 through completion of den visits in March 2005, a total of 652 individual bears were handled in and around CNF, 91 at Camp Ripley, and 84 at VNP. Nearly 500 of these have been radiocollared. As of April 2005, the start of the current year's work, we were monitoring 14 collared bears in the CNF, 9 at Camp Ripley, and 8 in VNP, as well as 6 released orphaned cubs. By April 2006, after deaths, failed radiocollars, and the addition of some new bears obtained through trapping, released orphaned cubs, and den visits, 42 bears collared bears were being monitored.

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, 5 or 6 (1 bear lost during the first week of the hunt may have been killed) of 22 (23-27%)

collared females from the CNF, 1 of 12 (8%) from Camp Ripley, and 2 of 7 (29%) from VNP were shot by hunters (bear hunting is not allowed on Camp Ripley or VNP, but bears are vulnerable to hunters when they leave these areas). This rate of hunting-caused mortality (20-22% overall) was equivalent to years when we used black-colored collars.

In addition to these hunter-related mortalities, 2 natural mortalities occurred in VNP. An 11-year-old female with cubs was found dead in June (parts of two cubs were also found), and a 2-year-old female was found dead in July. Body parts were too decomposed to discern a cause of death for either.

No collared bears were killed as nuisances, although in late summer we received several complaints regarding collared yearling bears that we had released in November 2004. These had been orphaned cubs, raised by a rehabilitation facility. One of these was later shot by a hunter.

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. This synchrony matches that exhibited in the age structure of the statewide bear harvest (Figure 1). This synchrony stemmed from a very poor 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.

Five study bears produced cubs in winter 2006. Four of these are on an even-year production schedule, whereas one that bore a single cub in 2005 and lost it, produced another litter this year.

Bears at Camp Ripley, where hard mast (especially oak) *(Quercus spp.)* is abundant, grow faster and thus have an earlier age of first reproduction than at the other 2 study sites, where oaks are more scarce. However, average litter size at Camp Ripley is smaller and have higher cub mortality higher than at CNF (Tables 2 and 3) because first litters by young females tend to be smaller and cub mortality than subsequent litters (Noyce and Garshelis 1994). VNP, having lower natural food availability than either Camp Ripley or CNF, had the oldest age of first reproduction, the smallest litters, and highest cub mortality. Cub production and survival also appeared to be most variable from year to year at VNP (Table 4).

We investigated age and year-specific variation in cub production within our long-term dataset in CNF. We measured cub production as 1) the proportion of collared females that produced a surviving litter of cubs (i.e., a litter in which at least 1 cub survived at least 1 year), and 2) the reproductive rate, defined as the number of cubs (both sexes) produced per female (as described by Garshelis et al. 2005). For year-specific analyses, we calculated productivity only for females at least 4 years old. We considered 4 years old the minimum age of sexual maturity in CNF, as only 2 of 83 (2%) collared bears in this area produced cubs at 3. Age-specific cub production increased until about 7 years old (Figure 2), at which point nearly all bears had produced their first cubs. From age 7 to 25 years, 48% of females produced surviving litters of cubs. If all bears produced cubs every other year, then 50%, on average, would have cubs in any given year. Of 115 observed intervals between successful litters, all but 7 were 2 years duration, yielding an average litter interval of 2.06 years (1/2.06 yields an expected 48.5% of females bearing cubs each year).

The reproductive rate includes both the proportion of females producing cubs and litter size. If litter size were constant by age and year, the proportion producing cubs and the reproductive rate would be redundant. Litter size, though, increased with age, averaging 2.0 for 3-year-old mothers, 2.3 for 4–6 year-olds, 2.7 for 7–9 year-olds, and 2.9 for 10–20 year-olds. We observed no cub production after age 25, but we observed only 1 collared bear that lived that long (a bear that is presently 32 years old and still being monitored).

Cub production among radio-collared females in CNF did not show an upward or downward trend during our 26 years of monitoring. However, statewide bear harvests have shown an increasing proportion of yearlings (Figure 1), either indicating increased reproduction, an altered age structure, or changing selectivity by hunters.

Cub mortality also has not shown any upward or downward trend over the course of our study (Tables 2–4). Mortality of male cubs has averaged about twice that of females in all areas (25% M vs 11% F in CNF; 38% M vs 14% F in Camp Ripley; 35% M vs 24% F in VNP). However, sex ratios at birth were skewed towards males in all areas (52–53%; Tables 1–3).

These results have been used as inputs in a statewide population model that is matched to our tetracycline-based population estimates (Garshelis and Noyce 2006).

ACKNOWLEDGMENTS

We thank the collaborators in this study: Brian Dirks and Julie DeJong at Camp Ripley, Steve Windels and Jen Fox at Voyageurs National Park, Paul Iaizzo at the University of Minnesota, and Tim Laske at Medtronic, Inc. We also thank the staff at Camp Ripley for trapping bears. Numerous volunteers assisted with den work.

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Table 1. Causes of mortality of radiocollared black bears ≥1 years 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	211	9	10
Likely shot by hunter[a]	8	1	0
Shot as nuisance	22	2	1
Vehicle collision	12	5	1
Other human-caused death	9	0	0
Natural mortality	7	3	3
Died from unknown causes	3	1	0
Total deaths	272	21	15

[a] Lost track of during the hunting season.

Table 3. Black bear cubs examined in dens of radiocollared mothers in Camp Ripley Military Reserve during March, 1992–2006.

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%	
Overall	20	47	2.4	53%	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 2. Black bear cubs examined in dens of radiocollared mothers in or near the Chippewa National Forest during March, 1982–2006.

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%	—
2006	2	6	3.0	83%	28%
Overall	170	443	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 Voyageurs National Park during March, 1999–2006.

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%	—
2006	1	2	2.0	50%	20%
Overall	21	45	2.1	53%	30%

[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.

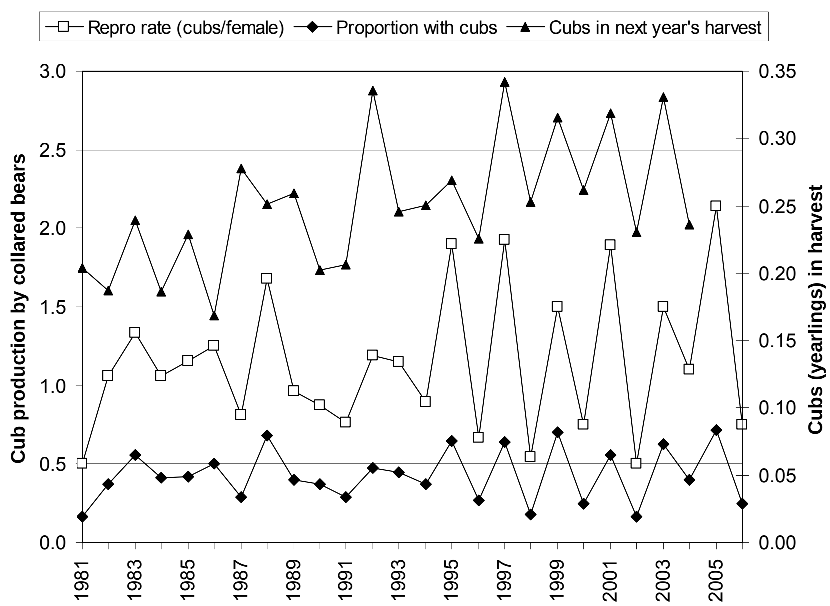


Figure 1. Comparison of reproductive data obtained from collared bears on the Chippewa National Forest (CNF) to the age structure of the statewide harvest. The strong reproductive synchrony observed among the collared bears in the CNF since 1995 (which was observed as well among collared bears at Camp Ripley and VNP) is reflective of births occurring statewide, as indicated by the varying proportion of yearlings in the harvest (yearlings in the harvest are slid back one year to match the year that they were born). Notably, the collared bear data from the CNF seems to match the statewide harvest data better in the last 10 years, than in earlier years. The proportion of yearlings in the harvest seems to be increasing (r^2 = 0.31, P = 0.005). The collared bear sample suggests a slight but as yet insignificant increase in the reproductive rate (M+F cubs per 4+ year-old female). Sample sizes vary from 5–25 females monitored per year (mean = 16).

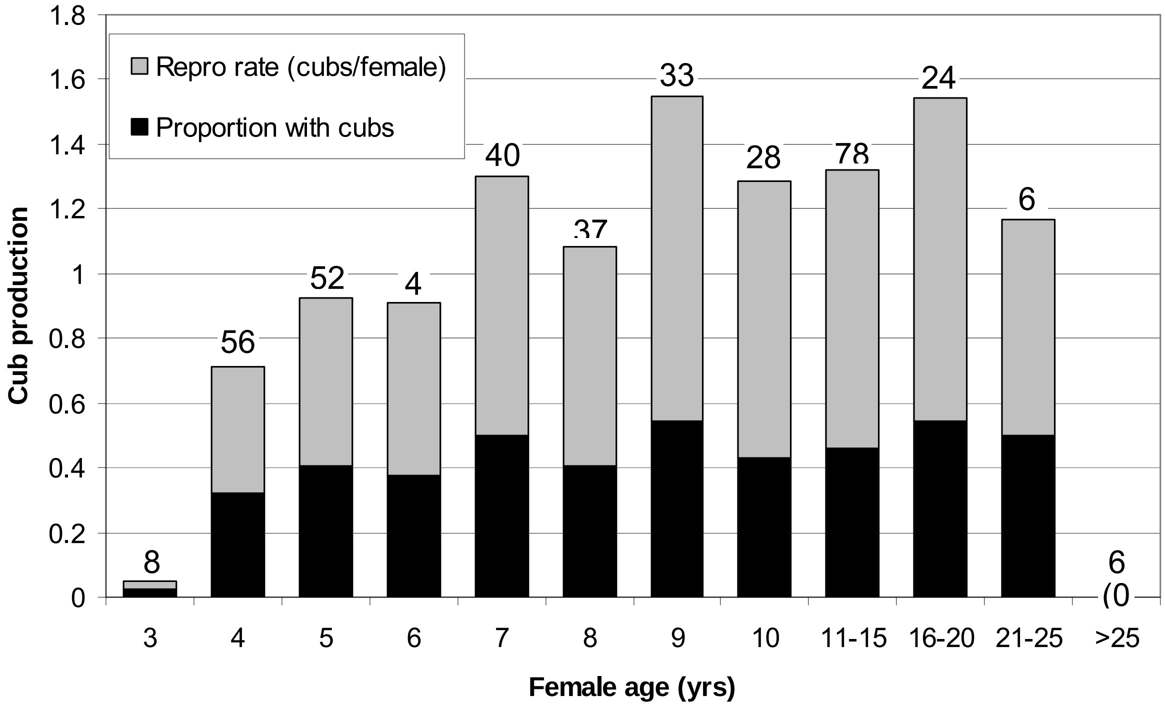


Figure 2. Age-specific cub production of bears in the Chippewa National Forest (central Minnesota) measured as the proportion of females with cubs during March den visits, 1982–2006, and cubs (M+F) per female. Sample sizes shown above bars represent bear-years (bears x years). However, only 2 individuals were monitored past age 20.

BOVINE TUBERCULOSIS IN WHITE-TAILED DEER IN NORTHWESTERN MINNESOTA

Michelle Carstensen Powell, Michael DonCarlos, and Lou Cornicelli

SUMMARY OF FINDINGS

Bovine tuberculosis (TB) was discovered in 5 cattle operations in northwestern Minnesota in 2005. The strain has been identified as one that is consistent with bovine TB found in cattle in the southwestern US and Mexico. To date, all of the infected cattle herds have been depopulated and the Board of Animal Health (BAH) is continuing to investigate the remaining quarantined herds in the area. In November 2005, the Minnesota DNR conducted bovine TB surveillance of hunter-harvested white-tailed deer (*Odocoileus virginianus*) within a 15-mile radius of the first 4 infected farms. One of the 474 deer tested was confirmed positive for bovine TB. The infected deer was harvested 1 mile southeast of Skime, which is in close proximity to one of the infected livestock operations. Further, 89 deer were harvested in spring 2006 through landowner shooting permits on the infected farms, yielding one additional positive deer. Because the infected deer were associated with infected livestock, share the same strain of bovine TB as the cattle, and no other infected deer were detected in the surveillance area, it is likely that the deer contracted the disease from cattle. The Minnesota Department of Natural Resources (DNR) will conduct hunter-harvested monitoring in fall 2006 to further monitor infection in the local deer population and 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. 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 (*Odocoileus virginianus*) 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, deer densities in the area of the infected Minnesota herds are much lower 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. Bovine TB is spread primarily through the exchange of respiratory secretions between infected and uninfected animals. This transmission usually happens when animals are in close contact with each other. Animals may also become infected with bovine TB by ingesting the bacteria from 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

An initial surveillance area was developed that encompassed a 15-mile radius around Skime and Salol, centering on the locations of the first 4 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 400 samples within the surveillance zone. Sampling was conducted during the first weekend of the firearms deer-hunting season (5–6 November 2005), 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 kill location. Maps were provided to assist the hunters in identifying the location (Township, Range, Section, and Quarter-section) of the kill. Cooperating hunters were given a Cooperator's Patch and entered into a 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. Collected samples were transported to Carlos Avery for processing and sorting, then submitted to the Veterinary Diagnostic Laboratory (VDL) at 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.

To assess farm-level prevalence of bovine TB, shooting permits for deer were issued in January 2006 to landowners of TB-infected herds or their fence-line contacts. Harvested deer were sampled in the same methods as previously described.

RESULTS AND DISCUSSION

In fall 2005, we collected 474 samples from hunter-harvested deer in the surveillance area (Figure 1). This includes 5 whole carcasses that were confiscated from hunters due to presence of suspicious lesions in the chest cavity of lymph nodes. Only one positively infected deer with bovine TB was diagnosed. The infected deer was located approximately 1 mile southeast of a bovine TB-infected cattle herd. The strain of bovine TB from this 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 an infected cattle herd, the strain type, and the fact that only 1 sampled deer (or 0.02%) was infected with the disease, supports our theory that this disease spilled-over from cattle to wild deer in this area of the state.

From January–April 2006, an additional 89 deer were harvested under shooting permits that were issued to landowners of bovine-TB infected cattle herds or their fence-line contacts. Given our theory of this disease originating in wild deer as a spill-over from infected cattle, it was highly likely that additional infected deer would be found on these farms given their increased risk of exposure to *M. bovis*. One carcass was confiscated from the landowner due to the presence of lesions that appeared consistent with bovine TB in the lungs and chest cavity of the deer, and was subsequently confirmed as positive for the disease.

The presence of bovine TB in cattle and wild deer in Minnesota has led the United States Department of Agriculture (USDA) to demote the state's bovine TB status from "free" to "modified accredited". This has resulted in mandatory testing of cattle and restrictions on cattle movements. The DNR is committed to assisting the BAH in regaining MN's TB-Free status. To accomplish this, the DNR will continue to conduct surveillance in 2006 (Figure 2) and beyond, and will implement a localized ban on recreational feeding. Additionally, DNR will provide fencing materials to affected livestock producers to protect stored forage from deer.

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's Jeff Lawrence, Dave Kuehn, and Carolin Humpal for volunteering their time to work at registration stations, as well as Steve Benson and Julie Adams for making our surveillance maps. Finally, the sampling of deer from shooting permits could not have been accomplished without the combined efforts of Paul Telander, John Williams, and the area managers and their staff.

Figure 1. Locations of hunter-harvested deer sampled for bovine tuberculosis in northwestern MN in fall 2005.

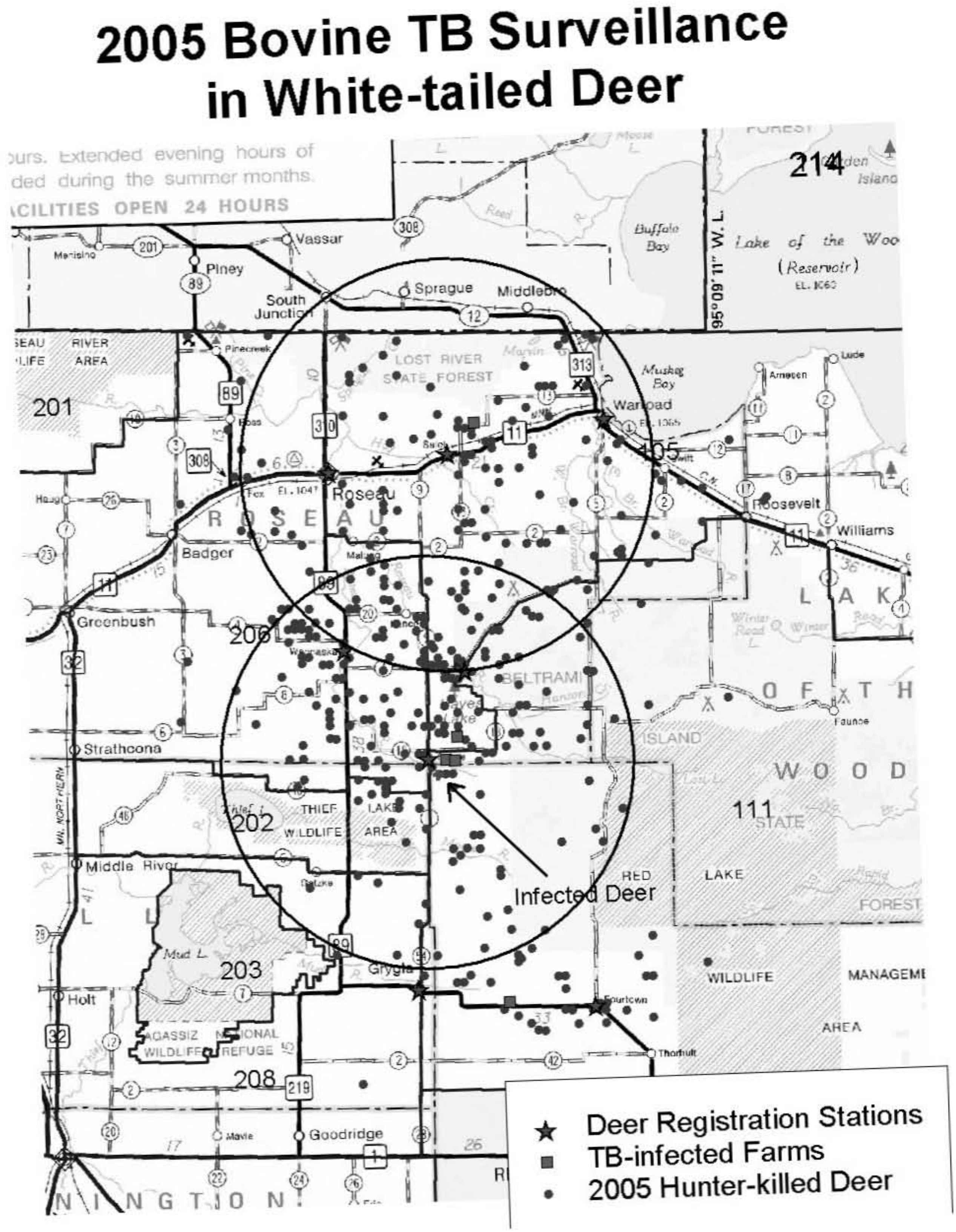
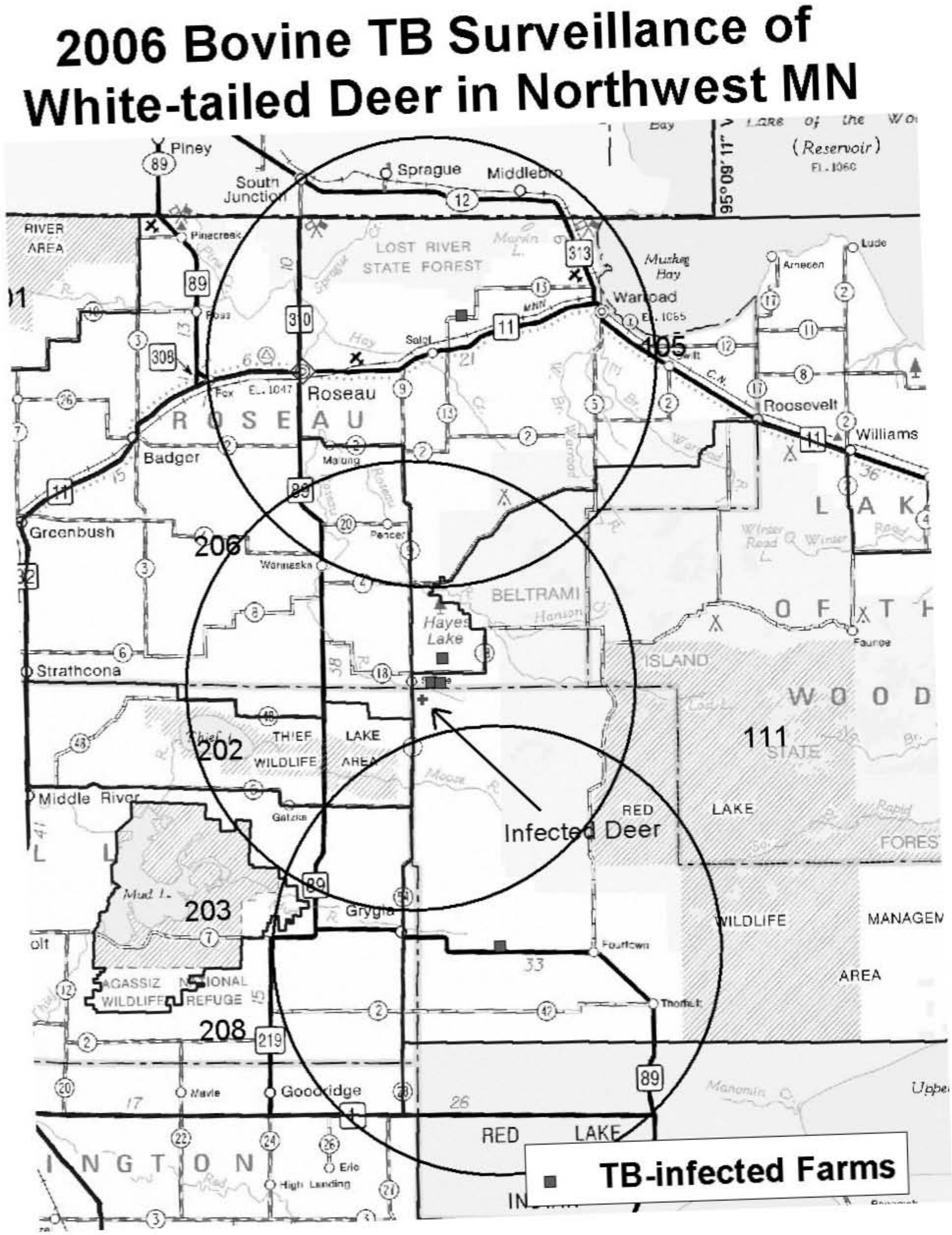


Figure 2. Surveillance zone planned for hunter-harvested surveillance for bovine tuberculosis in fall 2006.



AGE-SPECIFIC FERTILITY AND FECUNDITY IN NORTHERN FREE-RANGING WHITE-TAILED DEER: EVIDENCE FOR REPRODUCTIVE SENESCENCE?[1]

Glenn D. DelGiudice, Mark S. Lenarz, and Michelle Carstensen Powell

Abstract: White-tailed deer population performance is driven largely by survival and reproduction, and informed use of harvest management to approach regional population goals minimally requires balancing mortality (natural and human-related) with reproduction. Survival over the life cycle of deer is strongly related to age from birth through senescence. Given that age distributions of populations vary, this and similar reproductive information, would enhance our understanding of population performance and dynamics relative to intrinsic factors and regulatory mechanisms and their interaction with extrinsic factors. Our long-term (1991˜2002) objectives were to examine (1) serum progesterone as an indicator of pregnancy in free-ranging white-tailed deer (0.5˜15.5 yr old), (2) age-specific fertility and fecundity, and (3) the potential effect of reproductive senescence on population change. From 41 confirmed pregnant, adult (≥1.0 yr old) radiocollared does, with a mean age of 5.6 years old (95% CL = 4.4, 6.8), mean serum progesterone concentration at winter capture was 4.0 ng/ml (95% confidence limits [CL] = 3.6, 4.4). There were no relations between serum progesterone concentrations and julian date, age, or body mass at capture. Of these does, a minimum of 6 of 10 (60%) dams ≥ 10.0 years old gave birth to twins. We captured, aged, and blood-sampled a total of 284 females ranging in age from 0.5˜15.5 years ($\bar{x}$ = 4.9, 95% CL = 4.4, 5.4 yr). Based on a progesterone threshold indicative of pregnancy (1.6 ng/ml), mean progesterone of non-pregnant females ($\bar{x}$ = 0.4, 95% CL = 0.3, 0.5 ng/ml, *n* = 65) was less ($P \leq 0.05$) than in pregnant females ($\bar{x}$ = 3.8, 95% CL = 3.6, 4.0 ng/ml, *n* = 219). Only 1 of 55 (1.82%) fawns was pregnant, whereas, pregnancy was 87.5˜100.0% in adult does. Among adults, the lowest pregnancy rates occurred in yearlings, not in the oldest does. Further, estimated mean fecundity ranged from 1.31 fetuses:doe in yearlings to 2.20 fetuses:doe in 10.5-year olds. Mean fecundity in does 2.6˜15.6 years old was 1.8 fetuses:doe (95% CL = 1.7, 1.9 fetuses:doe). Again, serum progesterone was not related to julian date, age or body mass at capture. However, there was a significant difference in body mass between pregnant ($\bar{x}$ = 63.0, 95% CL = 61.9, 64.2 kg) and non-pregnant (($\bar{x}$ = 54.6, 95% CL = 49.1, 60.1 kg) adults and between pregnant ($\bar{x}$ = 55.1, 95% CL = 52.7. 57.4 kg) and non-pregnant ($\bar{x}$ = 48.4, 95% CL = 45.1, 51.7 kg) yearlings. Unlike for a number of other ungulate species, we observed no evidence of senescence relative to fertility and fecundity in adult female white-tailed does up to 15.5 years old. Because older does comprise a relatively small proportion of the population, our Leslie matrix modeling indicated that high pregnancy and fecundity rates of these females has little impact on population change (λ).

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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-March 1991-2005, we had 1,208 white-tailed deer (*Odocoileus virginianus*) captures, including recaptures. This long-term study's focus has been females, consequently males were ear-tagged and released. As of 31 March 2005, a total of 452 female deer, including 43 female newborns (captured during springs 1997, 1999-2002 as part of a companion study), had been recruited into this parent study. Highest fawn:doe ratios of the winter trapping periods occurred during 2001 (105 fawns:100 does) and 2005 (111 fawns:100 does). These winters were moderately severe to severe, but both followed 3 consecutive historically mild winters. The fawn:doe ratio has been as low as 32:100 (winter 1996–1997), attributable primarily to the historically severe winter 1995–1996. After the first year of the study, mean age of females remained stable and ranged from 5.1 (± 0.4 [SE], n = 94) in 2001 to 7.1 (± 0.6, n = 62) years old in 1993. During 2005, mean age was 5.7 (± 0.4) years old, compared to 6.1 (± 0.1) years old during the remainder of the study overall. The pregnancy rate of captured adult (≥1.0 years old) females has remained consistently high, 87.1% in yearlings to 100% in most other age classes up to 15.5 years old. Also, high age-specific fecundity persisted in even the oldest does captured (ranging between 1.6 and 2.0 fetuses:doe in females 7.5–15.5 years old. Significantly lower ($P \leq 0.05$) mean body mass at capture for non-pregnant (54.6, 95% CL = 49.1, 60.1 kg, n = 10) compared to pregnant does (63.0, 95% CL = 61.9, 64.2 kg, n = 171), which is indicative of an effect of inadequate nutritional condition during the fall rut. The wide-ranging severity of winter weather conditions (winter severity index [WSI] of 35 in winter 2005–2006 to 185 in winter 1995–1996) during the past 16 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. Mean winter mortality of adult females was 9.0% (± 1.91%), ranged from 1.9 to 29.3%, and was significantly related to WSI (r^2 = 0.52, P = 0.002). Mean non-winter (June–October) doe mortality was 4.7% (± 0.88%) and ranged from 0 to 11.1%. Mean annual mortality of females (including fawns) was 25.3% (± 2.49%), ranging from 9.1 to 47.6% through 2005. Wolf predation (24.4%), hunter harvest (23.4%), and "censored" (35.7%, i.e., lost to monitoring or still alive) accounted for the fates of most of the collared females through 2005.

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 (*Odocoileus virginianus*) at the population level. Historically, conifer stands have declined markedly relative to numbers of deer in Minnesota and elsewhere in the Great Lakes region. The level of logging of all tree species collectively, and conifer stands specifically, has recently reached the estimated allowable harvest. Most land management agencies and commercial landowners typically restrict harvests of conifers compared to hardwoods, because of evidence at least at the individual animal level, indicating the seasonal value of this vegetation type to various wildlife, including deer. However, agencies anticipate greater 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.

OBJECTIVES

The null hypothesis in this study is 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 are 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 sequential 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 survival.

METHODS

Study Design

This study employs 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 (Willow and Dirty Nose Lakes) and 2 treatment sites (Inguadona and Shingle Mill Lakes), a 5-year pre-treatment (pre-impact) phase, a conifer harvest serving as the experimental treatment or impact (4-year phase), and a 6-year post-treatment phase. The 4 study sites located in the Grand Rapids-Remer-Longville area of north central Minnesota are 10.4–22.0 km^2 (4.0–8.5 mi^2) in area. The study began with the Willow and Inguadona Lakes sites during winter 1990–91. The Shingle Mill and Dirty Nose Lakes sites were included beginning in winter 1992–93.

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). We just completed (31 December 2005) our 15th year of data collection and the 6th year of the post-treatment phase. This report is not a comprehensive summary of the study, rather I discuss the progress of numerous aspects, and I update various summary descriptive statistics.

Deer Capture

We captured white-tailed deer primarily with collapsible Clover traps (Clover 1956) during January–March 1991–2005 along the eastern and southern boundaries of the Chippewa National Forest, Minnesota (46°49'–47°11'N and 93°35'–94°20'W). We augmented our capture efforts during some winters with rocket-netting (Hawkins et al. 1968) and net-gunning from helicopters (Wildlife Capture Services, Marysvale, Utah). Generally, handling of each deer included chemical immobilization (intramuscular injection of a100/300 mg xylazine HCl/ketamine HCl combination), weighing, blood and urine-sampling (for assessment of nutritional, stress, and reproductive status [Warren et al. 1981, 1982; Wood et al. 1986; DelGiudice et al. 1987*a*,*b*, 1990*a*,*b*, 1994]), extraction of a 4th incisor for age-determination (Gilbert 1966), various morphological measurements, and administration of a broad-spectrum antibiotic. Does were checked for pregnancy by dop-tone or visual ultrasound and serum progesterone concentrations (pregnancy threshold of 1.6 ng/ml; Wood et al. 1986; DelGiudice, unpublished data). Female fawns and does were fitted with VHF radiocollars (Telonics, Inc., Mesa, Arizona) for monitoring their movements and survival, and 35 does (through January 2006) also were fitted with global positioning system (GPS) radiocollars (Advanced Telemetry Systems, Inc., Isanti, Minnesota). Upon completion of handling, all deer immobilizations were reversed with an intravenous injection of 15 mg yohimbine HCl. Additional details of deer capture and handling are provided elsewhere (DelGiudice et al. 2001, 2005, 2006; Carstensen Powell 2004).

We live-captured wolves (*Canis lupus*) with Newhouse number 14 steel leghold traps during May–September 1993–2005 to maintain radio contact for monitoring the movements of packs that ranged over the 4 deer study sites. Captured wolves were lightly anesthetized (xylazine/ketamine), weighed, blood-sampled, ear-tagged, radiocollared, injected with a broad-spectrum antibiotic, and released.

RESULTS AND DISCUSSION

Capture, Handling, Ages, and Reproductive Status of Study Deer

During this study, we had 1,208 deer captures, including recaptures. Because the study focuses on females, males were ear-tagged and released. As of 31 March 2005, a total of 452 female deer, including 43 spring-captured female newborns, had been recruited into the study. Highest fawn:doe ratios of the winter trapping periods occurred during 2001 (105 fawns:100 does) and 2005 (111 fawns:100 does). These winters were moderately severe to severe, but both followed 3 consecutive historically mild winters. The fawn:doe ratio has been as low as 32:100 (winter 1996–1997), attributable primarily to historically severe winter 1995–1996, and the consequence of record losses of newborns during the spring and summer of 1996.

As part of a newborn survival companion study, 47 male neonates were also spring-captured and radiocollared to monitor their survival and causes of mortality through early fall when collars dropped off. Additional information concerning the newborn deer portion of the study may be observed elsewhere (Carstensen Powell 2004, Carstensen Powell and DelGiudice 2005).

Measured at the end of each calendar year, or at death (or at last contact for "lost signals") within a specific year, mean age of collared female deer remained similar among the 4 study sites during the 5-year pre-treatment (1991–1995), 4-year treatment (1996–1999), and 6-year post-treatment phases (2000–2005). Consequently, observed differences in deer survival among sites within each of the study phases will not be confounded by differences in age among sites (DelGiudice and Riggs 1996). After 1991, mean age of deer on all 4 sites (pooled) also remained stable, and has ranged from 5.1 (± 0.4 [SE], n = 94 in 2001 to 7.1 (± 0.6, n = 62) years old in 1993 (Figure 1). During 2005, mean age was 5.7 (± 0.4) years old compared to 6.1 (± 0.1) years old during the remainder of the study. During the 15-year study, excluding newborns, females 0.5–9.5 years old at capture accounted for 85.7% of the study cohort, whereas senescent (relative to survival) does (10.5–15.5 yr old) accounted for the remaining 14.3% (Figure 2).

The elevated serum progesterone concentrations of pregnant adult females were stable throughout gestation and were unaffected by age and body mass at capture, which supports use of progesterone as a simple indicator of pregnancy (DelGiudice et al. 2006*b*). The pregnancy rate of all-age captured does (≥1.5 years old) has remained consistently high throughout the study, ranging from 87.1 to 100%, and exhibiting no indication of reproductive senescence relative to fertility or fecundity (Figure 3). Fecundity was lowest in yearlings (at winter capture) at 1.31 fetuses:doe, but remained high ($\bar{x}$ = 1.81 ± 0.06 fetuses:doe, *n* = 52) in 10.5–15.5-year olds (Fig. 3). There was a difference ($P \leq 0.05$) in mean body mass at capture for pregnant ($\bar{x}$ = 63.0, 95% CL = 61.9, 64.2 kg, *n* = 171) versus non-pregnant ($\bar{x}$ = 54.6, 95% CL = 49.1, 60.1 kg, *n* =10) adult females, as well as between pregnant ($\bar{x}$ = 55.1, 95% CL = 52.7, 57.4 kg, *n* = 30) and non-pregnant ($\bar{x}$ = 48.4, 95% CL = 45.1, 51.7 kg, *n* = 6) yearlings, which is indicative of an effect of inadequate nutritional condition during the rut (DelGiudice et al. 2006*b*). See the summary of DelGiudice, Lenarz, and Carstensen Powell for more details of age-specific reproduction in this female study cohort.

Capturing the Variability of Winter Severity

Weather is one of the strongest environmental forces impacting wildlife nutrition, population performance and dynamics. For northern deer in the forest zone, this becomes most evident during winter when diminished abundance, availability, and nutrient quality of food resources and severe weather conditions (e.g., snow depth) impose the most serious challenge to their survival. This long-term study allowed us to capture highly variable winter weather conditions, which will facilitate a more complete examination and understanding of the relationship between winter severity, conifer cover, and the many aspects of white-tailed deer ecology that we have been investigating. The Minnesota Department of Natural Resources' (DNR) winter severity index (WSI) is calculated by accumulating 1 point for each day (temperature-days) with an ambient temperature $\leq$ -17.8° C (0° F), and 1 point for each day (snow-days) with a snow depth $\geq$ 38.1 cm (15"). The WSI for our study sites has ranged from 35 (winter 2005–2006) to 185 (winter 1995–1996). However, it is noteworthy that at least 9 of the past 16 winters (including 2005–2006) were characterized as mild (maximum WSI values well below 100, Figure 4). Although we were not capturing and radiocollaring deer during winter 2005–2006, we continued monitoring survival and cause-specific mortality. It is apparent from Figure 4, that the number of snow-days ($\bar{x}$ = 35.6 ± 9.54) during each winter tended to be less and far more variable than the number of temperature-days ($\bar{x}$ = 50.6 ± 4.09), the biological significance of which relates to our statistical analyses of age-specific survival and weather data showing that snow conditions rather than ambient temperature impose a greater challenge to deer survival (DelGiudice et al. 2002, 2006).

Mean daily minimum temperatures by month and mean weekly (julian) snow depths perhaps provide more specific depictions of the variability of winter weather conditions with which deer contended (DelGiudice 2005). To relate these conditions to deer in a more biologically meaningful way, I calculated the *effective critical temperature* for an average size adult female deer (-7° C or 19.4° F), and reported the number of days per month when the maximum ambient temperature was at or below this threshold (DelGiudice 2005). In a similar calculation, DelGiudice (2005) reported the number of days when snow cover was $\geq$41 cm (16.1"), about two-thirds chest height of adult female deer, because energetically-expensive bounding often becomes necessary at this depth and overall movements become markedly restricted (Kelsall 1969, Kelsall and Prescott 1971, Moen 1976). These presentations of weather conditions clearly exhibit the pronounced variability of days during the study period when deer experienced potentially serious energetic consequences (DelGiudice 2005).

Status, Survival, and Cause-Specific Mortality of Study Deer

Through 31 December 2005, about half of the study deer had died from wolf predation and hunter harvest (Figure 5). The "crude mortality rate" of our study deer was calculated by dividing the number of collared deer that died during a reference period (e.g., winter defined as December–May) by the total number of deer that were collared and monitored during that period. Clearly, wolf predation and hunter harvest have been the primary mortality forces impacting the female study cohort. With each year, new data collected from the field, including recaptures of does with expired collars (i.e., "lost signals"), permit revision of mortality statistics. During 1 January 1991–31 December 2005, mean annual mortality rate of collared females ≥0.5 years old was 25.3% (± 2.49%), but ranged widely from 9.1 to 47.6% (Figure 6). The female mortality rate of 2005 was average at 25.3%. As has been mentioned in previous reports, the atypical high mortality of 1992 (47.6%) was largely attributable to elevated hunter harvest (37.1%) associated with an increase in antlerless permits, whereas during 1994 and 1996, a preponderance of old females, severe weather conditions, and wolf predation contributed to the higher mortality rates (Figure 6). The number of antlerless permits issued annually varied considerably during 1991–2005, and consequently, so did the hunter harvest ($\bar{x}$ = 12.4 ± 2.46%). As reflected by the hunter-caused mortality rates (Figure 6), no antlerless permits were issued in the vicinity of our winter study sites or of the spring-summer-fall ranges of our study deer during 1996 and 1997, and very few were issued during the 1998 season. However, during 2003-2005, the permit areas in which our study sites are located were being "managed," and either-sex hunting resulted in hunter-caused mortality rates which were among the highest of the study. Mean annual wolf-caused mortality of females was 9.8% (± 1.57%) and was slightly less than average in 2005 (Figure 6). Except for during 1994 and 1996, when winters were moderately severe to severe, annual wolf-caused mortality of female deer was 4.1–14.5%. Typically, wolf predation has had its greatest impact on the older segment of the study cohort of does (DelGiudice et al. 2002). Mean age of collared female deer killed by wolves during the first 14 years of the study was 8.0 (± 0.58) years old versus 4.8 (± 1.93) years old during 2005.

The penalized likelihood estimate of the all-causes, age-specific hazard (i.e., instantaneous probability of death) for the female study cohort was U-shaped, as has been shown for humans and other mammalian species (DelGiudice et al. 2002, 2006). Including survival data of 76 neonates, we were able to show that the risk of death is most pronounced from birth to 2 years old, remains relatively low through to about age 7, and then begins an increasing trend (Figure 7). Further, 13 years of data showed that although the U-shaped curve persisted from the first 6 years of the study to the following 7 years, the position of the curve relative to the y-axis (i.e., risk of death) changed significantly (P< 0.05), reflecting a lower overall hazard, largely in response to the less severe winter weather conditions. With neonates included in the female study cohort, we learned that the median age of survival was 0.83 years, which was consistent with a separate approach focused on neonate survival (Carstensen Powell 2004). Further, through extensive statistical analyses, we explored some relatively new, and in some respects more rigorous, analytical approaches to examining survival data of wildlife (DelGiudice et al. 2002, 2006).

Mean mortality of adult collared females during June–October 1991–2005 was 4.7% (± 0.88%, Figure 8). Most of the annual non-hunting mortality of study deer occurred during winter (December–May). Mean winter mortality of adult females was 9.0% (± 1.91%, Figure 9). The highest winter mortality rates (16.2–29.3%) of does occurred during 3 of the 4 most severe winters (1993–1994, 1995–1996, and 2000–2001, Figure 9). Mortality during winter 2005–2006 was the lowest of the study (1.9%). The relationship between WSI and percent winter mortality of adult female deer continued to be reasonably strong (r^2 = 0.52, P = 0.002, Figure 10). During winters 1990–1991 to 2005–2006, predation, and wolf predation specifically, were responsible for a mean 75.2% (± 7.2, range = 0–100%, n = 16) and 63.7% (± 8.6, range = 0–100%, n = 16), respectively, of the winter mortality of collared fawn and adult females. Monthly wolf predation of females was greatest during March and April (Figure 11).

Monitoring Wolf Activity

Over the past 15 years, wolf activity on the 4 sites appears to have increased. Wolves were extirpated from the area of the study sites during the1950s–1960s, but just a few years prior to beginning the present study, wolves re-entered and became re-established. The study was on the leading edge of wolf range expansion in Minnesota. Since spring 1993, we have captured and radiocollared 57 (31 females, 26 males) wolves from 7–9 packs that range over the 4 study sites (Table 1). Fates of these wolves include being killed by a variety of human-related and natural causes.

During 1993–2001, median survival of 31 wolves from date of capture was 1,328 days (3.7 years, 90% confidence interval = 686–1,915 days) (DelGiudice, unpublished data). Human-caused mortality (e.g., shot, snared, car-kills) has accounted for 12 wolf deaths versus 6 deaths by natural causes during 1993–2006 (Figure 12).

Based on aerial observations, pack sizes have ranged from 2 to 7 members (Table 1). As is somewhat typical of wolf packs, the territories of our collared wolves have been relatively stable and have ranged in size from 62 to 186 km^2 (24–72 mi^2). Radio-location data are being used to more closely monitor wolf activity and distribution relative to the distribution and movements of collared deer. As described above, year-round monitoring and examination of mortalities of collared deer provide additional important information concerning wolf activity on the study sites.

Habitat Analyses and Updates

Detailed baseline habitat analyses using stereoscope interpretation of color infrared air photos and geographic information systems (GIS, Arc/Info and ArcView) were completed. Forest stand types were classified by dominant tree species, height class, and canopy closure class. Open habitat types, water sources, and roads were also delineated. We continue to update the coverage to account for changes in type classification associated with succession during the past 15 years. The experimental treatment (i.e., conifer harvest) impacted 157 and 83 hectares (388 and 206 acres) of conifer canopy closure classes A (< 40%), B (40-69%), and C (≥70%) on the Inguadona and Shingle Mill Lakes sites. A very 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 35 GPS-radiocollared deer (programmed to collect data at 1–6-hour intervals over 24-hour daily periods) during 2000–2006, will be used to enhance our understanding of deer use of winter cover types relative to varying weather conditions. The rigor and focus of our analytical approach relative to the overall BACI (pre-treatment, treatment, and post-treatment) design will evolve during the upcoming year in consultation with our biometrician.

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Table 1. History of radiocollared gray wolves, north central Minnesota, 1993-2006 (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 8, 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	DISPERSED	MAR 17, 2004
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	ON THE AIR	
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	ON THE AIR	
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

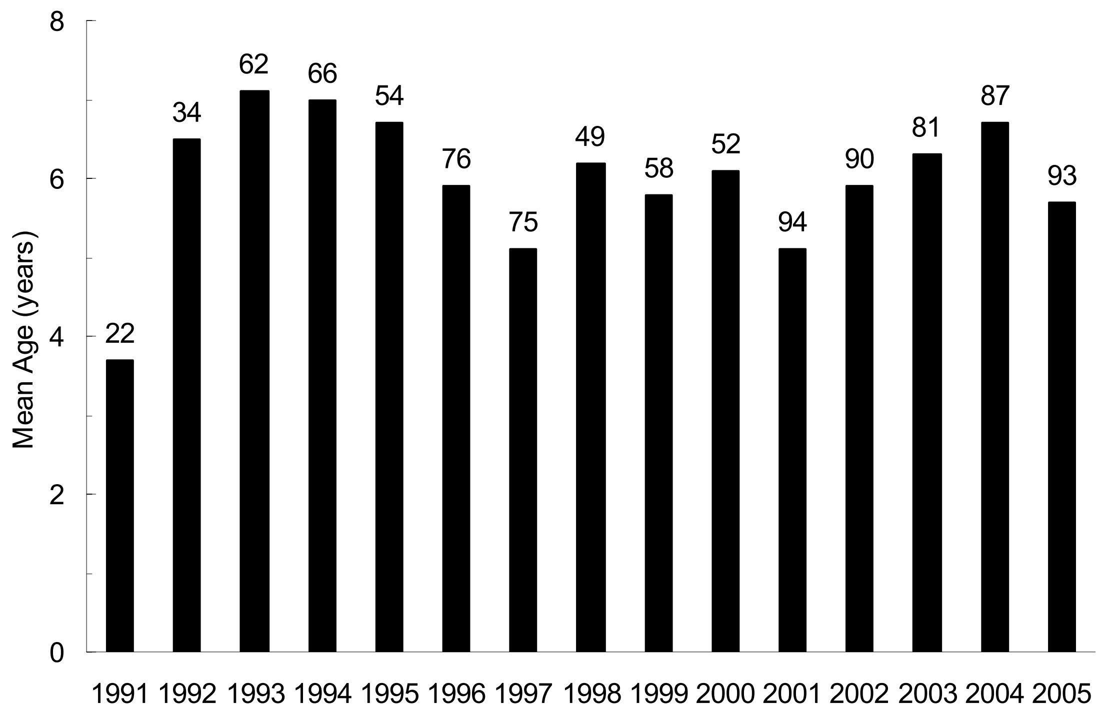


Figure. 1. Mean age of radiocollared female white-tailed deer among years, north-central Minnesota, 1 January 1991–31 December 2005. (Sample sizes are above bars.)

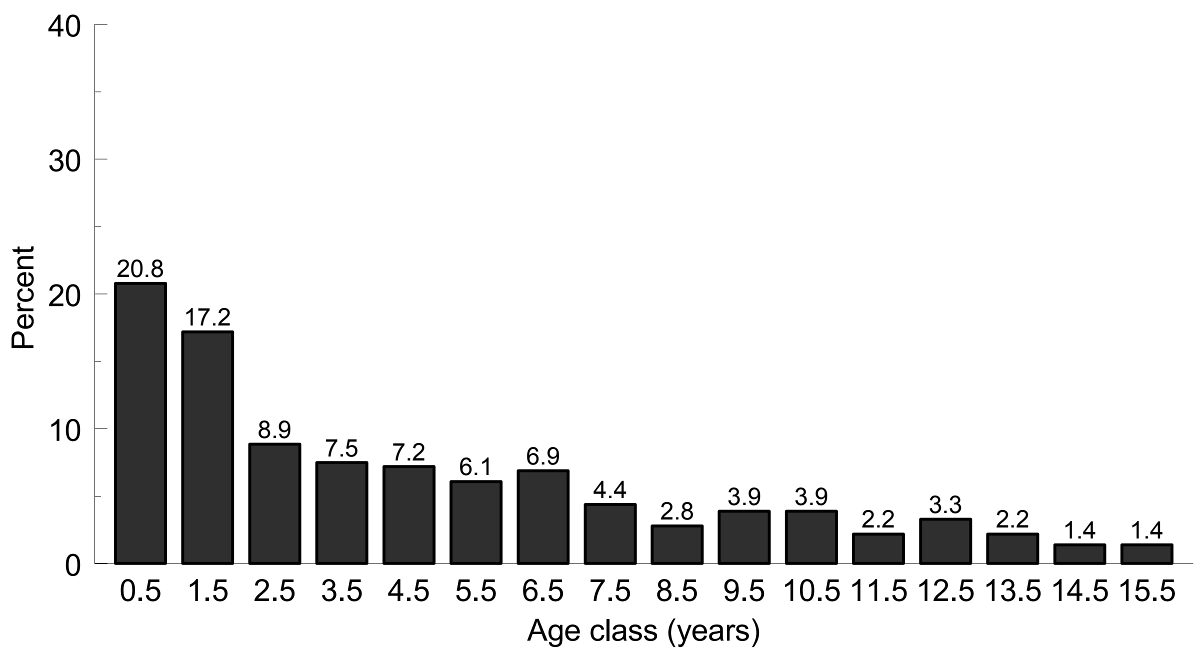


Figure 2. Age distribution of radiocollared female white-tailed deer (pooled across study sites), north-central Minnesota, 1 January 1991–31 December 2005.

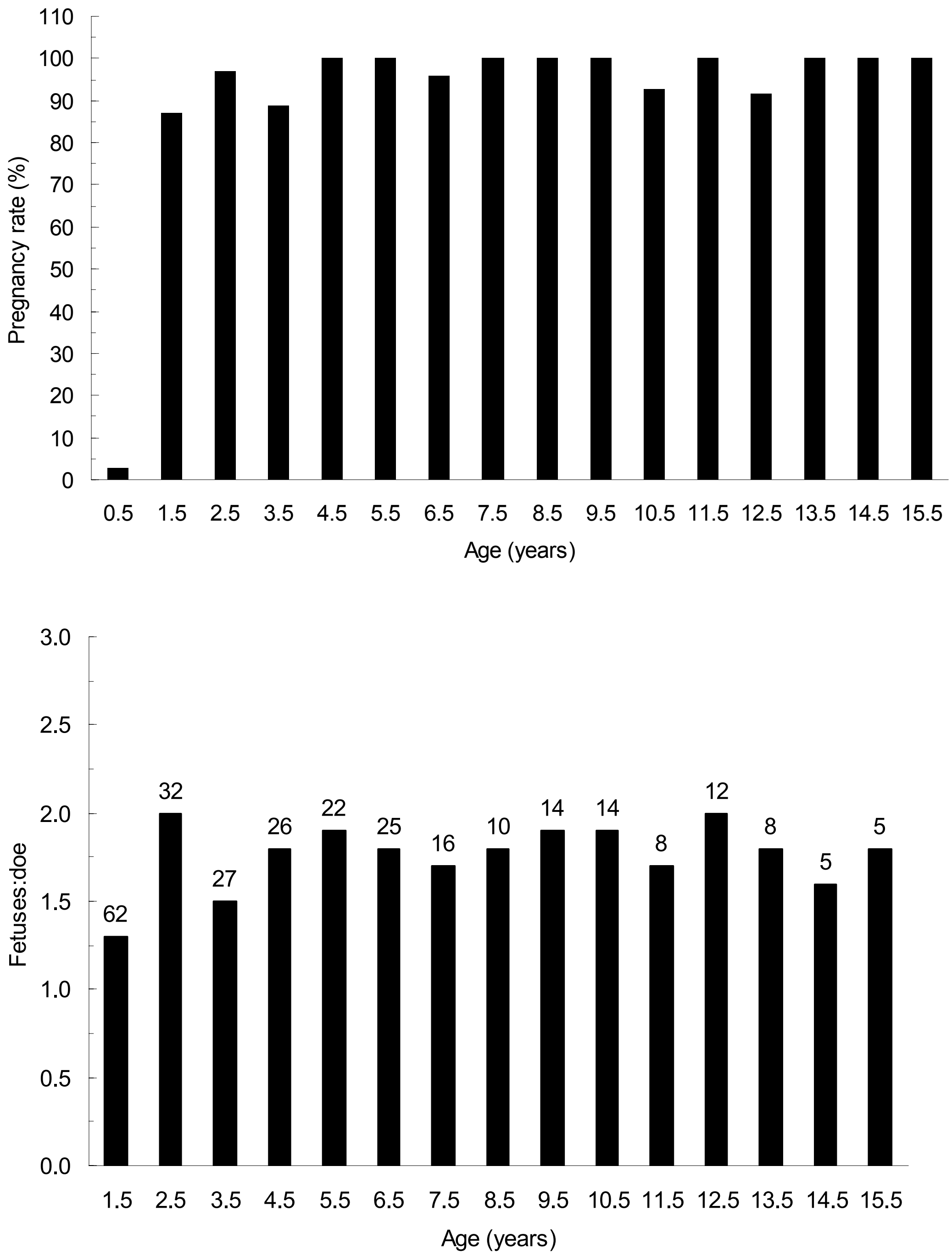


Figure 3. Age-specific pregnancy rate and fecundity (sample sizes are above bars) of radiocollared white-tailed deer (4 study sites pooled) in north-central Minnesota, winters 1991–2005.

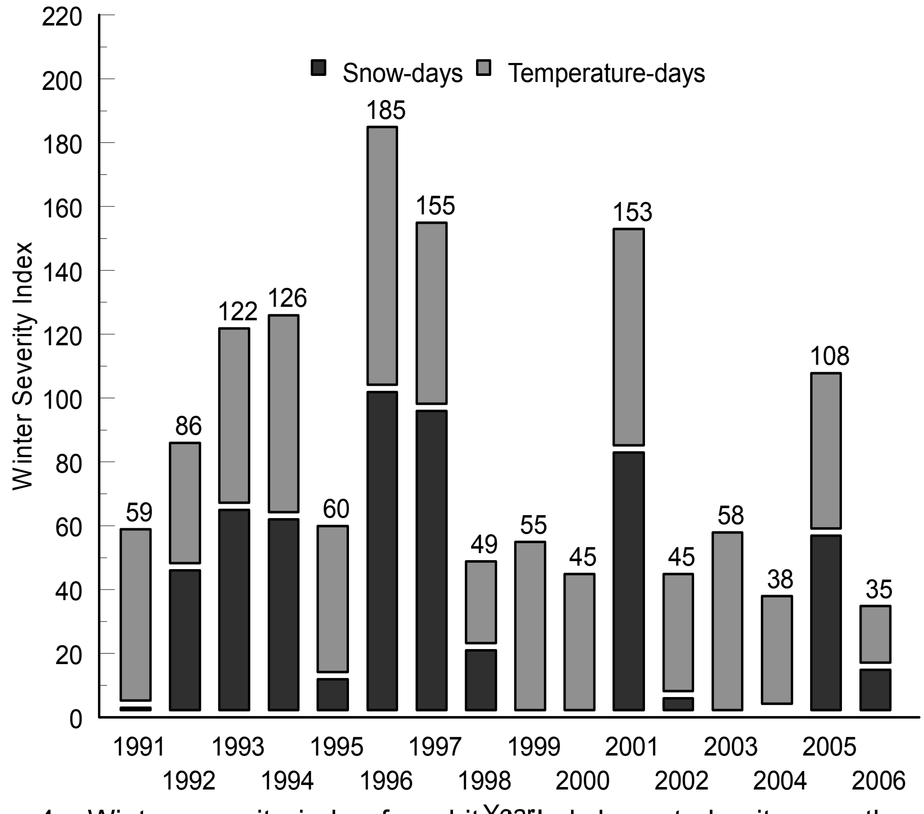


Figure 4. Winter severity index for white-tailed deer study sites, north-central Minnesota, winters 1990–1991 to 2005–2006. One point is accumulated for each day with an ambient temperature ≤ -17.8^{o} C (temperature-day), and an additional point is accumulated for each day with snow depths ≥38.1 cm (snow-day).

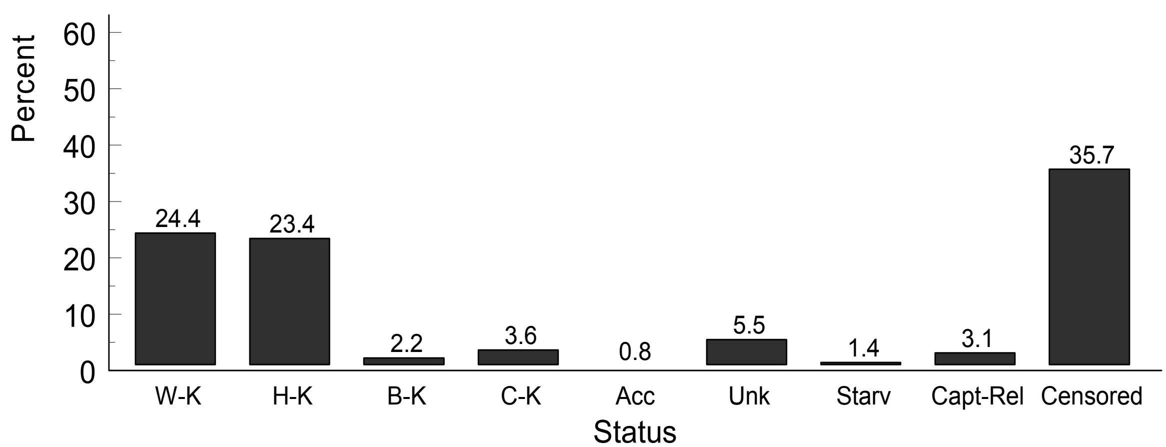
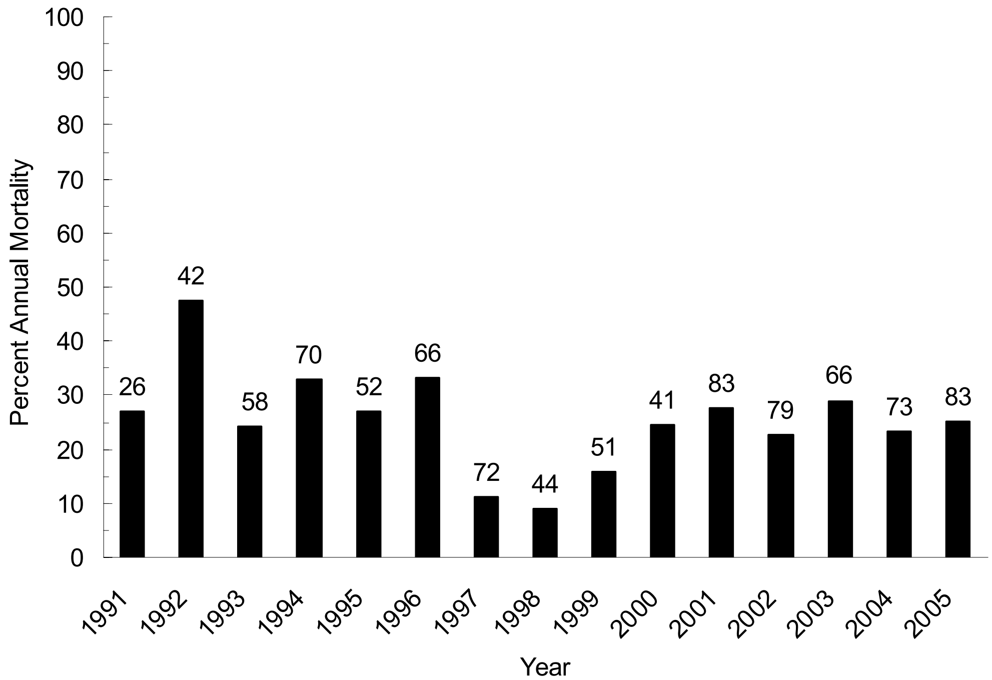


Figure 5. Status of radiocollared female deer, north-central Minnesota, 1 January 1991–31 December 2005. (W-K = wolf-kill, H-K = hunter-kill, B-K = bobcat-kill, C-K = car-kill, Acc = accidental, Unk = unknown cause, Starv = starvation, Capt-Rel = capture-related.) Censored deer include those that were still alive on 31 December 2005 or whose radio signals have been lost to monitoring (e.g., radio failure, dispersal from region of the study sites).



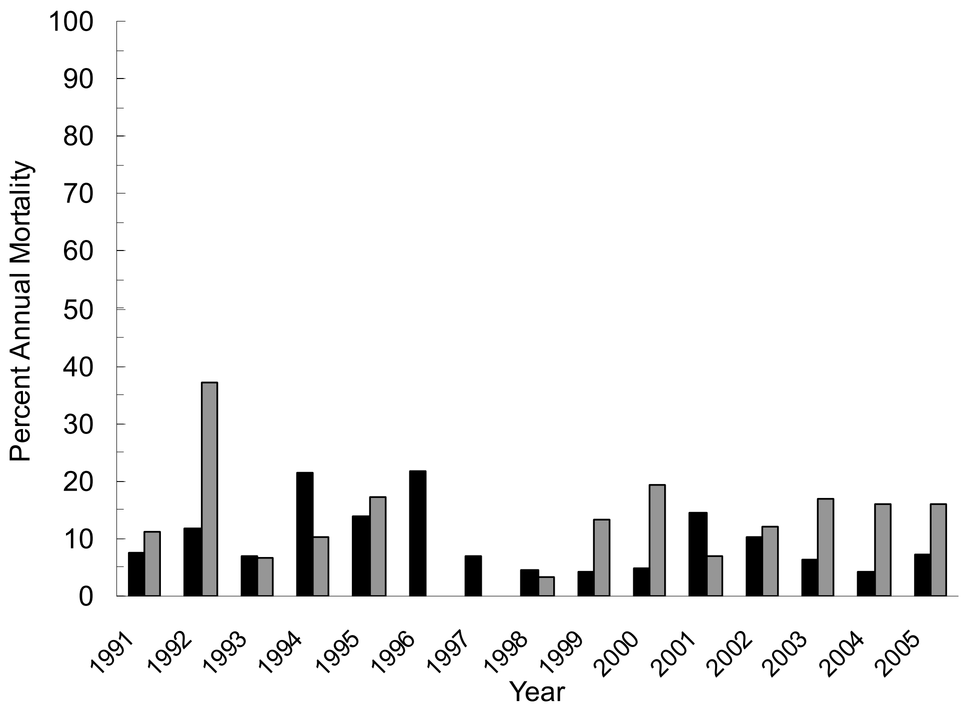


Figure 6. Annual (1 January–31 December) percent mortality of radiocollared, female white-tailed deer (top; sample sizes are above bars), and annual percent mortality attributable to wolf predation and hunter harvest (bottom, 4 sites pooled), north-central Minnesota 1991–2005. (Hunter harvest was calculated with the maximum number of collared females entering November.)

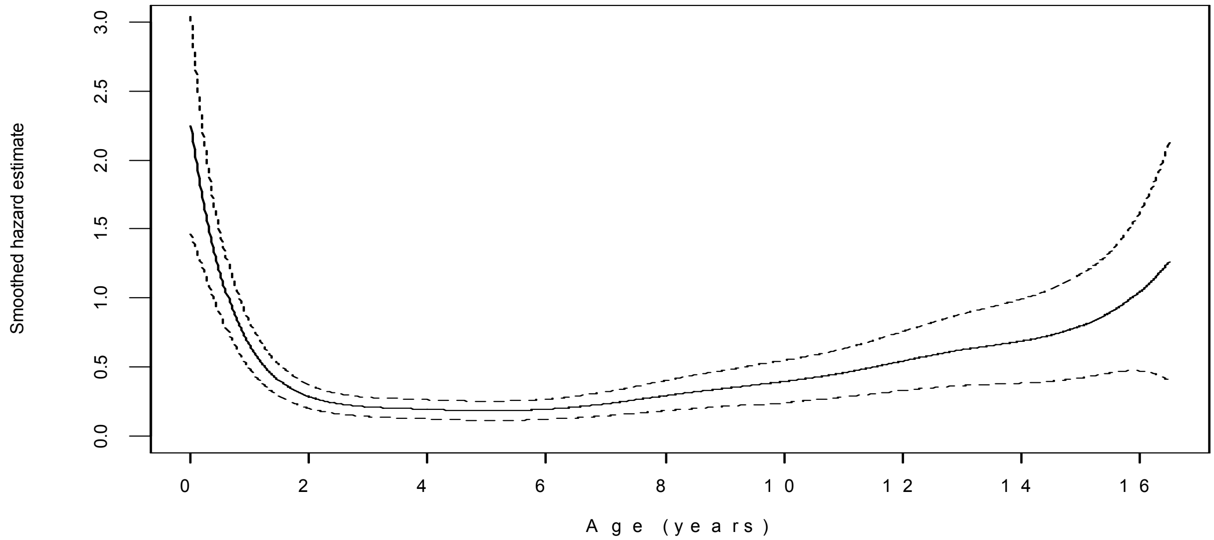


Figure 7. Penalized likelihood estimate of the 'all-causes' hazard (i.e., instantaneous probability of death) for radiocollared, female white-tailed deer (including neonates), north-central Minnesota. The data include 302 females ≥0.5 years old, monitored from 1 January to 31 December 2003, and 76 neonates (36 females, 40 males). Female neonates were monitored from 28 May 1997–31 December 2003, whereas males were censored at 0.5 years old (from DelGiudice et al. 2006).

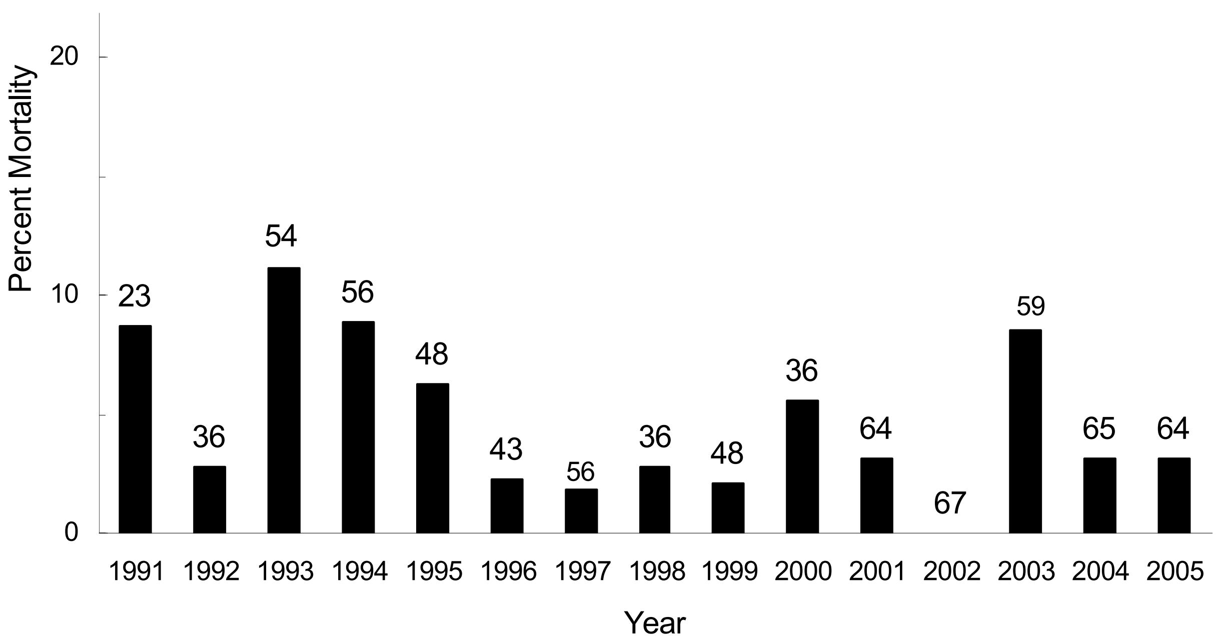


Figure 8. Percent non-winter (June-October) mortality of radiocollared, adult (≥1.0 yr old) female white-tailed deer (4 sites pooled), north-central Minnesota 1991–2005. (Sample sizes are above bars.)

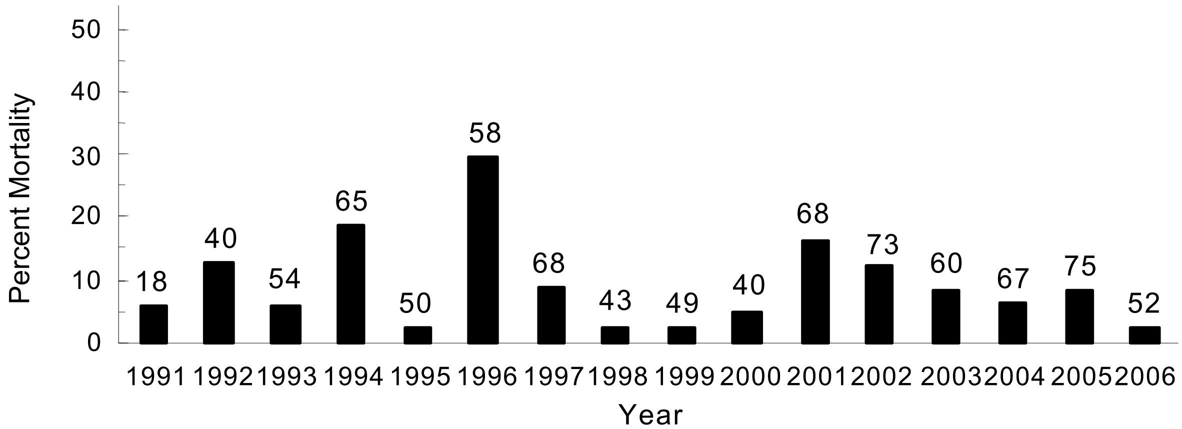


Figure 9. Percent winter mortality (December–May) of radiocollared, adult (≥1.0 year old) female white-tailed deer (4 sites pooled; sample sizes are above bars), north-central Minnesota, winters 1990–1991 to 2005–2006. (1990 = winter 1990–91, 1991 = winter 1991–92, etc…)

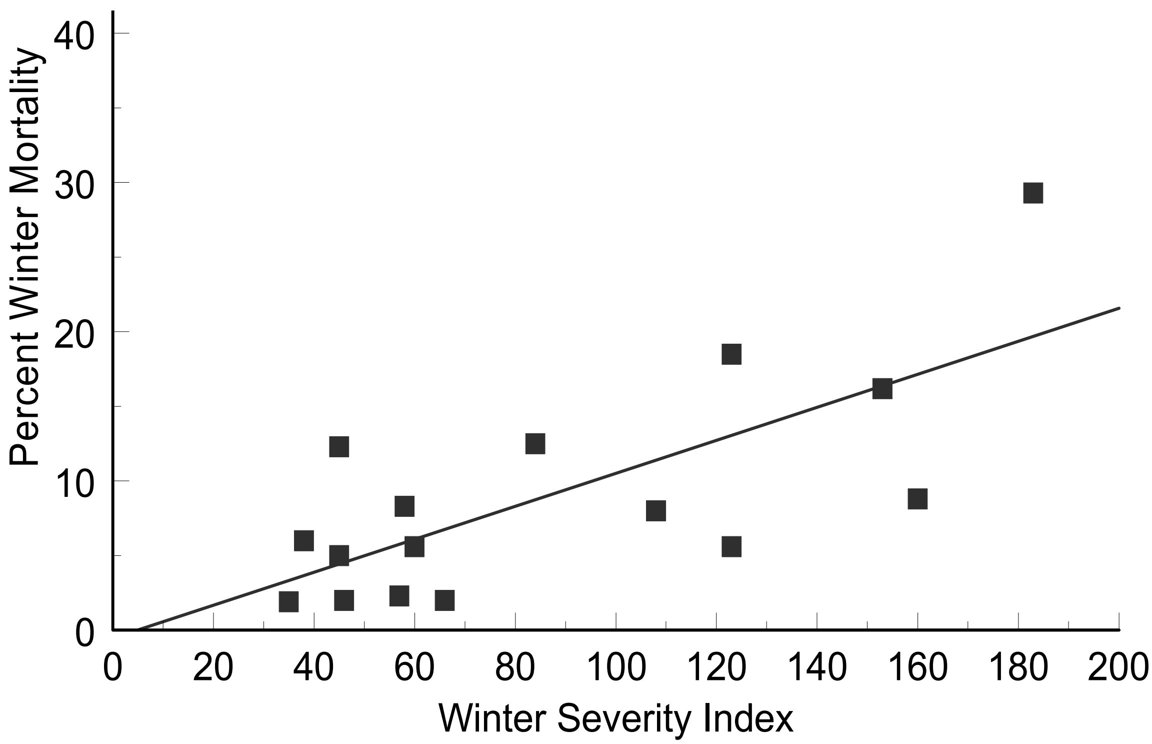


Figure 10. Relationship between Minnesota Department of Natural Resources' winter severity index (November–May) and percent winter (December–May) mortality (Y = -0.5511 + 0.1106*x*, r^2 = 0.52, *P* = 0.002) of radiocollared, adult (≥1.0 year old), female white-tailed deer (4 sites pooled), north-central Minnesota, winters 1990–1991 to 2005–2006.

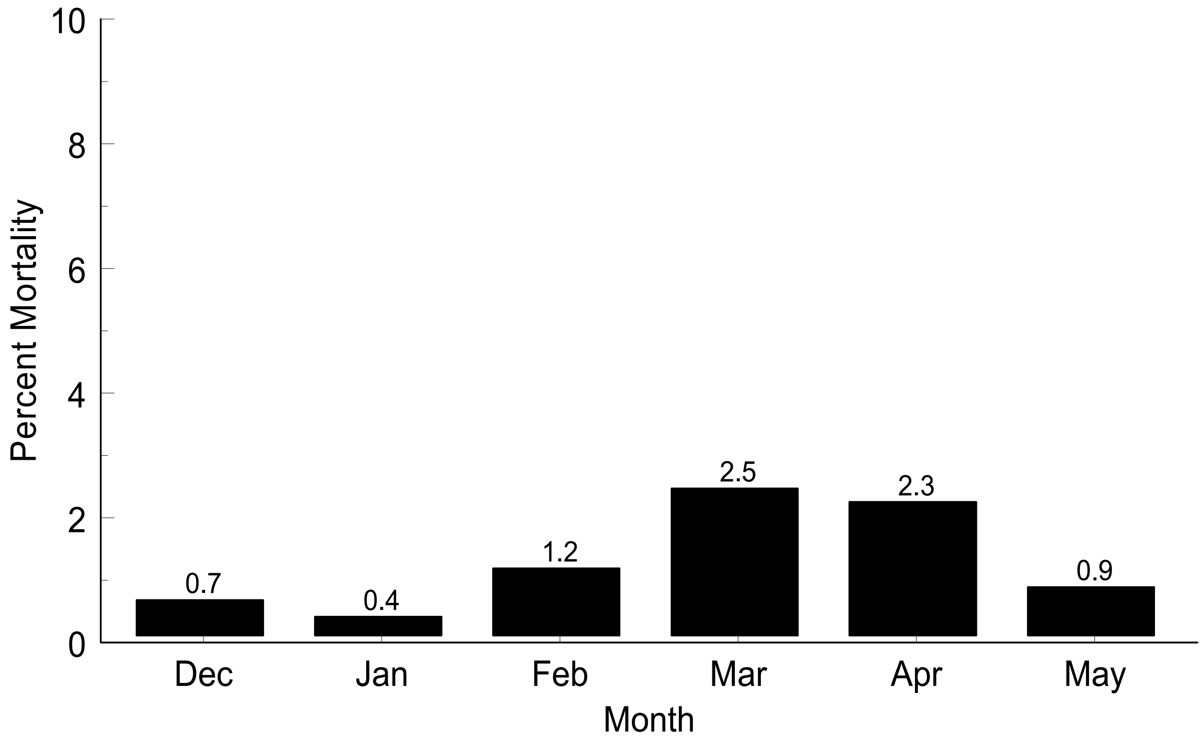


Figure 11. Monthly mortality of radiocollared female (fawns and adults) white-tailed deer caused by wolves (4 sites pooled), north-central Minnesota, winters 1990–91 to 2005–06.

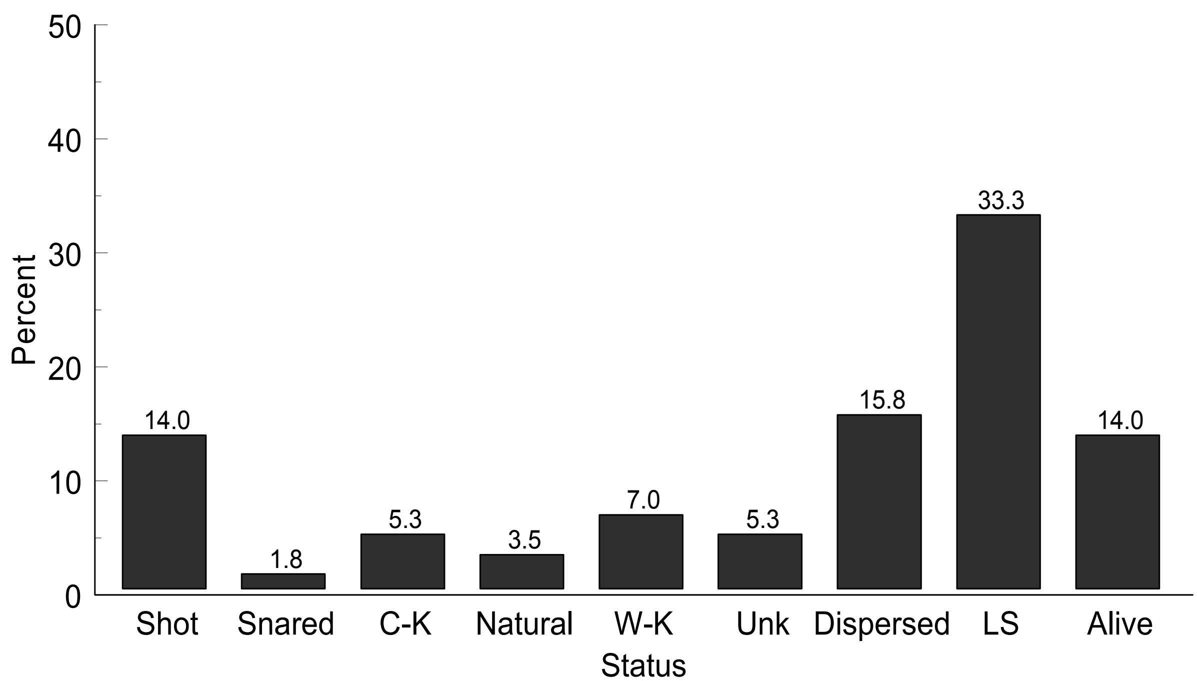


Figure 12. Status of radiocollared wolves, north-central Minnesota 1993–2006. (C-K = car-kill, Natural = natural causes, W-K = wolf-kill, Unk = unknown cause, LS = lost signal.)

SURVIVAL AND CAUSE-SPECIFIC MORTALITY OF WHITE-TAILED DEER NEONATES RELATIVE TO WINTER SEVERITY AND NUTRITIONAL CONDITION OF THEIR DAMS[1]

Michelle Carstensen Powell, Glenn D. DelGiudice, Barry A. Sampson, and David W. Kuehn

Abstract: Through maternal nutrition, winter severity may play a key role in subsequent newborn deer (*Odocoileus spp.*) survival, yet few studies of free-ranging deer have been able to establish a link between maternal body condition and survival of offspring. We captured free-ranging white-tailed deer (*O. virginianus*) neonates (n = 66) of radiocollared dams that survived severe (Winter Severity Index [WSI] = 153) and mild (WSI = 42) winters 2000–2001 and 2001–2002. Mean dates of birth (26 May ± 1.7 [SE] days and 26 May ± 1.3 days) and estimated birth-masses (2.8 ± 0.1 and 3.0 ± 0.1 kg) were similar between springs 2001 (n = 31) and 2002 (n = 35). Neonate survival was similar between years; pooled mortality rates for neonates were 14, 11, and 20% at 0–1, 2–4, and 5–12 weeks of age, respectively. Predation accounted for 86% of mortality, the remaining 14% of deaths were attributed to unknown causes. Black bears (*Ursus americanus*) were responsible for 57 and 38% of predation on neonates in springs 2001 and 2002, whereas, bobcats (*Felis rufus*) accounted for 50% in 2002. Wolves (*Canis lupus*) accounted for only 5% of predator-related deaths. Birth characteristics and blood profiles of neonates were examined as potential predictors of survival. Low birth-mass, reduced body size (e.g., girth and hind leg length), and elevated serum urea nitrogen (SUN, 26.1 ± 2.6 vs 19.3 ± 0.8 mg/dL) and tumor necrosis factor-α (TNFα, 82.6 ± 78.6 vs 2.3 ± 0.5 pg/mL) were reported in non-surviving versus surviving neonates by 1 week of age. Dams with reduced fat reserves during winter subsequently lost more neonates within 12 weeks of birth. Also, dams (n = 3) of neonates that died at 2–4 weeks of age had greater (P < 0.05) concentrations of SUN (19.0 ± 4.5 vs 11.1 ± 1.1 mg/dL) and creatinine (C, 2.7 ± 0.2 vs 2.3 ± 0.1 mg/dL) than dams (n = 20) of survivors. Even though a direct relation between winter severity and birth or blood characteristics of neonates was not detected in this study, evidence suggests that birth-mass and key serum indices of neonate nutrition were associated with their survival. Further, we were able to link winter severity and nutritional restriction of dams to reduced survival of their offspring. Clearly, additional study of free-ranging populations is needed to enhance our understanding of factors that may predispose neonates to natural sources of mortality.

[1] Abstract of manuscript submitted to the Journal of Wildlife Management.

MOOSE POPULATION DYNAMICS IN NORTHEASTERN MINNESOTA

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

SUMMARY OF FINDINGS

A total of 114 moose *(Alces alces)* (54 bulls and 60 cows) have been captured and collared since the study began in 2002. As of 31 March 2006, 55 collared moose (29 bulls and 26 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 two 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.

In 2002, moose were captured with 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

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sensor that increased the pulse rate (mortality mode) if it remained stationary > 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 2004-2006 aerial moose surveys. During the survey, test plots were identified that contained one 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 2006. 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 2006, 55 collared moose (29 bulls and 26 cows) have died. The cause of death in 23 cases could be identified (12 hunter kill, 2 poached, 5 train/ car/truck collision, 3 wolf predation, 1 natural accident, and 1 bacterial meningitis). Three 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 4 moose. Two died within the BWCAW and in 2 cases, we only found the radio-collar. Twenty-five collared moose appear to have died from unknown non-traumatic causes. In 10 cases, scavengers had consumed the carcasses, but evidence suggested that predators might not have killed them. In the remaining 15 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 (3 - January, 1 – March, 1 - April, 6 - May, 2 - June, 2 - July, 4 - August, 1 - October, 2 - November, 3 - December). To date, samples from unknown cases have tested negative for CWD, Rabies, Eastern Equine Encephalitis, and West Nile Virus. Sera from captured moose were tested for BVD, borreliosis (Lyme's disease), lepto, 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 hunter harvested moose 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, 3 died of unknown causes, a hunter killed 1, and 1 is listed as capture related because it died within 2 weeks of capture. Only 2 of the 5 skulls were examined for the presence of *P. tenuis* with results being positive in one case, negative in the other.

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

Limited data suggest that calf mortality was lower than in northwestern Minnesota. In late May 2004, 14 of 18 collared cows were accompanied by one or more new born calves (9 singles, 4 twins, 1 triplet). Three of the 4 calf-less cows were subsequently observed with a single calf. Twelve of the 23 calves (52%) survived until early May of 2005. In northwestern Minnesota, the average annual calf survival was 66% (Cox et al. In press). In late May 2005, 18 of 26 collared moose were accompanied by calves (16 singles, 2 twins). All 8 of the calf-less cows were subsequently observed with one or more calves (6 singles, 2 twins). A survival check will be conducted in late April 2006.

In January 2006, radio collared moose were located 38 times in the process of developing a sightability model. In 20 cases, the collared moose was observed using the standard survey protocol. In 18 cases, the collared moose was not observed, and telemetry had to be used to locate the collared moose. Six different models were evaluated, and the model with the highest predictive reliability incorporated a single covariate, visual obstruction (Giudice and Fieberg, unpublished). Total population size based on this sightability model was 7,272±26%, an estimate not significantly different from the 2005 estimate (6,519±30%) Ultimately, with additional data, this model will improve the accuracy and precision of the aerial survey.

ACKNOWLEDGMENTS

We thank the collaborators in this study including Glenn DelGiudice and Barry Sampson from DNR's Forest Wildlife Populations and Research Group, and Jim Rasmussen from the Minnesota Zoo for their assistance in capturing moose and collecting biological samples. We also thank Al Buchert, Mike Trenholm, Brad Maas, and John Heineman from the DNR's Enforcement Division for their piloting skills throughout the project. John Fieberg and John Giudice were essential to the development of the sightability model.

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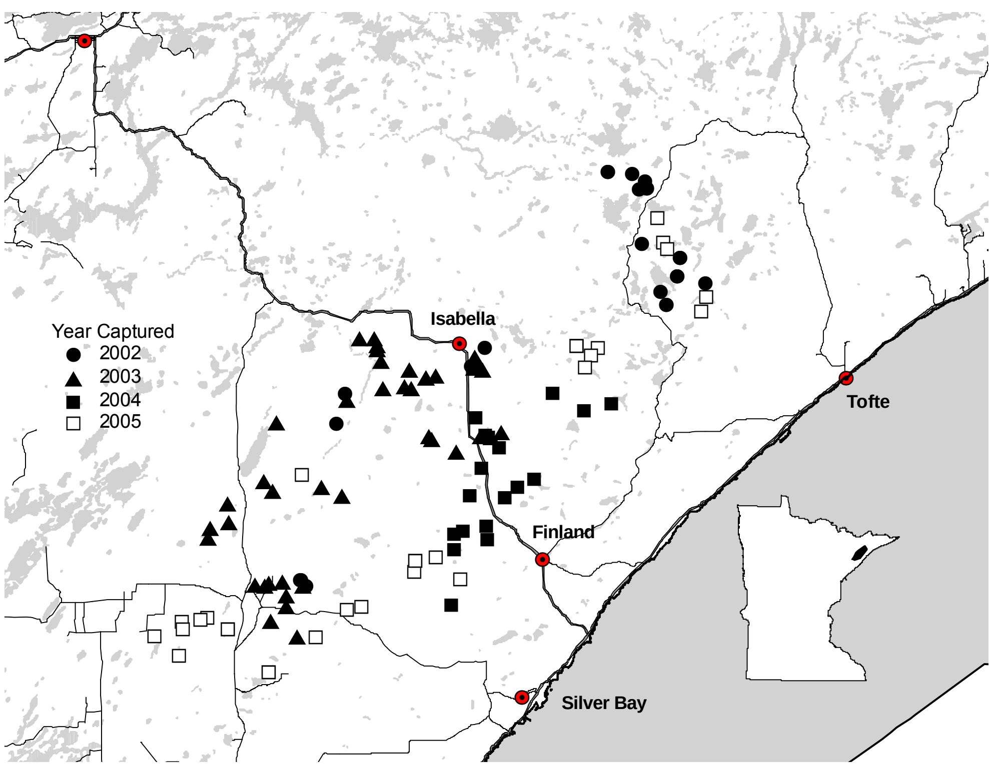
Table 1. Annual non-hunting and total mortality of collared moose. Number of collared moose in sample at beginning of calendar year is listed in parentheses.

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

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



IDENTIFYING PLOTS FOR SURVEYS OF PRAIRIE-CHICKENS IN MINNESOTA

Michael A. Larson

SUMMARY OF FINDINGS

To explore potential improvements in surveys of greater prairie-chickens *(Tympanuchus cupido pinnatus)* 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 PLS sections (~259 ha) 3 times during April and early May of 2005 to detect leks. 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. Preliminary analyses revealed no other landscape characteristics that were correlated with the probability of occupancy. Additional modeling and analysis may provide more inferences about predicting occupancy by prairie-chicken leks. 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 (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 prairie-chicken leks is determining where to survey within that range.

The availability of 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), and 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

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 effort by the 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.

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; and the "hat" character (e.g., $\hat{\psi}$) denotes the estimated value of a quantity. Additionally, *c* is 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 Public Land Survey (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.

Data collection

An observer visited each sample plot once during each of *T* = 3 consecutive biweekly periods from 4 April 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 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). Finally, 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).

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 $n_{\mathrm{nonoverlap}}$ = 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*.

Model fitting is not complete for this study. Excluding the domain parameter from the models may help reveal landscape characteristics that differentiate occupied and unoccupied plots. I will also fit the *a priori* models for ψ using different sources of land cover data (e.g., GAP level 3), one of which may prove more useful in discriminating occupied from unoccupied plots. Furthermore, by simplifying the model for *p* to include only the dominant 4 covariates (rather than all 16) in an exploratory analysis, both *a priori* and *a posteriori* models for ψ may reveal stronger relationships between occupancy and characteristics of the landscape.

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.

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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 (disturbance1)	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.

Probability	Parameter[a]	Estimated value	95% confidence limits 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.

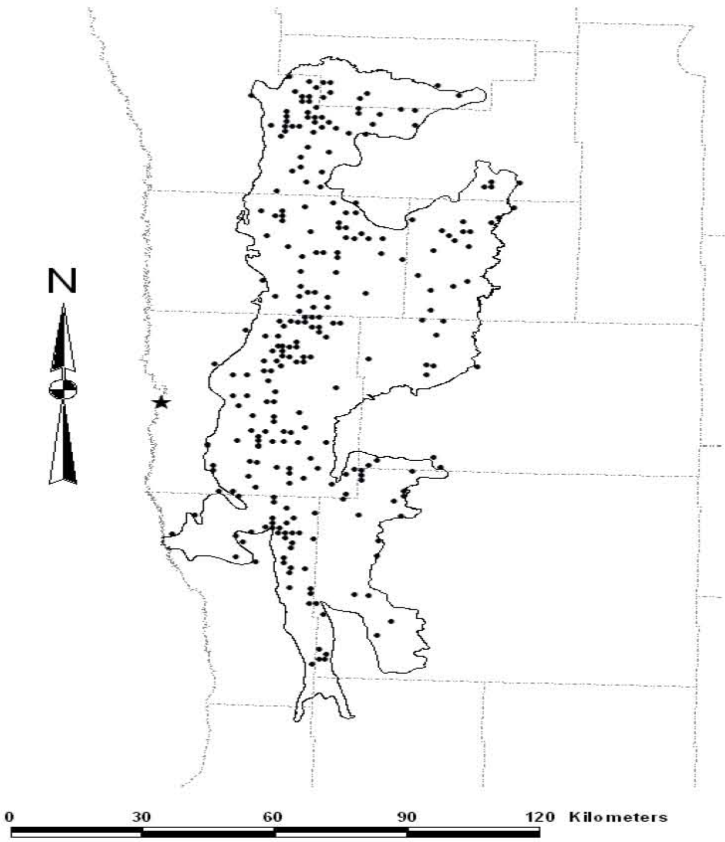

Figure 1. The Northwest prairie-chicken range based on land type associations of the Ecological Classification System (solid line) relative to county boundaries (dashed lines) in western Minnesota. Sample plots (dots) were not selected from areas >90 km southeast of Moorhead (star).

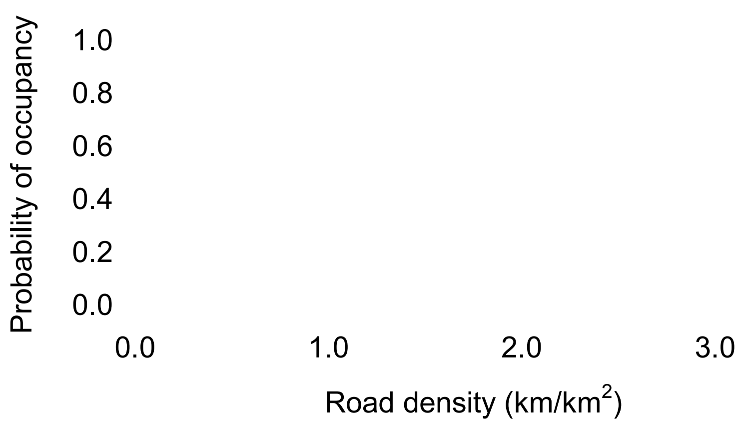

Figure 3. Predicted probabilities (heavy lines) and 95% confidence intervals (light lines) of a sample plot in Minnesota being occupied by a prairie-chicken lek during spring of 2005 over the observed range of road densities in the overlap domain (i.e., plots known to have contained a lek during 2004;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

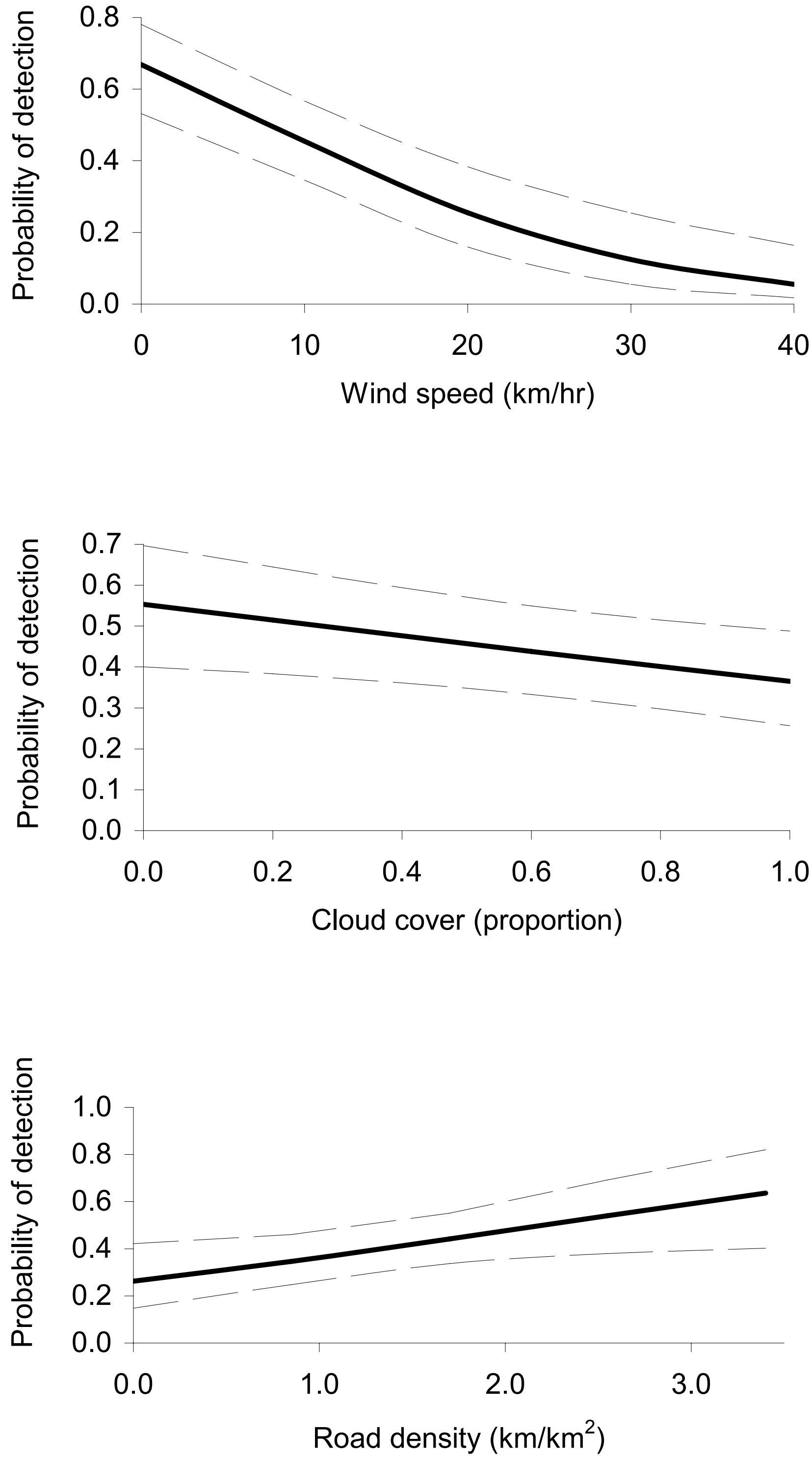

Figure 2. Predicted probabilities (and 95% confidence intervals) of detecting a prairie-chicken lek in sample plots in Minnesota during spring of 2005 over the range of observed values of selected model parameters.

AN EVALUATION OF DRUMMING COUNT SURVEYS OF RUFFED GROUSE IN MINNESOTA

Michael A. Larson

SUMMARY OF FINDINGS

The relationship between drum counts and true abundance of ruffed grouse (*Bonasa umbellus*) has not been established, and it is unknown to what extent currently active routes of the Drum Count Survey (DCS) are representative of all areas within the range of ruffed grouse in Minnesota. I developed this study to determine the most appropriate way to analyze DCS data collected under the current protocol and to propose and evaluate alternative monitoring protocols. I started quantifying and reporting the level of uncertainty in mean drum counts for the 2005 grouse survey report using bootstrap samples to estimate confidence intervals. To help determine appropriate regional boundaries for summarizing DCS results, I calculated correlation coefficients among annual mean drum counts in the 7 Ecological Classification System (ECS) sections in Minnesota that are forested. I also selected bootstrap samples of different numbers of routes within the newly defined DCS regions to determine if survey effort should change or be reallocated among regions to achieve sufficient precision to detect large changes in mean drum counts among years. The evaluation of alternative monitoring protocols has not begun yet. Drum counts during the last 2 cycles of the ruffed grouse population were highly correlated among the 4 ECS sections comprising the Laurentian Mixed Forest (LMF) province, which covers the core and bulk of the range of ruffed grouse in Minnesota. Correlations were lower and apparent long-term population dynamics were noticeably different for the other 3 ECS sections, which are along the periphery of ruffed grouse range. Within those 4 new DCS regions, the number of routes could be reduced from ~95 to 30–60 without losing much precision in the LMF province, could remain approximately the same in the Lake Agassiz and Aspen Parklands section (n = 8 routes) and Paleozoic Plateau section (n = 5 routes), and should perhaps be increased from 14 to ≥25 routes in the Minnesota and Northeast Iowa Morainal section.

INTRODUCTION

The Minnesota Department of Natural Resources (DNR) has conducted counts of drumming ruffed grouse (*Bonasa umbellus*) annually since 1949 (Petraborg et al. 1953). The Drumming Count Survey (DCS) occurs along roads during spring and consists of observers driving a route approximately 16 km long, stopping 10 times at approximately equal intervals to listen for 4 minutes to count the number of drums heard (not drummers detected). Traditionally, counts were averaged across routes within 5 regions of the state. The DCS is intended to document the trajectory of the ruffed grouse population throughout its range in Minnesota. In practice, the DCS is used almost exclusively to inform the public about the status of the ruffed grouse population relative to its periodic cycle of abundance, which repeats approximately every 10 years. Mean counts throughout ruffed grouse range within the state (hereafter, state-wide) were correlated (r = 0.82, n = 26) with the number of ruffed grouse killed by hunters during the subsequent autumn (Berg 1977). Counts also appear to be correlated with rates of harvest (i.e., number of grouse killed per hunter).

Despite the apparent success of the DCS, its current implementation limits the quality and validity of inferences that can be made. The DCS is treated as an index, but the relationship between counts and true abundance has not been established. Many factors potentially confound the count–abundance relationship (e.g., proportion and frequency of grouse drumming, probability of detection), but none are accounted for under current protocols. The inherent assumption that the relationship does not vary in a systematic pattern either spatially or temporally (Yoccoz et al. 2001, Pollock et al. 2002) is not supported by theory or empirical evidence (Gullion 1966, Rodgers 1981).

Making inferences about drumming counts or the abundance of ruffed grouse in regions of the state or state-wide requires that the locations at which the DCS is conducted are representative of the larger area of interest. The routes along which drumming counts are made likely are not representative because they were not established under a probabilistic spatial sampling design. Most routes were established by local wildlife managers, who undoubtedly used different criteria for deciding where to place each route. Furthermore, the number of routes established in an area may have depended upon the interest of the local manager or other cooperator in monitoring the ruffed grouse population.

Deciding whether to change the DCS will depend upon the benefits of potential improvements and the costs associated with them. I will propose and investigate methods for improving the statistical validity of the DCS. I will also compare current and alternative methods using statistical (e.g., precision) and logistical (e.g., investment of time) criteria. This study, therefore, will provide a scientific basis for deciding which DCS design and analysis protocol to implement. Potential changes to the DCS may increase the usefulness of the resulting data for monitoring the effectiveness of management activities, validating relationships between the fitness of ruffed grouse and characteristics of their habitat, analyzing the causes of the 10-year cycle in abundance, and informing ruffed grouse hunters about the likelihood of their success.

Objectives

- 1. To determine the most appropriate way to analyze DCS data collected under the current protocol.
  - (a) Estimate precision of mean drumming counts from existing survey data.
  - (b) Determine appropriate regional boundaries for reporting results from the DCS based on ecological land classifications and the spatial scale of homogeneity in drumming counts.
  - (c) Determine the effect on precision of changing the number of routes in the DCS.
- 2. To propose and analyze the efficacy of alternatives to the current methods of collecting and analyzing data from the DCS by addressing the issues of bias and precision in resulting estimates of abundance and population trajectory.

METHODS

The statistic of interest, or index value, from the DCS is the number of drums heard per stop (i.e., drums/stop, or dps). Given that the route, not the stop, is the sampling unit, the mean dps for each route is calculated first. Then the mean dps for a geographic area is calculated as the mean of route-level means. The precision of index values, however, typically has not been reported. I used 10,000 bootstrap samples of route-level means to estimate a percentile confidence interval (CI) for mean index values for each of the 5 ruffed grouse zones and each of the 7 Ecological Classification System (ECS) sections in the ruffed grouse range (Figure 1). These 95% CIs quantify the uncertainty in the mean index values.

The analysis of precision was conducted for an annual survey report, so I used all historic DCS data that were available in digital format (i.e., 1982–2005). I used data from 1984–2004 for all other analyses in this study. That range of years included only the last 2 full population cycles; 1983, 1993, and 2004 were thought to be the last 3 low points of the cycle.

Appropriate regional boundaries for reporting results from the DCS should combine areas with ruffed grouse populations with similar long-term population dynamics and separate areas with populations whose long-term dynamics are less similar. To define boundaries that meet that definition, I relied on Spearman's rank correlations among annual mean drum counts in the 7 forested ECS sections. I supplemented the correlation analysis by considering qualitative similarities and differences among ECS sections in graphs of annual mean drum counts over time.

Whereas comparisons of drum counts over time is valid if the sample of routes in an area is representative of all potential route locations (i.e., mean survey conditions remain relatively constant over time), comparisons of counts among geographic areas is not. The relationship between DCS counts and actual ruffed grouse densities is unknown, so differences in the magnitude of drum counts between areas could be due to a number of factors unrelated to populations of ruffed grouse. For example, observed counts in 2 areas with identical densities of ruffed grouse could differ substantially due entirely to differences in the mean level of traffic noise along survey routes or other characteristics of route locations in the 2 areas. When evaluating potential regional boundaries for summarizing DCS results, therefore, I deemed comparisons of the magnitude of mean drum counts among ECS sections much less important than the criteria mentioned in the previous paragraph.

If the DCS continues with no substantive changes, it may be desirable to reallocate survey effort among survey regions, or it may be possible to reduce survey effort and still retain sufficient precision. I evaluated the effect of the number of routes on the precision of mean counts by selecting 10,000 bootstrap samples of various percentages greater and less than the existing number of routes. For each of the new DCS regions [see objective 1(b)], I used data from 2 years between which the DCS should indicate a significant difference in mean counts (i.e., nonoverlapping 95% CIs). In most cases they were the most recent low and high points in the approximately 10-year cycle (i.e., 2004 and either 1998 or 1999). If it is desirable for the DCS to document smaller differences in mean counts, this portion of the analysis could be expanded to include more conservative minimum differences.

The portion of this study related to objective 2 has not begun yet, so the methods are not provided.

RESULTS & DISCUSSION

I estimated the precision of mean drum counts during preparation of the 2005 grouse survey report, so complete results for objective 1(a), including time-series graphs of mean drum counts and CIs for each ruffed grouse survey zone and ECS section for each of the last 24 years, are available in that document (Larson 2005). Median index values for bootstrap samples were within 0.03 dps of the 120 survey means by zone and 0.06 dps of the 168 survey means by ECS section for all annual estimates since 1982. Furthermore, bootstrap medians were within 0.02 dps of 89% of the survey means by ECS section. Therefore, no bias-correction was necessary, and CI limits were defined as the 2.5^{th} and 97.5^{th} percentiles of the bootstrap frequency distribution.

Analysis of historical data indicated that precision in the counts was correlated with the magnitude of mean counts (r = 0.78 , n = 168 = 24 years $\times$ 7 ECS sections). These analyses of precision were useful for interpreting changes in mean drum counts among years, and they will facilitate direct comparison with the precision of estimates resulting from alternative survey methods.

The correlations in annual mean drum counts were greatest among the 4 ECS sections of the Laurentian Mixed Forest (LMF) province (i.e., Northern Superior Uplands = NSU, Northern Minnesota and Ontario Peatlands = MOP, Northern Minnesota Drift and Lake Plains = DLP, and Western Superior Uplands, including a small portion of the Southern Superior Uplands in eastern Carlton County = WSU; $\overline{r}$ = 0.67, range = 0.40–0.93, n = 6 2-way comparisons). Understandably, the lowest correlation in that group was between mean counts in the MOP and WSU sections, which were the only 2 that did not share a border. The correlation between annual mean drum counts in the Minnesota and Northeast Iowa Morainal (MIM) section and those in the 4 sections of the LMF province was somewhat less ($\overline{r}$ = 0.59, range = 0.42–0.69, n = 4). Annual mean drum counts in the Lake Agassiz and Aspen Parklands (AAP) section were most highly correlated with those in adjacent sections (r = 0.4 and 0.6 with the MIM and MOP sections, respectively) but were much less correlated with those in the other sections ($\overline{r}$ = 0.24, n = 4). Correlations were least between annual mean counts in the Paleozoic Plateau section and the other sections ($\overline{r}$ = 0.03, n = 6).

Qualitative comparisons of annual mean drum counts among the ECS sections followed patterns similar to those in the correlation results. Drum counts in the sections of the LMF province exhibited distinct, dramatic fluctuations corresponding with the approximately 10-year population cycle (Figure 2). Drum counts in the MIM section exhibited distinct but much less dramatic long-term fluctuations, and those in the AAP section exhibited minor, erratic fluctuations relative to the population cycle. Drum counts in the PP section exhibited a long-term decline and no cyclical pattern (Figure 2).

I recommend, therefore, that results from the DCS be summarized in 4 regions—the LMF province and the other 3 ECS sections. The LMF province represents an ecologically meaningful combination of sections that corresponds well with the core and bulk of the range of ruffed grouse in Minnesota. Drum counts in the AAP, MIM, and PP sections exhibited distinctly different long-term patterns than those in other sections. This was intuitively compelling because those sections are in the periphery of the range of ruffed grouse in Minnesota and they support vegetation communities that differ markedly in the quantity and quality of habitat they provide for ruffed grouse.

In the LMF province, bootstrap sample sizes of ≥20 routes produced 95% CIs that did not overlap for the most recent low and high points in the population cycle (mean dps = 0.80 and 2.06, respectively; Figure 3). Currently there are >90 active routes in the province, so many of them could be eliminated without adversely affecting the precision of mean drum counts. Once at least 60 routes were included, the increase in precision from the addition of routes was minimal (Figure 3).

In the AAP section, the CIs did not overlap when ≥6 routes were included (Figure 4). In the MIM section, ≥25 routes were required to produce nonoverlapping CIs (Figure 5). Route-level means in the MIM section were more variable than in the other 2 peripheral sections, so increasing the number of routes there from the 14 that are currently active could be justified by a need for greater precision. In the PP section, ≥5 routes were required for sufficient precision (Figure 6). The existing number of active routes in the AAP and PP sections (*n* = 8 and 5 routes, respectively) was sufficient for an adequate level of certainty in detecting the selected magnitude of change in drum counts. Inclusion of 15 routes in each of these 2 sections likely would produce sufficient precision to detect a difference of 0.5–0.6 between 2 mean counts.

ACKNOWLEDGMENTS

I sincerely appreciate the help of all the DNR staff and volunteer cooperators who have conducted ruffed grouse surveys. I also appreciate the efforts of Bill Berg, who coordinated the collection of grouse survey data for many years, and John Erb and others who translated most of the grouse survey data into a digital format.

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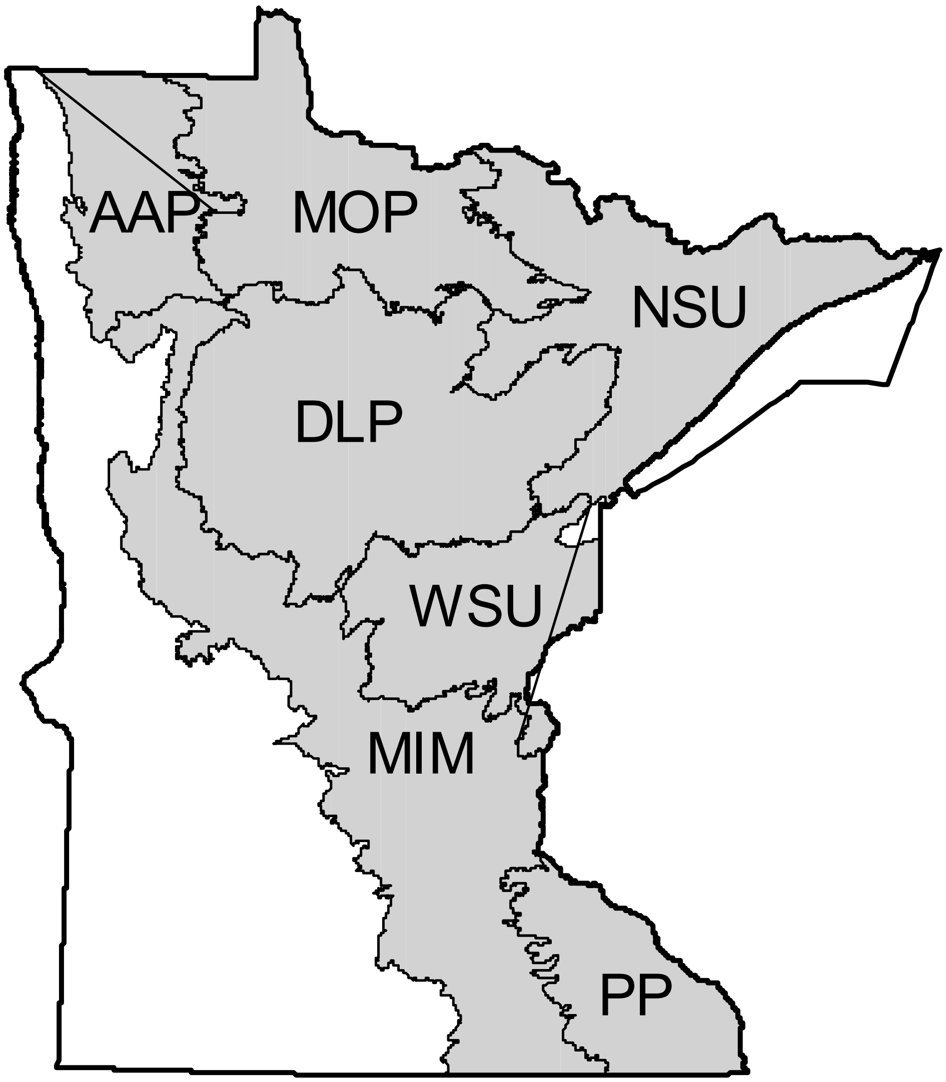
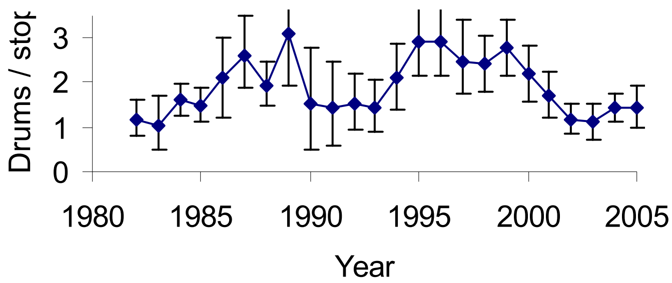


Figure 1. Forested sections of the Ecological Classification System in Minnesota. The MOP, NSU, DLP, and WSU sections constitute the Laurentian Mixed Forest province. AAP = Lake Agassiz & Aspen Parklands, MOP = Northern Minnesota & Ontario Peatlands, NSU = Northern Superior Uplands, DLP = Northern Minnesota Drift & Lake Plains, WSU = Western Superior Uplands (including a small portion of the Southern Superior Uplands in eastern Carlton County), MIM = Minnesota and Northeast Iowa Morainal (only the northern half of which is surveyed for ruffed grouse), and PP = Paleozoic Plateau.



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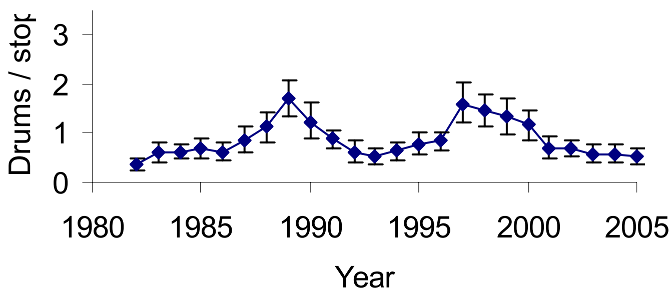

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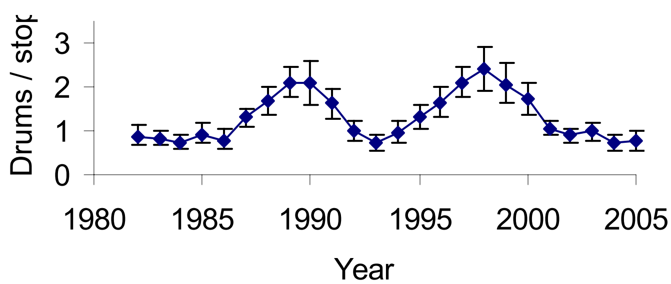
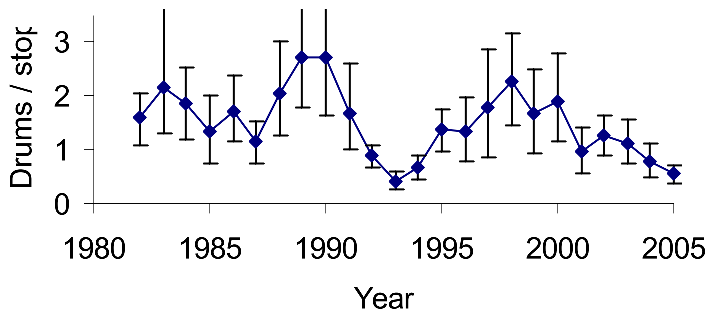


C.

Figure 2. Ruffed grouse drum count index values in the 7 forested ECS sections in Minnesota (Panel A: MOP, panel B: NSU, panel C: DLP, panel D: WSU, panel E: AAP, panel F: MIM, panel G: PP; abbreviations explained in the text and the caption for Figure 1). Vertical error bars represent 95% confidence intervals based on bootstrap samples. The upper end of 7 error bars were truncated so the scale of the y-axis would be identical for all panels.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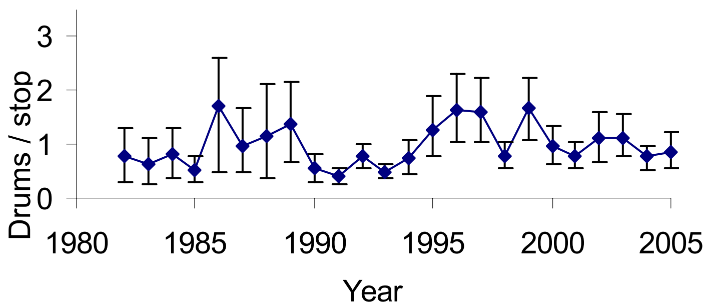
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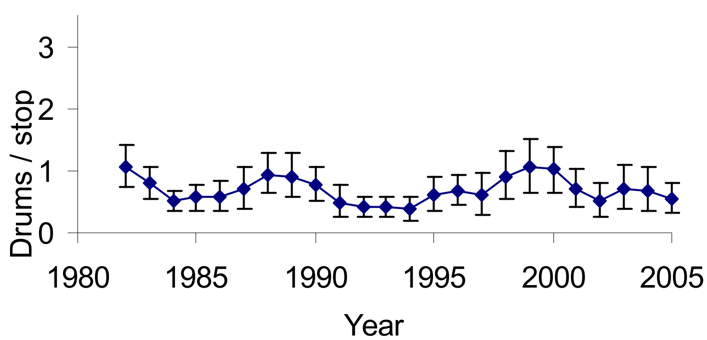
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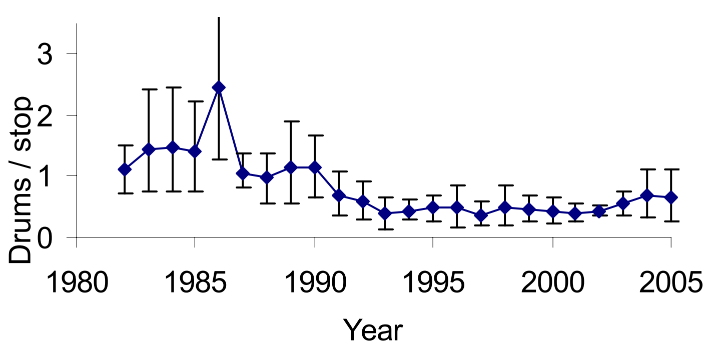
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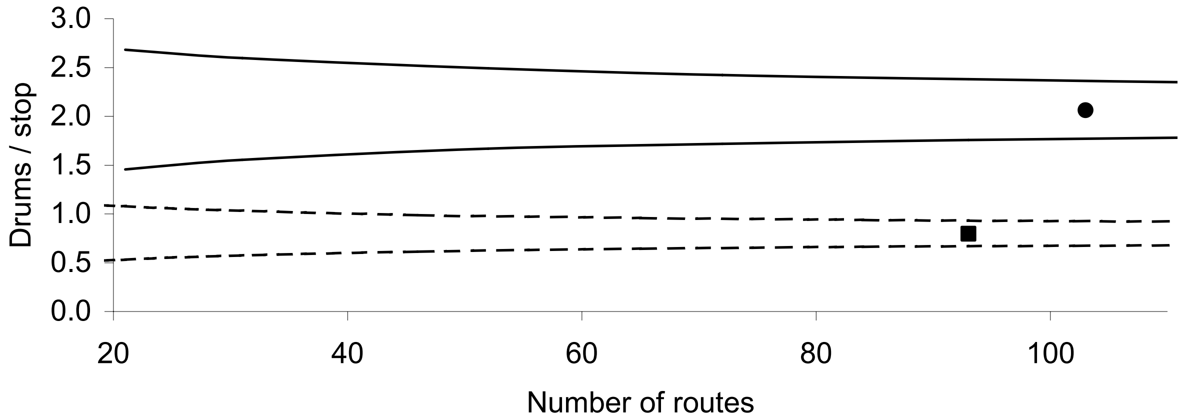

Figure 3. Ninety-five percent confidence intervals for bootstrapped samples of different numbers of routes in the Laurentian Mixed Forest province of Minnesota using ruffed grouse survey data from 1998 (solid lines) and 2004 (dashed lines), when mean drum counts were 2.06 (*n* = 103 routes, circle) and 0.80 (*n* = 93 routes, square), respectively.

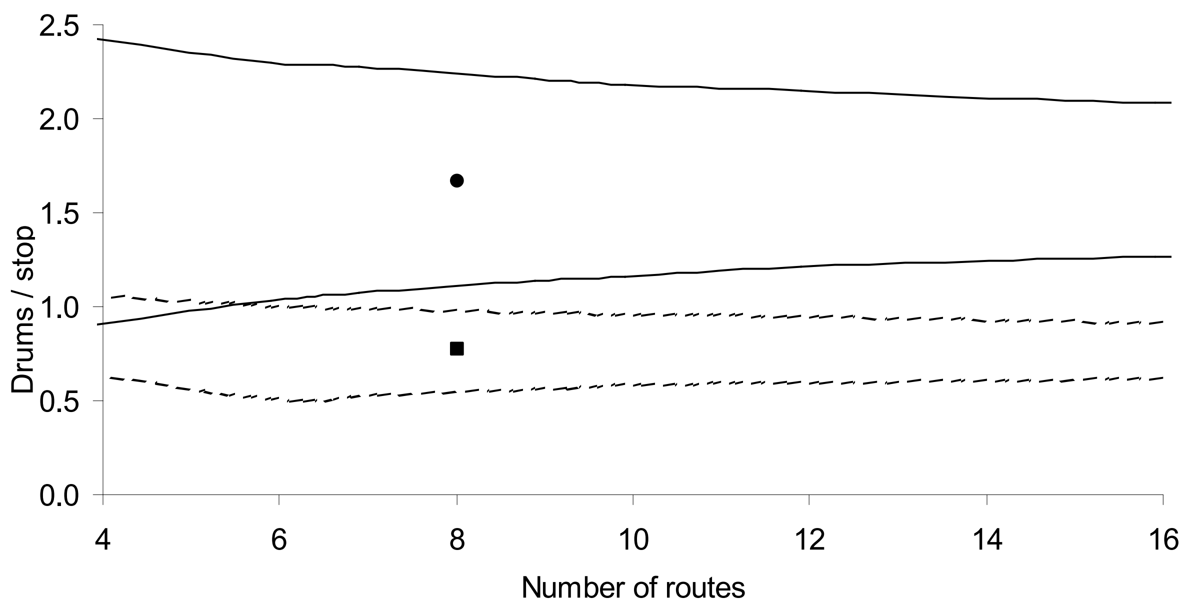

Figure 4. Ninety-five percent confidence intervals for bootstrapped samples of different numbers of routes in the Lake Agassiz and Aspen Parklands section of Minnesota using ruffed grouse survey data from 1999 (solid lines) and 2004 (dashed lines), when mean drum counts were 1.68 (circle) and 0.78 (square, n = 8 routes during both years).

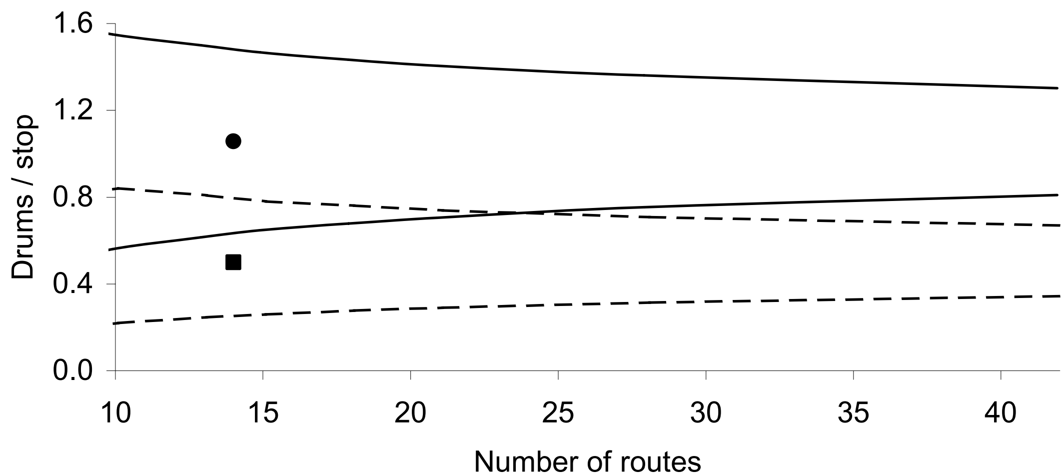

Figure 5. Ninety-five percent confidence intervals for bootstrapped samples of different numbers of routes in the Minnesota and Northeast Iowa Morainal section of Minnesota using ruffed grouse survey data from 1990 (solid lines) and 1993 (dashed lines), when mean drum counts were 1.06 (circle) and 0.50 (square, n = 14 routes during both years).

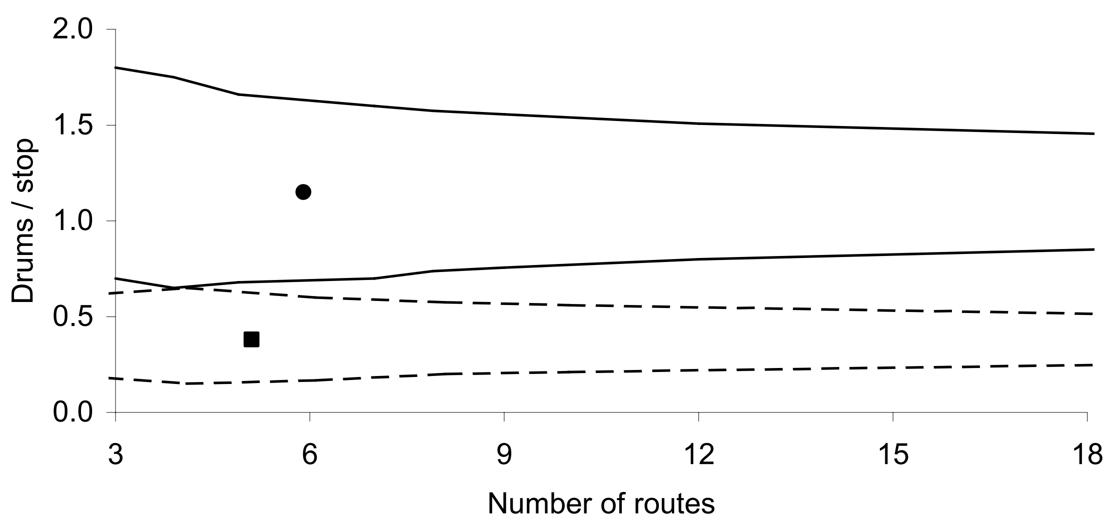

Figure 6. Ninety-five percent confidence intervals for bootstrapped samples of different numbers of routes in the Paleozoic Plateau section of Minnesota using ruffed grouse survey data from 1990 (solid lines) and 1993 (dashed lines), when mean drum counts were 1.15 (n = 6 routes, circle) and 0.38 (n = 5 routes, square).

SIMULATED EFFECTS OF FOREST MANAGEMENT ALTERNATIVES ON LANDSCAPE STRUCTURE AND HABITAT SUITABILITY IN THE MIDWESTERN UNITED STATES[1]

Stephen R. Shifley[2], Frank R. Thompson III[2], William D. Dijak[2], Michael A. Larson, and Joshua J. Millspaugh[3]

Abstract: Understanding the cumulative effects and resource trade-offs associated with forest management requires the ability to predict, analyze, and communicate information about how forest landscapes (1,000s to > 100,000 ha in extent) respond to silviculture and other disturbances. We applied a spatially-explicit landscape simulation model, LANDIS, and compared the outcomes of seven forest management alternatives including intensive and extensive even-aged and uneven-aged management, singly and in combination, as well as no harvest. We also simulated concomitant effects of wildfire and windthrow. We compared outcomes in terms of spatial patterns of forest vegetation by age/size class, edge density, core area, volume of coarse wood debris, timber harvest, standing crop, and tree species composition over a 200-year simulation horizon. We also used habitat suitability models to assess habitat quality for four species with diverse habitat requirements: ovenbird (*Seiurus aurocapilla*), prairie warbler (*Dendroica discolor*), hooded warbler (*Wilsonia citrina*), and gray squirrel (*Sciurus carolinensis*). Management alternatives with similar levels of disturbance had similar landscape composition but different landscape patterns. The no-harvest scenario resulted in a tree size-class distribution that was similar to scenarios that harvested 5% of the landscape per decade; this suggests that gap phase replacement of senescent trees in combination with wind and fire disturbance may produce a disturbance regime similar to that associated with a 200 year timber rotation. Greater harvest levels (10% per decade) resulted in more uniform structure of small or large patches, for uneven- or even-aged management, respectively, than lesser levels of harvest (5% or no harvest); apparently reducing the effects of natural disturbances. Consequently, the even-aged management at the 10% level had the greatest core area and least amount of edge. Habitat suitability was greater, on average, for species dependent on characteristics of mature forests (ovenbird, gray squirrel) than those dependent on disturbance (prairie warbler, hooded warbler) and habitat suitability for disturbance dependent species was more sensitive to the management alternatives. The approach was data-rich and provided opportunities to contrast the large-scale, long-term consequences for management practices from many different perspectives.

[1] Forest Ecology and Management. 2006. Volume 229:361-377
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SOFTWARE REVIEW: LANDSCAPE HSImodels SOFTWARE[1]

William D. Dijak[2], Chadwick D. Rittenhouse[3], Michael A. Larson, Frank R. Thompson III[2], and Joshua J. Millspaugh

Abstract: Habitat suitability index (HSI) models have been used to evaluate habitat quality for wildlife at the local scale. Rarely have such models incorporated spatial relationships of habitat components. We introduce Landscape HSImodels, a new Microsoft Windows program that incorporates typical HSI components as well as landscape evaluations of habitat for 21 species of wildlife. Spatial relationships of habitat include edge effects, patch area, distance to resource and habitat composition. A moving window approach evaluates habitat within an area typical of home ranges and territories. The software and sample data are available free of charge from the U.S. Forest Service, North Central Research Station at http://www.ncrs.fs.fed.us/hsi/.

[1] Manuscript tentatively accepted for publication in Wildlife Society Bulletin. 2007. Volume 35
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TRACKING THE RAPID PACE OF GIS-RELATED CAPABILITIES AND THEIR ACCESSIBILITY[1]

Barry A. Sampson and Glenn D. DelGiudice

Abstract: With the rapid expansion of geographic information system (GIS) technology and its integration into the wildlife biology field, it is becoming increasingly clear that having access to the full scope of its analytical tools will greatly improve our ability to study, understand, and manage wildlife populations. We use our long-term, white-tailed deer (*Odocoileus virginianus*) research project as a case study to highlight the significant advances in GIS that are benefiting investigations of wildlife. From initiation of our research, we included early GIS capabilities, and we attempted to utilize advances as they occurred. Herein, we document changes that occurred in 'wildlife GIS' over the last 15 years and how we applied them in our work. Since the 1972 launch of the first Landsat satellite, fitted with various scanners, the combined use of satellite imagery and GIS has become invaluable to landscape-level wildlife habitat work. Other remote sensing products, including digital orthophoto quads, digital raster graphics, Farm Service Agency leaf-on photos, and land-use land-cover data, interpreted with the expanding analytical capabilities of a GIS, have greatly increased the breadth, accuracy, and precision of such work. GIS technology is being used increasingly in conjunction with global positioning system (GPS) radiocollars, with fewer restrictions compared to the conventional very high frequency (VHF) telemetry systems, to study the movements, habitat use, vegetation impacts, and survival of large mammals. We identify numerous GIS tools and data that are currently available and discuss their potential value to wildlife researchers and managers. The Minnesota Department of Natural Resources (MNDNR) GIS staff has developed and expanded a large suite of easily accessible Arcview extensions that are available for free download from the MNDNR website (www.dnr.state.mn.us/mis/gis/tools/Arcview/extensions.html). We briefly describe a number of these that are particularly useful for wildlife research and management, including the Arcview EPPL7 extension, Arcview tools extension, Stream-mode digitizing extension, Garmin GPS extension, DNR random sample generator, and the DNR wildlife survey extension. We provide other website addresses that serve as sources for a large number of wildlife specific GIS tools and extensions, including spatial and theme conversion; animal movements; home range analysis; and GPS waypoint uploading, downloading, and map editing.

[1]Abstract of paper in press in the Wildlife Society Bulletin. 2006. Volume 34 .

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BLACK TERN NEST HABITAT SELECTION AND FACTORS AFFECTING NEST SUCCESS IN NORTHWESTERN MINNESOTA[1]

Stephen J. Maxson, John R. Fieberg, and Michael R. Riggs[2]

Abstract: We documented nest habitat selection, nests success, and factors affecting nest success of Black Terns (*Chlidonias niger*) at Agassiz National Wildlife Refuge in northwestern Minnesota. Over three years, 289 Black Tern nests and 400 random sites were sampled on search areas totaling 1,331 ha. Four habitat characteristics were measured at each nest and random site: (1) mean water depth, (2) distance to open water, (3) dominant vegetation within a 2-m radius, and (4) amount of open water within a 2-m radius. Habitat variables were highly correlated with each other, making it difficult to estimate independent effects of each habitat variable on nest-site selection. However, conditional logistic regression models indicated that locations closer to open water and in deeper water were more likely to be associated with nest sites. Locations in bulrush and sedge/grass were also more likely than those in cattails to be associated with nest sites, although 68% of nests were in cattail reflecting the greater availability of that habitat in the study area. Nest success ranged from 48-69% (Apparent) and 33-62% (Mayfield) among years. Except for five nests that were abandoned or had infertile eggs, nests that failed to hatch appeared to have been depredated. Nest success was higher for nests with larger clutch sizes, nests located farther away from other nests, and for nests initiated earlier in the nesting season. Nests with 3-egg clutches were 2.8 times as likely to hatch as 2-egg nests. The odds of a nest being successful increased by 25% for each 5 m increase in distance to the nearest nest and decreased 7% for each additional day that passed before the nest was initiated. Nest success was not related to nest cluster size and was negatively related to the strength of nest site-selection (estimated from logistic regression models).

[1] Abstract of paper submitted to Waterbirds
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INFLUENCE OF LAND USE ON MALLARD NEST STRUCTURE OCCUPANCY[1]

Michael C. Zicus, David P. Rave, Abhik Das[2], Michael R. Riggs[3], and Michelle L. Buitenwerf

Abstract: We investigated the relationship between land use and mallard (*Anas platyrhynchos*) occupancy of single- and double-cylinder nest structures on a 658 km^2 (254 mi^2) western Minnesota study area from 1997-1999. We used hierarchical logistic regression to spatio-temporally model structure occupancy as a function of land use, number of nearby structures, number of mallard pairs with access to the structure, size of the open-water area including the structure, and structure type. We fit models to data from 4 different sized buffers around each structure to investigate scale influences. Goodness of fit, predictive ability, and amount of reduced spatio-temporal correlation were similar for each buffer-size model. We made inferences using the 1.6 km radius buffer model because it produced the lowest deviance. The amount and attractiveness of nesting cover (i.e., as indexed by VOMs) within a buffer interacted with nest initiation period (P = 0.003). VOMs and nest occupancy were positively associated early in the nesting season, but the pattern reversed later in the nesting season. Structure occupancy and area of open water around a structure were related quadratically (P = 0.004), with odds of a structure in median sized open-water areas being occupied increasing until the open-water area was ~16 ha. Year and nesting season period interacted (P = 0.002), reflecting different nest initiation phenology. Number of pairs with access to a structure had no effect on nest initiations (P = 0.7), perhaps due to our inability to account for within-season changes in pair numbers. Number of nearby structures (P = 0.8) was unrelated to initiation probability, but structure density was low (0.05/km^2). We suspect that mallard settling patterns and an unmeasured temporal relationship between VOMs and numbers of pairs with access to structures produced the VOM by period interaction. Structures deployed in larger open water areas where surrounding residual upland cover is abundant can improve mallard nest success early in the nesting season when duckling survival is the greatest and can reduce hen mortality associated with nest destruction and re-nesting.

[1] Abstract of paper in press in the Journal of Wildlife Management. September 2006; 70(5).
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USING GIS TO PREDICT MALLARD NEST STRUCTURE OCCUPANCY

John R. Fieberg, Michael C. Zicus, and Dan Hertel[1]

SUMMARY OF FINDINGS

We used the relationships described in a study of mallard nest structures to build a Geographic Information System (GIS) based model that would predict the probability of structure use by mallards. We assessed the model performance using data from a long-term study and used the assessment to illustrate a useful approach to predictive model building and validation. The model employed an existing GIS developed to aid in waterfowl management in western Minnesota. We used 3 predictors: 1) nest structure type, 2) 4 measures of the size of open water area containing the structure, and 3) a measure that described the mean aggregate visual obstruction of all residual cover during the early part of the nesting season (15 March – 20 April) in a buffer with a 1.6 km radius around each structure. We built the predictive model using the approach outlined by Harrell (2001), which is an alternative to data-based model selection methods (e.g., stepwise variable selection). We used a bootstrap procedure to obtain an unbiased measure of future predictive performance of the models that we fit. Unfortunately, we failed to produce a GIS model with much predictive power. Constantly changing features in the landscape were likely responsible for the difficulty in predicting biological outcomes. The process we employed forced us to think about the problem rather than using a data-based selection algorithm to determine the most important variables in the model.

INTRODUCTION

Knowing which type of nest structure to use and where to deploy them in a landscape should be important to waterfowl managers. Zicus et al. (2006a) studied mallard (*Anas platyrhynchos*) nest structure occupancy in an attempt to understand how landscape features affected structure use. They were interested in the effect of 5 covariates, and their final fitted model was complex, including 3 interactions and 1 main effect. More nests were initiated as the size of the open water area where structures were deployed increased. Simultaneously, cover influence interacted with period of the nesting season such that nesting probability was positively associated with cover height and density early in the season, and negatively associated with cover height and density late in the season.

Nest success in structures is generally good (Eskowich et al. 1998) with early nests having higher nest success (M. Zicus, Minnesota Department of Natural Resources, unpublished data). Consequently, hen mortality associated with renesting (Sargeant et al. 1984) would be reduced for hens nesting in structures early in the year. Further, brood and duckling survival from early-hatched nests is believed to be greater than that of later-hatched nests (e.g., Rotella and Ratti 1992, Dzus and Clark 1998, Krapu et al. 2000). These understandings led Zicus et al. (2006a) to recommend that nest structures be deployed in larger wetlands where early-season residual cover in the surrounding uplands was most abundant within 1 km of the structure. Geographic Information System (GIS) models might provide powerful tools to help waterfowl managers decide where nest structure should be placed in complex landscapes.

OBJECTIVES

- Build a GIS-based model that wildlife managers can use to help determine best placement of mallard nest structures;

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- assess the model performance using data from a long-term study; and
- as a secondary objective, illustrate a useful approach to predictive model building and validation.

METHODS

We used the relationships described in a study of mallard nest structures (Zicus et al. 2006a) to build a GIS-based model that would predict the probability of structure use by mallards. The response that we were interested in modeling was the mean number of mallard ducklings (DUCKS) produced in each structure (Zicus et al. 2006b). We used 3 predictors: 1) nest structure type (TYPE), 2) 4 measures of the size of open water area containing the structure (NWI, GAP, FSA03, FSA97), and 3) a measure that described the mean aggregate visual obstruction (MVOM) of all residual cover during the early part of the nesting season in a buffer with a 1.6 km radius around each structure.

DATA USED TO BUILD THE MODEL

We began with a GIS developed to aid in waterfowl management in western Minnesota (D. Hertel, unpublished data). Classified Landsat Thematic Mapper data from 2000 and 2001 was used to estimate the area of each habitat class within buffers (1.6 km radius) around each nest structure.

The following variables were included in the model:

DUCKS. – We determined the mean number of ducklings from 110 nest structures across the entire nesting season from 1996 – 2003 (M. Zicus, unpublished data).

TYPE. – We considered 2 types of cylindrical nest structures, those having either a single or a double cylinder (Zicus et al. 2006a).

Open water area measures. – Different measures of the size of the open water area containing the structure were determined to compare model performance with different data sources. These measures were from: 1) open water polygons in National Wetland Inventory data (i.e., NWI; D. Hertel, unpublished data), 2) areas classified as open water in MN-GAP land cover data (i.e., GAP; Minnesota Department of Natural Resources 2004, U. S. Geological Survey 1989), 3) open water areas digitized from 2003 Farm Services Agency (FSA) aerial photography (i.e., FSA03; M. Zicus, unpublished data), and 4) open water areas digitized 1997 FSA aerial photography (i.e., FSA97; Zicus et al 2006a). The distribution of the NWI water data was highly skewed. As a result, we expected a few data points with extreme values (e.g., >100 ha) to have substantial influence on the model fit. Therefore, we also considered log(NWI + 0.1) which had a more bell-shaped distribution. Both NWI and GAP data are readily available for large areas of western Minnesota, whereas FSA97 and FSA03 data were included here to determine the potential gain in predictive power that might be obtained if efforts were made to obtain more up-to-date measures of open water.

MVOM. – We created a variable for the mean aggregate visual obstruction measurement (MVOM) for 15 March – 20 April for each buffer around each structure (D. Hertel, unpublished data). First, each 28 m x 28 m GIS cell within a particular habitat class in the buffer was assigned a habitat-specific VOM (Table 1). Next, a weighted VOM was calculated for each cell in a particular habitat class by multiplying the area of that habitat class in the buffer by the habitat-specific VOM. A mean aggregate visual obstruction measurement (MVOM) was then calculated for all cells in the buffer by summing the weighted VOMs across all habitat classes in the buffer and dividing by the total area of the buffer.

MODELING

We built predictive models using the approach outlined by Harrell (2001). We first determined a reasonable degree of model complexity using guidelines based on our sample size. This approach can be summarized as "determine the number of degrees of freedom (df) that can be spent, and then spend them without any further model simplification." Harrell suggested a minimum of 10 – 20 observations per parameter considered, including those that account for potential non-linear effects. Burnham and Anderson (1998) suggested a similar liberal rule of 10 observations per predictor. Consequently, we believed 5-10 parameters to be a maximum for the 110 structures that we observed.

We used Spearman's ρ^2 (i.e., between response and predictors) to help determine how to apportion the df among the available predictors (e.g., to account for potential non-linearities) (Harrell 2001). Spearman's ρ^2 is a generalization of the rank correlation between two variables that can account for nonmonotonic relationships (e.g., using quadratic ranks) (Harrell 2001:127). We included all variables for which we examined ρ^2 in the model (i.e., ρ^2 was used only to determine the degree of non-linearity in the model). These steps defined an *a priori* full model from which we made our inferences; thereby avoiding problems associated with model selection algorithms (e.g., over fit models that predict new data poorly and biased p-values and confidence intervals arising from models selected using data-based selection procedures).

We used a bootstrap procedure to obtain an unbiased measure of future predictive performance of the models that we fit (Harrell 2001). We fit the model to 1,000 bootstrapped data sets, and the fitted parameters were used to calculate predicted values for all observations in the original dataset (as well as the bootstrap data set). We then calculated two R^2 values for each bootstrap replication: 1) using the original data and predicted values from the bootstrap model fit, and 2) using the bootstrap data and the predicted values from the bootstrap model fit. The difference between these two values is an estimate of "optimism" (i.e., resulting from fitting and "testing" the model on the same dataset). A final adjusted R^2 value was then determined by subtracting the mean "optimism" from the R^2 obtained from the original fit of the model to the full dataset. Bootstrap calculations were carried out using functions in the Design library of the R computing package (Harrell 2001, R Core Development Team 2005). We also calculated the usual adjusted R^2.

RESULTS

MODEL COMPLEXITY

Values of Spearman's ρ^2 indicated that both TYPE and MVOMs had less potential for explaining variation in DUCKS than open water area (Figure 1). Consequently, we assumed the MVOM effect was linear (i.e., a single df was used to model the relationship between MVOMs and DUCKS). The relatively greater values of Spearman's ρ^2 for open water area and previous work (Zicus et al. 2006a) suggested that more dfs should be spent to model the effect of open water area. Values of Spearman's ρ^2 were considerably higher for the digitized water measures (FSA03 and FSA97) than either NWI or GAP measures of open water.

Two models were fit using digitized water data (FSA03 and FSA97):

DUCKS = TYPE + MVOM + water (using a linear spline with 2 df), and (1)

DUCKS = TYPE + MVOM + water (using a restricted cubic spline with 2 df) (2).

Model (1) used a single knot (i.e., the location where the slope was assumed to change), while model (2) used 3 knots (2 of these were located at the boundary of the data; the fit of a restricted cubic spline is constrained to be linear outside the range of the boundary knots). The medians of non-zero observations (3.66 and 3.14 for FSA03 and the FSA97 data, respectively) were chosen as the knot location for the linear spline. Knots for the cubic spline used the 10^{th}, 50^{th}, and 90^{th} percentiles of the data.

The GAP data only had 6 observations that were >0 and were not considered further. Given the low values of Spearman's ρ^2 for the NWI water data, we considered a model that assumed the effect of open water area was linear. In addition, we examined a model with a 2 dfs restricted cubic spline with knot locations again determined using the 10^{th}, 50^{th}, and 90^{th} percentiles of the data.

ESTIMATES OF PREDICTIVE POWER

Models that used FSA03 and FSA97 water data performed considerably better than models using the NWI or GAP water data (Table 2). However, none of the models performed particularly well. The model using the FSA97 data had an R^2 of 0.14, suggesting that the open water area measured in Zicus et al. (2006a) along with structure type and MVOM values explained 14% of the variation in mean duckling production per structure. However, bootstrap validation suggested this model would perform considerably worse when applied to new data (i.e., it would explain only 6% of the variation). By comparison, R^2 measures for models using the NWI data were all less than 5% and their adjusted measures were negative, suggesting that the grand mean might predict new data better than the fitted model.

TYPE and MVOM values had p-values considerably >0.05 in all of the models, suggesting that they were not associated DUCKS (see also exploratory plots with smoothing lines; Figure 2). These results suggest that the MVOM values are not likely to be useful for predicting the mean duckling production (across all periods and years) in nesting structures, and that the available measures of open water area (NWI and GAP) are of questionable value for modeling duckling production.

DISCUSSION

Models having strong predictive ability are often difficult to construct (Steyerberg et al. 2001, Ambler et al. 2002, Steyerberg et al. 2003). There are a number of reasons why our efforts may have failed to produce a GIS model with much predictive power. First, mean visual obstruction measurements (MVOM) within 1 km of each structure may not accurately reflect the importance of surrounding cover. In particular, the height and density of cover in individual buffers having the same land use could actually differ markedly. Second, while Zicus et al. (2006a) recommended making structure placement decisions using early spring landscape conditions (as described by aggregate MVOMs in the buffer), their recommendations were intended to encourage production of young early in the season and not necessarily the maximum production of young across the entire nesting season. Zicus et al. (2006a) found that occupancy rates increased with VOM measurements early in the nesting season and decreased with VOMs later in the nesting season. Given the time-varying effect of VOM on occupancy rates, it was not surprising to discover that MVOM was unrelated to season-long duckling production. Lastly, although cover and water body size both vary temporally, we were forced to use measurements of these variables from a single year. The relationship between these habitat measurements and the average productivity of structures (across the 8 years of the study) may be much weaker than the relationship between habitat covariates and productivity in any given year.

The question as to how much predictive power a model would need to have in order to be useful is difficult to answer. Regardless, the models using either NWI or GAP measures of open water had essentially no predictive power, and a better measure of open water would be needed to produce a model with even low predictive ability. FSA97 open water values produced the model with the most predictive ability, but even this was low, perhaps because water conditions had changed significantly between 1997 and 2003. Identifying specific locations for management actions such as nest structures will be difficult when the desired biological outcomes are determined by features in the landscape that are constantly changing. A sensible strategy for structure placement and management would be to place structures in larger wetlands (>4.0 ha) where early-season residual cover in the surrounding uplands is most abundant (Zicus 2006a; Minnesota Department of Natural Resources. 2006. Using cylindrical nest structures to increase mallard nest success. Unpublished pamphlet.). This should reduce the number of structures that never get used as 19 of 20 structures that were not used during the 8-year study were deployed in open water areas <0.8 ha in size (M. Zicus, unpublished data). In addition, we recommend that managers continue to collect data on structure use as well as habitat measurements surrounding the structure (e.g., cover types, wetland size) so that we might refine our models in the future.

Despite the poor predictability of the models considered, we believe the general modeling approach is a useful alternative to data-based model selection methods (e.g., stepwise variable selection). Harrell (2001:56-57) provides 7 disadvantages of stepwise selection methods (repeated verbatim below):

1. It yields R^2 values that are biased high.
2. The ordinary *F* and χ^2 test statistics do not have the claimed distribution. Variable selection is based on methods (e.g., *F* tests for nested models) that were intended to be used to test only prespecified hypotheses.
3. The method yields standard errors of regression coefficient estimates that are biased low and confidence intervals for effects and predicted values that are falsely narrow.
4. It yields P-values that are too small (i.e., there are several multiple comparison problems) and that do not have the proper meaning, and the proper correction for them is a very difficult problem.
5. It provides regression coefficients that are biased high in absolute value and need shrinkage. Even if only a single predictor were being analyzed and one only reported the regression coefficient for that predictor if its association with *Y* were "statistically significant," the estimate of the regression coefficient $\hat{\beta}$ is biased (too large in absolute value). To put this in symbols for the case where we obtain a positive association ($\hat{\beta}$ >0), E($\hat{\beta}$ | $P < 0.05$, $\hat{\beta}$ > 0) > β.
6. Rather than solving problems caused by collinearity, variable selection is made arbitrary by collinearity.
7. It allows us to not think about the problem.

Wildlife biologists have become familiar with problems associated with stepwise selection methods due to the popular book by Burnham and Anderson (2002) on model averaging and multi-model inference. As a result, model averaging and multi-model inference using AIC weights (Burnham and Anderson 2002) have become exceedingly prevalent in the wildlife literature. Unfortunately, few alternatives to AIC model averaging have been presented in applied ecology/wildlife journals (Guthery et al. 2005), and therefore model averaging is applied routinely without critical thinking. We would argue that approaches that utilize a full model with candidate predictors chosen based on subject matter considerations will often provide a viable alternative to model averaging/multi-model inference. The former approach offers several advantages over the AIC-based model-averaging paradigm. For example, more time can be spent on diagnostics and model validation since a single model is considered rather than a suite of candidate models. In addition, if interest lies in estimation (rather than prediction), calculation of valid confidence intervals is straightforward (estimates of regression coefficients and σ^2 are not biased from considering multiple models or model reduction) (Harrell 2001, Ambler 2002).

The benefits of using a full model for inference are likely to be greatest when the effective sample size is >10 – 20 times the number of candidate predictors (Harrell 2001, Ambler 2002). For problems where the ratio of effective sample size to number of predictors is smaller, we recommend first trying to eliminate variables that do not have strong biological support (e.g., based on prior studies). This process is advantageous because it forces the researcher to think about the problem rather than using a data-based selection algorithm to determine the most important variables. In addition, it is generally beneficial to eliminate redundant variables, variables with lots of missing values, and variables that have very narrow distributions (Harrell 2001). If the number of remaining predictors is still >10 – 20 times the effective sample size, model averaging or other methods of shrinkage (e.g., penalized estimation or lasso) may offer improved predictions (Harrell 2001, Ambler 2002).

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Table 1. Land use cover types and source of visual obstruction measurements (VOM) used to estimate mean visual obstruction measurements (MVOMs) in the GIS model.

GIS model		Source data		
Cover type	VOM (dm)[a]	Cover type	VOM (dm)	Reference
Grassland	1.16	CRP grass	1.30	Zicus[d]
		WMA grass	1.02	Zicus
		WPA grass	0.86	Zicus
		Other grass	0.86	Zicus
Cropland	0.001	Cropland[b]	0.001	Mack 1991
Hayland	0.80	Hayland	0.80	Mack 1991
Right-of-way	0.75	Gravel township road	0.71	Zicus
		Gravel county road	0.40	Zicus
		Gravel CSAH[c]	0.40	Zicus
		Paved CSAH	0.65	Zicus
		State highway	0.41	Zicus
		Railroad	1.60	Zicus
Woodland	1.70	Woodland	1.70	Mack 1991
Odd areas	1.70	Odd areas	1.70	Mack 1991
Vegetated wetlands	0.67	Seasonal	1.00	Mack 1991
		Semi-permanent	2.00	Mack 1991
		Temporary	0.50	Mack 1991
		Permanent	1.00	Mack 1991
Open water/barren	0.00	Open water/barren	0.00	Mack 1991

[a]Visual obstruction measurement corresponding to residual conditions in early spring (15 March – 20 April). Values are weighted by the area of the various source types occurring in western Minnesota.

[b]Mack (1991) presents values for many types of cropland. The value for fall-plowed cropland was used.

[c]CASH = county state aid highway.

[d]VOM is the mean value for 1997-1999 based on unpublished data collected as part of Zicus et al. (2006a).

Table 2. Measures of future predictive accuracy of GIS models predicting average duckling production from 110 nest structures in Grant County Minnesota, 1997 – 2003.

	R^2		
Model[a]	Original	Adjusted (from linear regression)	Adjusted (bootstrap)
FSA03, lsp	0.087	0.052	0.009
FSA03, rcs	0.084	0.050	0.009
FSA97, lsp	0.138	0.105	0.061
FSA97, rcs	0.134	0.102	0.056
NWI, linear	0.024	-0.013	-0.042
NWI, rcs	0.042	0.006	-0.045
Log(NWI), linear	0.027	0.000	-0.036
Log(NWI), rcs	0.053	0.017	-0.031

[a]lsp = linear spline model with 1 knot (2 dfs); rcs = restricted cubic spline model with 2 knots (3 dfs).

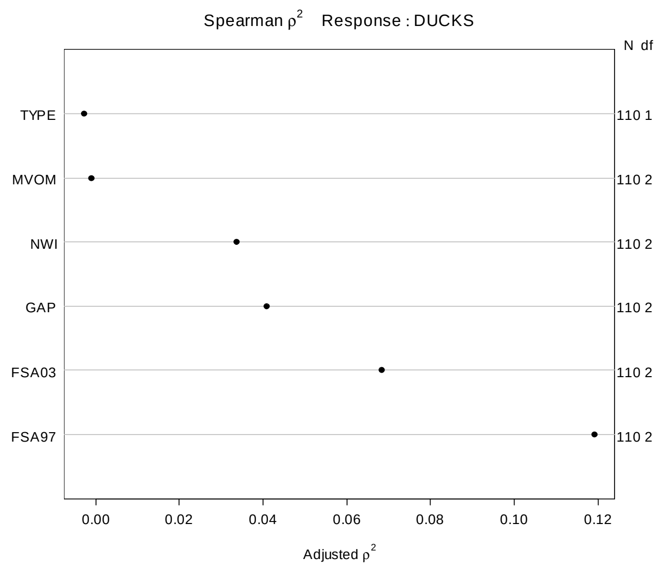

Figure 1. Spearman's ρ^2 indicating the strength of the relationship between mean ducking production (DUCKS) and each predictor variable (TYPE = indicator variable for structure type, MVOM measures, NWI open water measure, GAP open water measure, FSA03 open water measure, FSA97 open water measure).

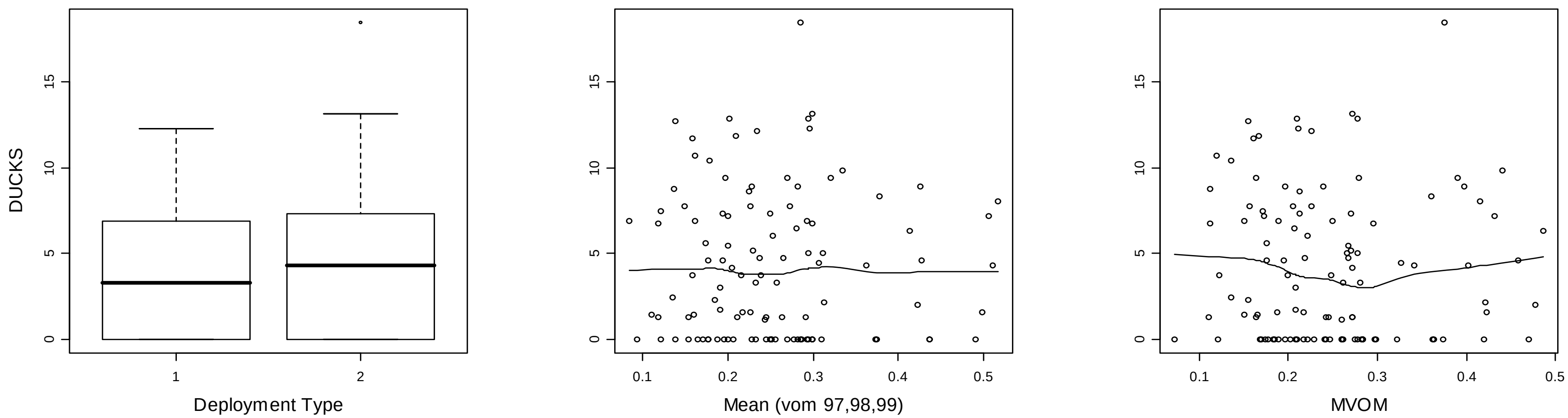

Figure 2. Exploratory plots of mean duckling production/year for each structure versus structure type, mean VOM measures across 1997-1999 (M. Zicus, unpublished data), and MVOM (D. Hertel, unpublished data). Lines represent smooth curves estimated using locally weighted regression via the lowess function in R.

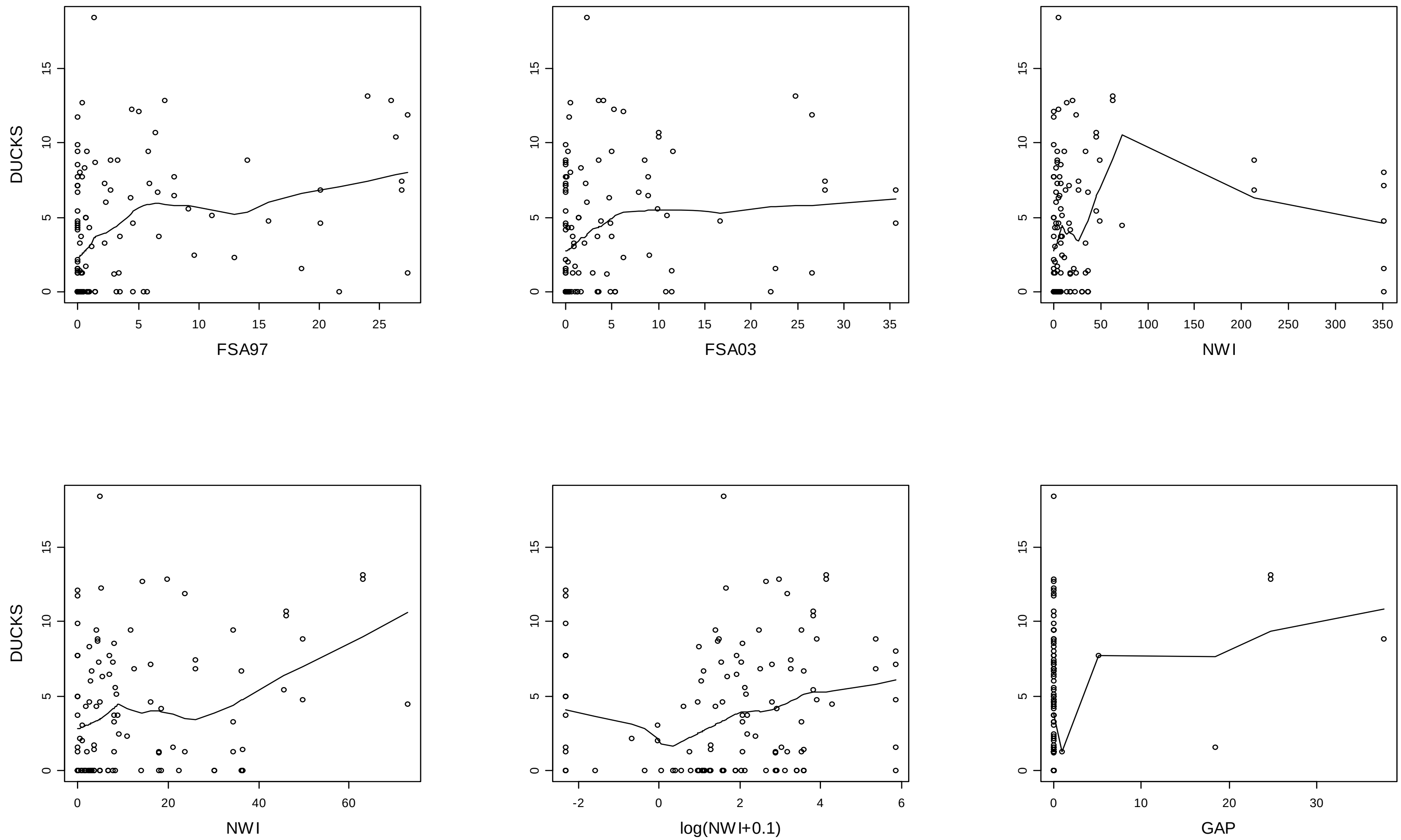

Figure 3. Exploratory plots of mean duckling production/year/structure versus FAS97 open water, FSA03 open water, NWI open water (all values), NWI open water (only values < 100), log(NWI + 0.1), and GAP open water. Lines represent smooth curves estimated using locally weighted regression via the lowess function in R.

EFFECTS OF SUBCUTANEOUS TRANSMITTER IMPLANTS ON MOURNING DOVES[1]

James B. Berdeen and David L. Otis[2]

Abstract: An important assumption of telemetry studies is that radiomarking does not negatively affect study animals. To test this assumption for mourning doves (*Zenaida macroura*), we evaluated whether subcutaneous transmitter implants (STI) would affect bird weight in cage studies and hunting mortality in field studies. At three weeks post-implantation, caged adult birds in the sham surgery and control groups gained and STI birds lost weight. Males gained and females lost weight. When percent weight change (PWC) for caged adult and juveniles was pooled the trends were similar, suggesting a STI treatment effect. In the field study, 16.3% of observed mortalities of STI birds during July–November 1998–2000 occurred during the first 3 days post-release. The overall 45-day summer period survival rate was relatively high, 0.9446 (95% CI = 0.8907–0.9986), when birds were entered into the population at-risk on the fourth day post-release. Although most observed mortalities were hunting-related (62.7%), similar direct recovery rates (P = 0.186) for STI (14.7%) and leg-banded birds (9.2%) suggests that implanted radios did not increase a bird's vulnerability to hunting mortality in the year of marking. However, the difference between the direct recovery rates of the 2 cohorts may be large enough to be biologically significant. Further research is needed to determine whether STI birds are especially susceptible to hunting mortality.

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COST SAVINGS FROM USING GIS-BASED "REAL TIME" IN A RING-NECKED DUCK SURVEY

John R. Fieberg, Robert G. Wright, and Michael C. Zicus

SUMMARY OF FINDINGS

Staff in the Minnesota Department of Natural Resources Wetland Wildlife Populations and Research Group recently began surveying Public Land Survey (PLS) sections with helicopters to estimate numbers of breeding ring-necked ducks (*Aythya collaris*). Data were recorded on paper tally sheets in 2004, while the 2005 survey utilized customized GPS and GIS software to record data directly to a tablet style computer. These customizations allowed the observers to display the aircraft's flight path over aerial photography or maps, and record both the flight path and animal observations directly to ArcView GIS shapefiles in real time. This provided an efficient means of data capture and may reduce the amount of flight time required to conduct the survey. We estimated that the approach saved ~1.75 minutes of flight time per sample plot using statistical methods developed for estimating causal effects in observational studies. As a result, survey cost was reduced by ~$2,100 when aircraft and staff expenses were considered.

INTRODUCTION

Geographic Information System (GIS) staff at the Minnesota Department of Natural Resources (MDNR) recently developed an ArcView GIS 3.x (Environmental Systems Research Institute, Redlands, California USA) extension called DNR Survey (MDNR 2005a), which provides menu-driven data entry forms for recording animals observed during aerial surveys. DNR Survey was designed to be used with DNR Garmin software (MDNR 2005a,b), a Global Positioning System (GPS) receiver, and a tablet style computer. This configuration allows the observer to view the aircraft's flight path over aerial photography, maps and survey boundaries, and record both the flight path and animal observations directly to ArcView GIS shapefiles, all in real time. This "real time" survey technique provides for efficient data capture and greatly enhances navigation between and within sample units. These efficiencies can reduce both the aircraft and staff costs associated with conducting aerial surveys. A ring-necked duck (*Aythya collaris*) breeding survey conducted in 2004 and 2005 is among several recent surveys conducted using the real time survey technique. In 2004, we surveyed 200 Public Land Survey (PLS) sections as survey plots, without the use of real time technology. The technology was employed in 2005 when 251 plots were surveyed.

Quantifying the amount of time saved by employing the real time survey technique can be difficult because other factors that may influence flight time are usually not held constant across years. For example, observers, number of ducks seen on a plot, and amount of potential nesting habitat in a plot differed between the 2 years in the ring-necked duck survey. These difficulties are common in observational studies, where covariates are not balanced between treatment and control groups. As a result, students in introductory statistics classes are often taught that observational studies can only provide evidence of correlation and not causation (Schield 1995).

We provide a brief introduction to statistical estimation of causal effects via counterfactuals. We then use data from the 2004 – 2005 ring-necked duck survey to illustrate the use of matching for estimation of a causal treatment effect in an observational study (i.e., where randomization to treatment group is not possible). Throughout, we will refer to observations in 2005 as "treated" and observations in 2004 as "controls", with the goal of estimating the causal effect (in terms of flight time savings) of using the real time survey technique.

OBJECTIVES

- To estimate the time and cost savings from employing the GIS-based real time survey technique in 2005, and
- to introduce a useful methodology for estimating causal effects in observational studies.

CAUSAL INFERENCE AND MATCHING COUNTERFACTUAL MODEL

Define two possible responses for each PLS survey plot:
$Y_i(t=1) = Y_i(1)$ = a continuous random variable representing the observation time for survey plot *i* flown under the "treatment condition" (i.e., using the real time technique in 2005)

$Y_i(t=0) = Y_i(0)$ = a continuous random variable representing the observation time for survey plot *i* flown under the "control" condition (i.e., no real time technique in 2004)

Similarly, we can define actual realizations of these random variables as $y_i(1)$ and $y_i(0)$ (typically, it is not possible to observe both random variables and therefore they are termed "counterfactuals" or potential outcomes). We then define a "realized causal effect" (Ho et al. 2005a) for sample plot *i* as: $y_i(1) - y_i(0)$. Realized causal effects are not observed and cannot be estimated because we record only $y_i(1)$ or $y_i(0)$, never both. Instead, we can attempt to estimate the expected causal effect for sample plot *i*, i.e., $E[Y_i(1) - Y_i(0)]$. Further, we can estimate the average treatment effect (ATE) overall and the average treatment effect for the treated (ATT) as:

$$ATE = \frac{1}{n}\sum_{i=1}^{n} E[Y_i(1) - Y_i(0)]$$ and

$$ATT = \frac{1}{n_T}\sum_{i=1}^{n_T} E[Y_i(1) - Y_i(0)],$$

where n_T = the number of treated observations and the second sum is only over treated subjects (Ho et al. 2005ab). The sampling frame for the ring-necked duck survey changed significantly between 2004 and 2005 (Zicus et al. 2005, 2006); therefore, these 2 effects may differ. The ATT provides a measure of the treatment effect that applies to a sampling frame similar to that used in 2005 (since it estimates causal effects only for treated observations), while the ATE provides an estimate of treatment effect that applies to the combined 2004 and 2005 sampling frames (since it estimates causal effects for control and treated observations).

$E[Y_i(0)]$ and $E[Y_i(1)]$ will usually depend on covariates (e.g., number of ducks observed on the survey plot, hectares of water/nesting cover in the survey plot) and are often estimated using regression models. In observational studies, the distribution of important covariates will often differ between treatment and control groups since the sampling units are not randomized to treatment group. This imbalance has important implications for model-based estimates of treatment effects as estimates will be biased if important confounders are not included in the model, or if the relationship between these confounders and the response is mis-specified. For example, a linear relationship might be assumed when the true relationship is non-linear (Ho et al. 2005a). Similarly, model-based estimates of treatment effects may depend heavily on the assumed model (e.g., estimates may be influenced by inclusion/exclusion of various covariates, assumptions regarding the degree of non-linearities and extent of interactions, distributional assumptions, etc.).

Matching control and treated observations with respect to potential confounders can help minimize the bias (and improve the robustness) of model-based estimates of expected causal effects (Ho et al. 2005a). Matching can be done in a number of ways, including exact matching (i.e., matching based on exact values of covariates) and nearest neighbor methods. Matching serves as a “preprocessing” step that pairs treated and control observations with respect to important covariates, resulting in a data set that is more balanced between these two groups (Ho et al. 2005ab). This balance helps provides assurance that any observed differences between control and treatment groups is due to the treatment rather than inherent differences between the two groups.

METHODS

We limited our analysis to observations that were made by 2 observers (DR and JH). Observer effects were large (Figure 1a) and DR and JH were the only pair of observers that flew plots in both 2004 and 2005. Before formulating and fitting regression models relating plot survey time to covariates (hectares of nesting cover, hectares of water, total ring-necks observed), we used functions in the R package MatchIt (Ho et al. 2005b) to create a dataset where each control observation (2004) was matched to a single treated observation (2005) using nearest neighbor matching with the distance between observations measured using a propensity score (Ho et al. 2005a). The propensity score measures the probability of an observation belonging to the treatment group as a function of covariates (hectares of nesting cover, hectares of water, total ring-necks observed) and is typically estimated using logistic regression. Treatment observations that were outside the convex hull of the control data (and vice versa) were discarded (Ho et al. 2005b, Stoll et al. 2005, King and Zeng in press,), leaving 113 controls matched to 113 treated observations. The convex hull is the smallest convex set containing all observations (in two dimensions, this is a polygon; e.g., “minimum convex polygon’s” are often considered in animal home range analyses). Observations outside the convex hull of the data are “far away” from the rest of the data and require extrapolation (rather than interpolation) to estimate their counterfactuals (King and Zeng in press).

The Models for Survey Time

We fit 4 models to the matched and original datasets. In each model, the response was the time required to fly each survey plot:

1. A least-squares regression model with linear effects of hectares of nesting cover, hectares of water, and total number of ring-neck ducks observed. In addition, an additive treatment effect was assumed.
2. A Poisson regression model that assumed the log (mean survey plot time) was linearly related to hectares of nesting cover, hectares of water, and total ring-necks observed. In addition, an additive treatment effect was assumed on the log scale.
3. A Poisson regression model that assumed the relationship between log(mean survey plot time) and hectares of nesting cover, hectares of water, total ring-necks observed were non-linear. We used orthogonal polynomials of degree 2 to account for the non-linearities. In addition, an additive treatment effect was assumed on the log scale.
4. A Poisson regression model that assumed the log(mean survey plot time) was linearly related to hectares of nesting cover and hectares of water. We used 2 degrees of freedom to model the effect of the number of observed

ducks. First, we included an indicator variable to reflect differences between plots that contained ducks and those plots that did not contain ducks. In addition, we included the number to reflect the assumption that the mean survey time increased of ducks observed as a covariate linearly (on the log scale) for each additional duck observed. Finally, an additive treatment effect was assumed on the log scale.

Letting *X* represent all covariates of interest, the expected survey time in each of the models is given by:

Model 1:

$$E[Y_i(1) \mid X] = \beta_0 + nest\_acres_i\beta_1 + water\_acres_i\beta_2 + ducks_i\beta_3 + \gamma$$

$$E[Y_i(0) \mid X] = \beta_0 + nest\_acres_i\beta_1 + water\_acres_i\beta_2 + ducks_i\beta_3$$

Model 2:

$$E[Y_i(1) \mid X] = \exp(\beta_0 + nest\_acres_i\beta_1 + water\_acres_i\beta_2 + ducks_i\beta_3 + \gamma)$$

$$E[Y_i(0) \mid X] = \exp(\beta_0 + nest\_acres_i\beta_1 + water\_acres_i\beta_2 + ducks_i\beta_3)$$

Model 3:

$$E[Y_i(1) \mid X] = \exp(\beta_0 + nest\_acres_i\beta_1 + nest\_acres_i^2\beta_2 + water\_acres_i\beta_3 + water\_acres_i^2\beta_4 + ducks_i\beta_5 + ducks_i^2\beta_6 + \gamma)$$

$$E[Y_i(0) \mid X] = \exp(\beta_0 + nest\_acres_i\beta_1 + nest\_acres_i^2\beta_2 + water\_acres_i\beta_3 + water\_acres_i^2\beta_4 + ducks_i\beta_5 + ducks_i^2\beta_6)$$

Model 4:

$$E[Y_i(1) \mid X] = \exp(\beta_0 + nest\_acres_i\beta_1 + water\_acres_i\beta_2 + I(ducks_i > 0)\beta_3 + ducks_i\beta_4 + \gamma)$$

$$E[Y_i(0) \mid X] = \exp(\beta_0 + nest\_acres_i\beta_1 + water\_acres_i\beta_2 + I(ducks_i > 0)\beta_3 + ducks_i\beta_4)$$

For model 1, the effect of treatment is estimated directly by γ (assuming the model is correct) since $\gamma = E[Y_i(1) - Y_i(0)]$ regardless of the value of *X*. For models 2-4, the effect of treatment is assumed to be multiplicative and therefore $E[Y_i(1) - Y_i(0)]$ will depend on *X*. In such cases, one may choose to estimate the causal effect of treatment for an observation with all covariates set to the mean values in the data, $E[Y_i(1) \mid X = \bar{x}] - E[Y_i(0) \mid X = \bar{x}]$. However, this causal effect may not be very meaningful [e.g., this "subject" may be very different from any of the subjects in the study, particularly for model 4 where one of the covariates is an indicator variable that is always either 0 or 1]. Thus, for models 2 – 4 we report an estimate of the multiplicative effect of treatment on survey time [i.e., exp(γ)].

For models 1, 2, and 4, we also estimated the ATE and ATT in the matched and full datasets (we did not estimate the ATE or ATT for model 3 because of minor complexities with applying the approach when using orthogonal polynomials and because models 2, 3, and 4 all gave similar estimates of γ). We followed the steps outlined in Ho et al. (2005b):

1. We fit the model (1, 2 or 4) first to the control observations (without γ in the model). We used the fitted model to estimate E[Y_i(0)] for all of the treated observations in the dataset.

2. We fit the model (1, 2 or 4) to the treated observations (again without γ in the model). We used the fitted model to estimate E[Y_i(1)] for all of the control observations in the dataset.

3. We estimated ATE using:

$$A\hat{T}E = \frac{1}{n}\left(\sum_{i=1}^{n_t}\left\{y_i(1) - E[Y_i \overset{\wedge}{(0)}]\right\} + \sum_{i=1}^{n_c}\left\{E[Y_i \overset{\wedge}{(1)}] - y_i(0)\right\}\right)$$

where "^" denotes estimated values determined using steps 1 and 2 and n_c = the number of "control" observations and n_t = the number of "treated" observations in the matched/full dataset.

4. We estimated the ATT using:

$$A\hat{T}T = \frac{1}{n_t}\sum_{i=1}^{n_t}\left\{y_i(1) - E[Y_i \overset{\wedge}{(0)}]\right\}.$$

Importantly, the estimation procedure fits separate models to the control and treatment observations (steps 1 and 2). These steps provide a means of essentially "imputing" values for $y_i(1)$ for control observations and $y_i(0)$ for treated observations. Using separate models in the two steps helps to reduce bias by eliminating the assumption of constant parameter values for treated and control observations (Ho et al. 2005b). Uncertainties in the estimates of ATE and ATT were determined by generating 1,000 random samples of all model parameters from their asymptotic sampling distributions (i.e., a multivariate normal distribution) using the R package, Zelig (Imai et al. 2005). For each set of sampled parameters, we estimated the ATE/ATT and then report the 0.025 and 0.975 percentiles across the set of 1,000 estimated ATEs/ATTs.

Lastly, we examined plots of flight time/plot versus date to determine if flight times decreased systematically as observers became more experienced with the survey. We also examined residuals plots to assess the fit of the regression models.

Cost Comparisons

Survey costs include airtime for the helicopter and the pilots, air and ground time for the observers, and lodging and meals for the pilots and observers. We determined the difference in the cost of the 2005 survey compared to the expected cost had we not used the real time approach. We determined the difference in airtime costs by multiplying the per-plot ATT by the number of plots surveyed in 2005 (251) and the helicopter/pilot rental rate ($230/hr). The difference in ground time was calculated by assuming 40-minute refueling stops for every 2.67 hrs of flight time (D. Rave, unpublished data). We determined observer cost difference by multiplying the air and ground time by an average observer salary (plus fringe) of $32/hour. Lodging and meal costs for the survey crew was assumed to be $150/day.

RESULTS

Matching significantly improved the balance between treated and control units with respect to important covariates (Figures 2, and 3). Model-based estimates of γ were quite consistent across models (2 – 4) using either the full or matched data sets (Table 1). Estimates of γ were all statistically significant (all $p < 0.05$). The linear model estimated that the 2005 survey technique would save on average >2 minutes/plot, while the Poisson regression models estimated approximately a 30% time reduction per plot (Table 1).

Conclusions regarding the importance of hectares of water, hectares of nesting cover, and number of observed ducks were also similar across the fitted models. Survey time was estimated to increase with hectares of water and number of ducks observed ($p < 0.05$), but hectares of nesting cover was not significantly related to survey time ($p > 0.05$) (Figure 1).

Average Treatment Effect (Overall and for the Treated)

Estimates of the ATE and ATT from the matched dataset were more conservative and also had slightly wider confidence intervals (reflecting the smaller sample size) than estimates from the full dataset (Table 2). While estimates ATE were similar to estimates of the ATT for the full dataset, estimates of the ATT were consistently lower than the corresponding estimates of the ATE for the matched dataset (average time savings ~1.75 minutes/plot compared to ~2 minutes/plot) (Table 2). Survey time did not appear to systematically decrease in either year (Figure 4).

Cost Comparisons

Use of the real time survey approach resulted in an estimated savings of >$2,100 over the expected cost of the 2005 survey if it had been conducted without using the real time approach (Table 3). Almost a full day was saved in airtime alone which resulted in further saving for on the ground refueling time and travel expenses for lodging and meals.

DISCUSSION

Estimating treatment effects from observational data can be problematic because treatment and control groups often differ with respect to important covariates that may also influence the response of interest. Regression models are frequently used to obtain adjusted estimates. Unfortunately, estimates of treatment effects will remain biased unless the relationships between confounders and the response are correctly specified in the model. In addition, model-based estimates of treatment effects may be highly sensitive to assumptions of the regression model (e.g., inclusion/exclusion of covariates, assumed non-linearities or interactions). Matching (treated and controlled units) with respect to important covariates can reduce the sensitivity of estimated treatment effects to model assumptions and also reduce bias (Ho et al. 2005a,b).

Matching significantly improved the balance between treated and control units with respect to factors thought to influence flight time. Therefore, we expected estimates of causal effects to be more consistent across models. Somewhat surprisingly, we found that estimates of γ, ATE, and ATT, while slightly more consistent for the matched data, were quite robust using either the full and matched datasets.

The ATT provided an estimate of the treatment effect for plots surveyed in 2005, while the ATE provided an estimate of the treatment effect for the combined 2004 and 2005 plots. On average, plots sampled in 2004 contained more ducks, even in the matched dataset (Figure 3). Since flight time increased with numbers of ducks, it was not surprising that estimates of the ATT were lower than estimates of the ATE for the matched data. On the other hand, estimates of ATE and ATT were more similar for the full dataset because it consisted mainly of 2005 survey plots (the matched data included 113 plots from each of the 2004 and 2005 surveys, while the full dataset included 251 plots from 2005 and only 130 from 2004).

In calculating cost savings from use of the real time survey technique, we used estimates of the ATT from the matched dataset. As a result of matching and dropping observations outside of the convex hull of the data, this estimate did not consider all plots sampled in 2005. Therefore, our estimate of cost savings may be biased. Using the most conservative estimate of the ATT from the full dataset (~2 minutes/plot) resulted in an estimate of cost savings of ~$2400. This estimate, while using all plots sampled in 2005, is likely to be more model-dependent as a result of imbalance with respect to important covariates between the 2004 and 2005 sample plots. Other methods exist for estimating causal effects in observational studies (Lunceford and Davidian 2004), and these may be explored in the future.

We cannot rule out the possibility that systematic differences among plots flown in 2004 and 2005 were partially responsible for observed reduction in flight times. However, our estimate of time savings was robust to assumptions regarding the effects of numbers of ducks, hectares of nesting cover, and hectares of water. Further, we controlled for observer differences by only considering observations made by the same 2 observers in both years. Lastly, survey time did not appear to systematically decrease in either year, and observers believed that it took only 1 or 2 plots to “get up to speed on things” (D. Rave, personal communication) suggesting experience with the survey was not responsible for the reduced flight times. While exact cost savings are impossible to determine, we believe that actual flight time was reduced by ~7 – 8 hours through the use of the real time approach. This amounts to ~10% of the total survey flight time (Zicus et al. 2006). Although we did not attempt to estimate them, further savings were realized because the survey data were recorded directly in ArcView shapefile. Consequently, data entry from field sheets and the related data checking were eliminated.

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Table 1. Estimates of the treatment effect (i.e., time savings from sampling plots with the real time survey technique in 2005). For model 1, the effect of treatment is assumed to be additive: γ = $E[Y_i(1) \mid X = x] - E[Y_i(0) \mid X = x]$. For models 2 – 4, the effect of treatment is assumed to be multiplicative, γ = $E[Y_i(1) \mid X = x] / E[Y_i(0) \mid X = x]$. For models 2 – 4, we determined 95% confidence intervals using $\exp\left[\hat{\gamma} \pm 1.96 \cdot se(\hat{\gamma})\right]$.

	Before matching		After matching	
Model	Estimate	95% C.I.	Estimate	95% C.I.
1	-2.19	(-1.54, -2.84)	-2.05	(-1.18, -2.92)
2	0.71	(0.65, 0.77)	0.67	(0.59, 0.76)
3	0.73	(0.67, 0.80)	0.71	(0.62, 0.82)
4	0.73	(0.67, 0.79)	0.71	(0.62, 0.81)

Table 2. Estimates (minutes/plot) of the average treatment effect (ATE) and average treatment effect for the treated (ATT) using the full and matched datasets in the 2004 and 2005 ring-necked duck breeding pair survey.

	Average treatment effect (ATE)				Average treatment effect for the treated (ATT)			
	Before matching		After matching		Before matching		After matching	
Model	Estimate	95% C.I.	Estimate	95% C.I.	Estimate	95% C.I.	Estimate	95% C.I.
1	-2.18	(-1.61, -2.71)	-1.98	(-1.26, -2.64)	-2.18	(-1.61, -2.71)	-1.74	(-1.14, -2.40)
2	-2.24	(-1.81, -2.70)	-2.00	(-1.40, -2.54)	-2.24	(-1.79, -2.73)	-1.83	(-1.31, -2.34)
4	-2.11	(-1.65, -2.55)	-1.90	(-1.29, -2.43)	-2.02	(-1.58, -2.49)	-1.74	(-1.20, -2.25)

Table 3. Approximate cost savings realized by using the real time survey approach in the 2005 ring-necked duck breeding pair survey. Calculations were based on a reduction of ~1.75 minutes of survey time per plot (average causal treatment effect) for all plots.

Expense	Crew member	Hours or days	Cost/hr or day ($)	Cost ($)
Air time (hrs)	Pilot	7.3[a]	230[b]	1,684
	Observer	7.3[a]	32[c]	234
Ground time (hrs)	Observer	1.84[d]	32[c]	59
Travel (days)	Pilot and observer	1.0	150	150

[a]Air time is equal to the per plot average causal treatment effect times 251 plots divided by 60 minutes.

[b]Air time rate includes helicopter cost and pilot salary and fringe.

[c]Average observer salary and fringe.

[d]Ground time is equal to the hours of air time divided by 2.67 (hours between refueling) times 0.67 (hours to refuel).

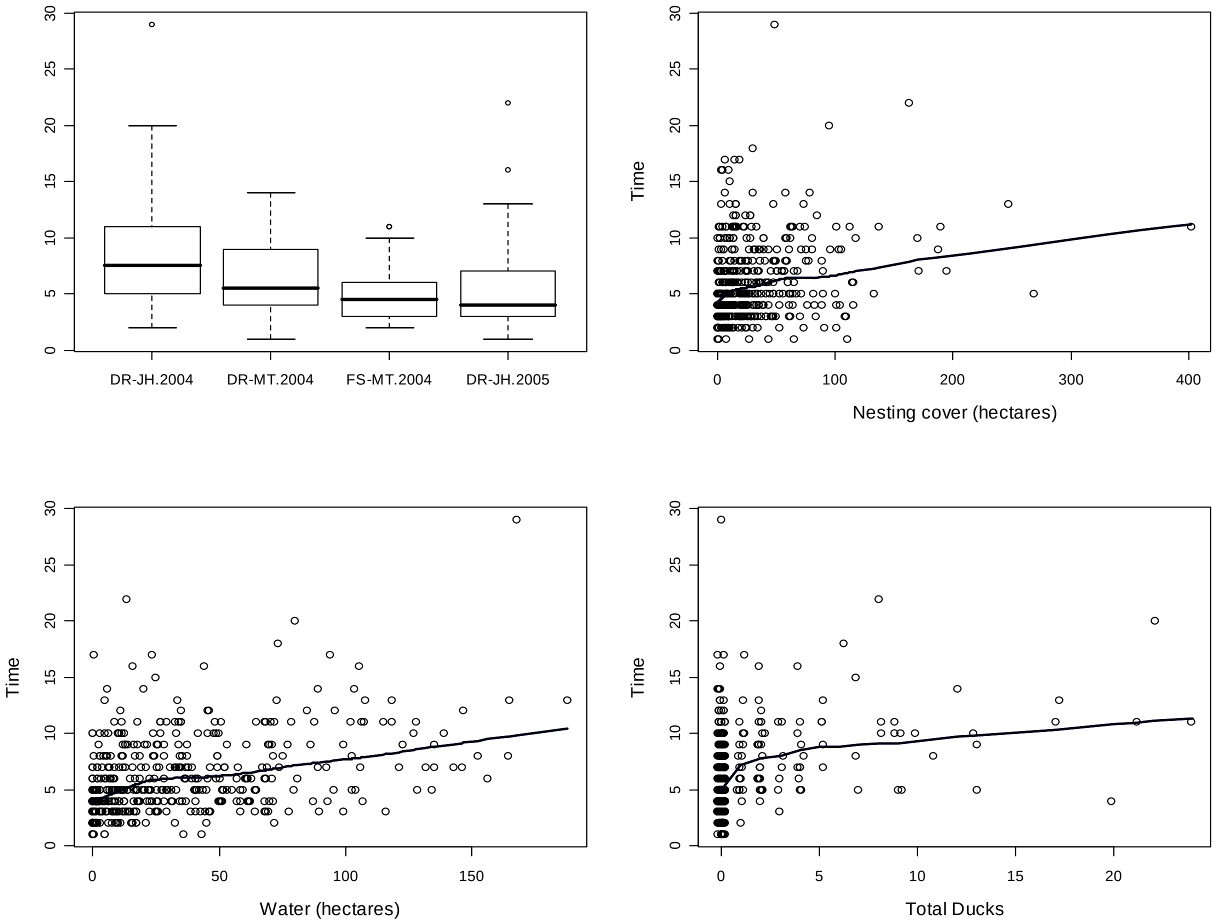

Figure 1. Time required to fly each plot versus covariates (observer/year, nest acres, water acres, ducks observed). Lines indicate loess smooths of the data using the lowess function in R (R Core Development Team 2005).

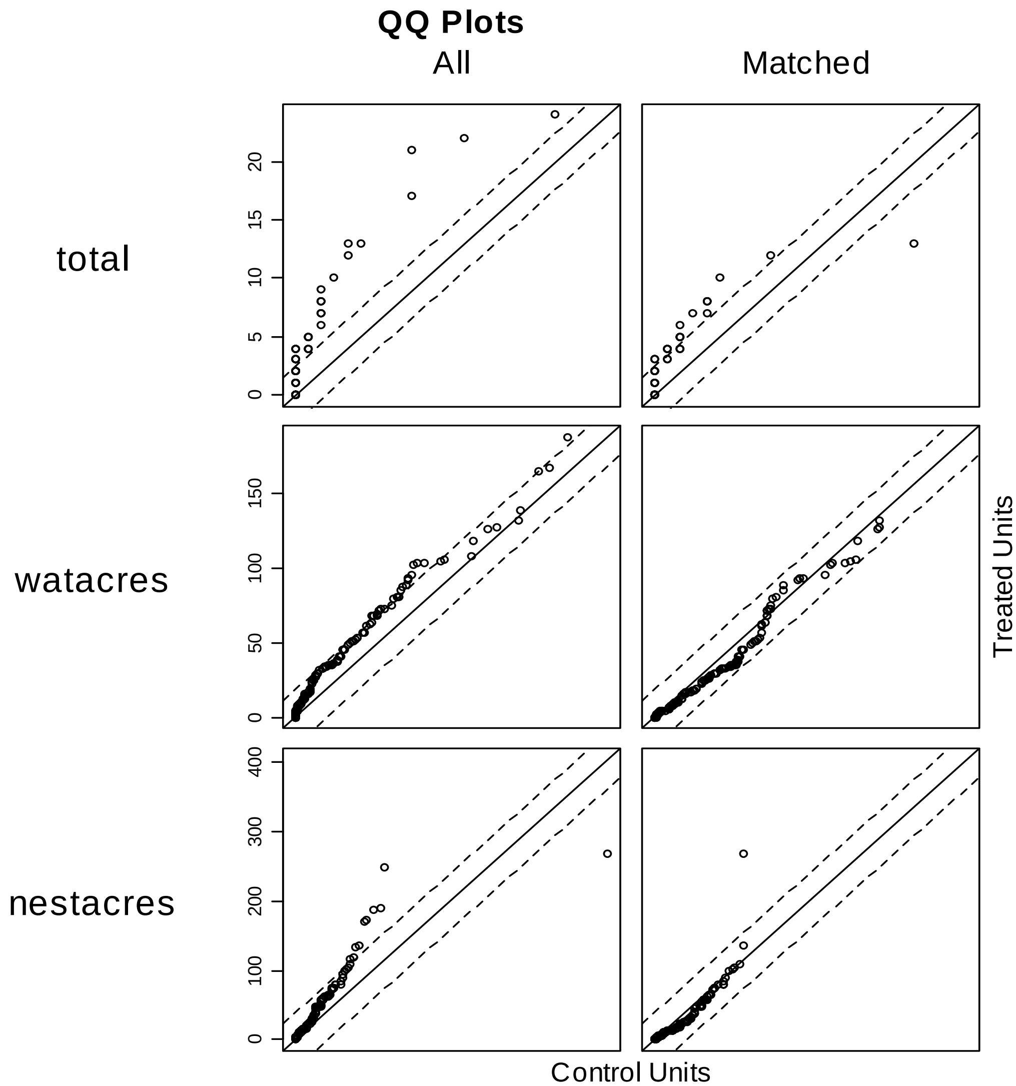

Figure 2. Quantile-quantile plots of the empirical distributions of each covariate before and after matching.

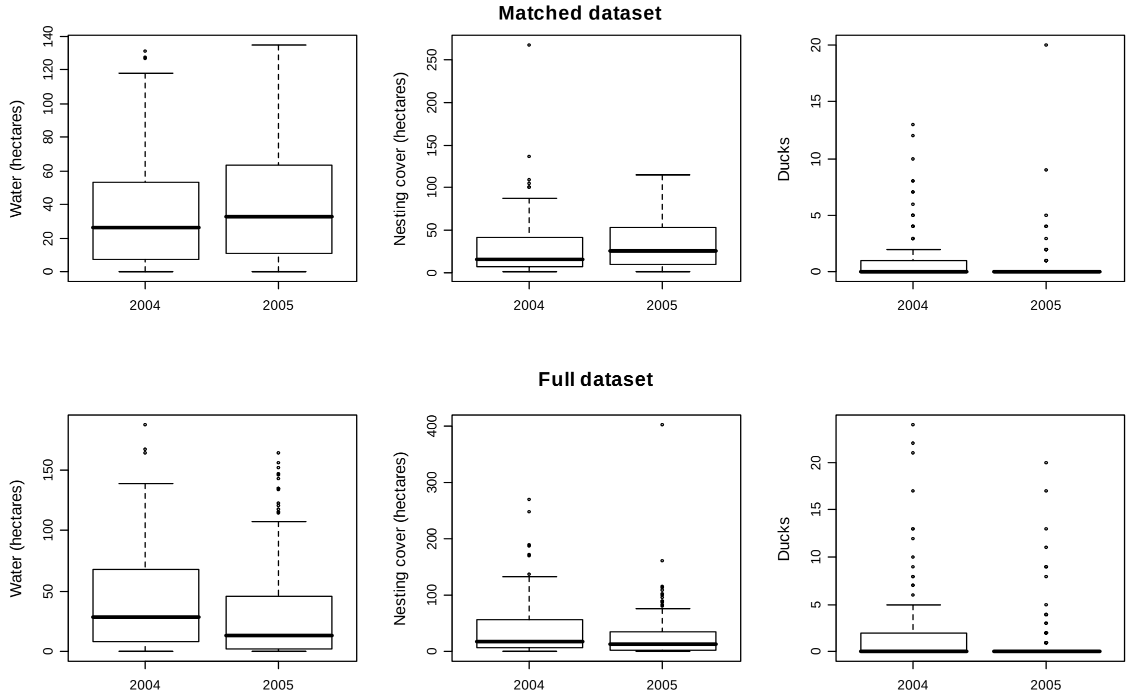

Figure 3. Boxplots of water (hectares), nesting cover (hectares), and ducks observed for the matched and full datasets.

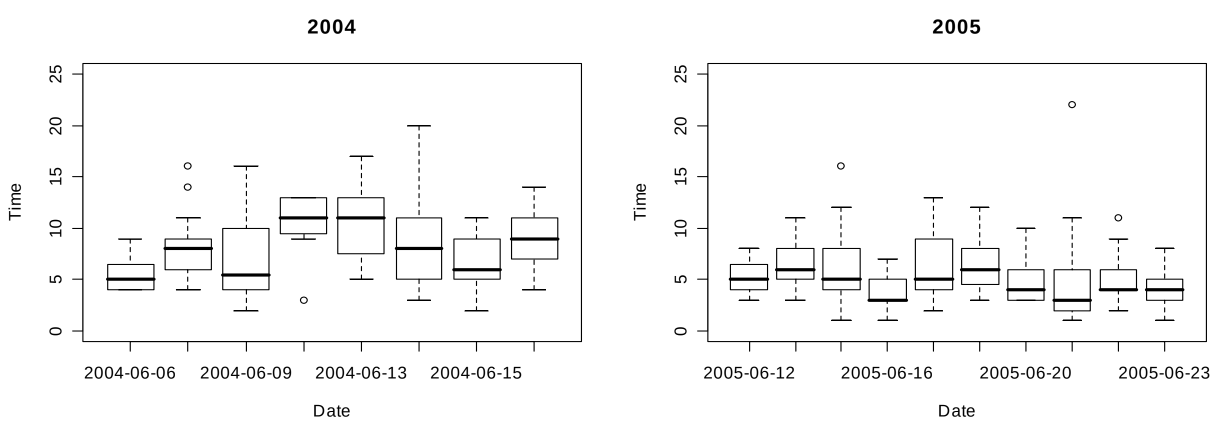

Figure 4. Boxplots of survey plot times versus date.

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 second year of a pilot survey to better understand the issues involved in monitoring these important but poorly studied ducks. The helicopter-based counts (12 – 23 June 2005) entailed 10 flight days and included a portion of Minnesota 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 section-sized survey plots, and 4 habitat classes were defined based on the amount of nesting cover in each plot. In contrast to 2004, we combined results from 2 separate surveys to estimate population size. We apportioned 230 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 21 plots to estimate population size in the remaining habitat. The combined population was estimated to be ~11,300 indicated breeding pairs (~25,000 birds). Numbers of ducks counted from the air and the ground on 14 lakes differed more in 2005 than in 2004, and the difference likely due to the 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, and 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 has been developing a forest wetlands and waterfowl initiative. The status of ring-necked ducks (*Aythya collaris*) 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 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 Public Land Survey (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 second 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 2006 and future operational surveys.

METHODS

Two separate surveys were conducted in 2005 to reduce the bias associated with the 2004 estimate. We continued to use a stratified random sampling design with 2 stratification variables: Ecological Classification System (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 modified the habitat class definitions used for stratification in 2004 (Table 1). Based on 2004 results, we also included 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, 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.

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 230 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 20 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:

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

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.

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 2005 survey utilized an ArcView 3.x extension (DNRSurvey) in conjunction with a GPS 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 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 (94) were located in the Northern Minnesota Drift and Lake Plains Section. However, the fewest plots (8) were located in the Lake Agassiz, Aspen Parklands section this year rather than the Northern Superior Uplands

Section (Table 2). The highest and lowest sampling rate again occurred in the Lake Agassiz, Aspen Parklands Section and Northern Superior Uplands section, respectively. A total of 21 rather than 20 habitat class 3 and 4 plots was surveyed because a replacement plot was flown before permission to survey one of the originally selected plots was granted. The survey was conducted 12 – 24 June and entailed 11 survey-crew days. Observed pairs represented 36% of the indicated pairs tallied during the survey compared to 57% in 2004 (Table 3).

Estimated Pair Density

Mean pair density on habitat class 1 and 2 plots ranged from a high of 3.40 pairs/plot in the Lake Agassiz, Aspen Parklands Section to a low of 0.09 pairs/plot in the Northern Minnesota and Ontario Peatlands Section (Table 4). Mean pair densities were lower in 2005 in 4 of the 6 ECS sections compared to 2004. 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 3,490 in the Northern Minnesota Drift and Lake Plains Section to a low of 239 in the Northern Minnesota and Ontario Peatlands Section (Table 5). Fewer breeding pairs were estimated in 2005 in 3 of the 6 ECS sections than in 2004. Considering both years, pair numbers were greatest in the Northern Minnesota Drift and Lake Plains Section and fewest in the Western and Southern Superior Uplands and the Northern Minnesota and Ontario Peatlands sections.

The estimated population of ring-necked ducks on habitat class 1 and 2 plots ranged from a high of 6,981 in the Northern Minnesota Drift and Lake Plains Section to a low of 477 in the Northern Minnesota and Ontario Peatlands Section (Table 6). As with indicated breeding pairs, fewer ducks were estimated in 2005 in 3 of the 6 ECS sections than in 2004. 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 and the Northern Minnesota and Ontario Peatlands sections.

In 2005, 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 11,329 (90% confidence interval = 5,359 – 17,298), and the estimated ring-necked duck population was 24,943 (90% confidence interval = 12,476 – 37,411).

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 shifted somewhat to the east compared to 2004 (Figure 1). Estimates from 2004 and 2005 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

There was a greater discrepancy between boat counts and the aerial counts of indicated breeding pairs for the individual lakes included in the *Bemidji Area Ring-necked Duck Pair Survey* in

2005 than in 2004 (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. Poorer agreement between the 2 surveys in 2005 than in 2004 was likely due to the greater time that elapsed between the boat counts and the aerial surveys.

STRATIFICATION EVALUATION

Analysis of variance indicated that the stratification used in the 2005 survey performed well. Indicated pairs were related significantly to ECS sections ($F_{5,218}$ = 7.17, P <0.001) and to habitat classes within the ECS sections ($F_{1,218}$ = 28.7, P <0.001). The importance of habitat class varied among ECS sections ($F_{5,218}$ = 7.94, P <0.001), although more mean indicated pairs were seen in habitat class 1 plots than in class 2 plots in 5 of 6 ECS sections. Pair density was greatest in the Lake Agassiz, Aspen Parkland habitat class 1 stratum plots. In contrast, no indicated pairs were observed in habitat-class 2 plots in the Northern Minnesota and Ontario Peatlands (n = 16) or Lake Agassiz, Aspen Parkland sections (n = 3). However, indicated pairs also were not observed in high-density plots in the Western and Southern Superior Uplands (n = 11). Our best estimate of relative efficiency of the stratified design compared to a simple random sample suggested we would have needed approximately 1.2 times as many plots to achieve the same precision under a simple random sampling design. However, we lacked variance estimates for 3 strata because no birds were observed on sample plots in those strata. Thus, standard error estimates and design effects should be interpreted cautiously.

Data Acquisition

Generally less time was required to survey a plot in 2005 than in 2004 (Table 8). Survey time ranged from 1 – 22 minutes (mean = 5.2) compared to 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 and 2005 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, but a second survey based on a simple random sample of plots in habitat classes 3 and 4 was needed to provide 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. Furthermore, emergent shoreline-vegetation associated with larger lakes containing fish was defined as potential ring-necked duck nesting cover when stratification decisions were based on MN-GAP data alone. Ring-necked ducks do not occupy these types of lakes during the breeding season. Stratification would likely be improved somewhat by not including emergent shoreline-vegetation associated with these larger lakes when quantifying potential nesting cover in each PLS section. Additional GIS data would be required to identify this cover.

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 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 zero 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 intend to begin modeling to evaluate various plot sizes in 2006.

Recommendations

- Conduct the 2006 survey using the same proportional allocation of 230 habitat class 1 and 2 plots among the 6 ECS sections. Conduct the 2006 survey choosing a simple random sample of 20 habitat class 3 and 4 plots. Rationale: 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 2004 and 2005 data alone suggest somewhat different geographical distributions for indicated breeding pairs, and a third year would help better define the core area.

- Begin the survey as soon after 5 June as possible. Rationale: 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. Rationale: 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. Rationale: 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.

ACKNOWLEDGMENTS

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

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Table 1. Habitat classes assigned to Public Land Survey section plots in the Minnesota ring-necked duck breeding pair survey area, June 2004-2005.

	Definition[a]		%[b]	
Habitat class	2004	2005	2004	2005
1	Plots with ≥ the median amount of MN-GAP class 14 and/or 15 cover within 250 m of and adjacent to open water (i.e., potentially high pairs).	Plots with ≥ the median amount of MN-GAP class 10, 14, and/or 15 cover within 250 m of and adjacent to open water (i.e., potentially high pairs).	15.3	24.5
2	Plots with < the median amount of MN-GAP class 14 and/or 15 cover within 250 m of and adjacent to open water (i.e., potentially moderate pairs).	Plots with < the median amount of MN-GAP class 10, 14, and/or 15 cover within 250 m of and adjacent to open water (i.e., potentially moderate pairs).	15.3	24.5
3	Plots with no MN-GAP class 14 and/or 15 cover that include open water that is within 250 m of a shoreline (i.e., potentially low pairs).	Plots with no MN-GAP class 10, 14, and/or 15 cover that include open water that is within 100 m of a shoreline (i.e., potentially low pairs).	25.2	7.7
4	Plots with no MN-GAP class 14 or 15 cover and no open water within 250 m of a shoreline (i.e., potentially no pairs).	Plots with no MN-GAP class 10, 14, and/or 15 cover and no open water within 100 m of a shoreline (i.e., potentially no pairs).	44.2	43.3

[a]Plots are Public Land Survey sections. MN-GAP cover class 10 is described as lowlands with <10% tree crown cover and >33% cover of low-growing deciduous woody plants such as alders and willows. MN-GAP cover class 14 is described as wetlands with <10% tree crown cover that is dominated by emergent herbaceous vegetation such as fine-leaf sedges. MN-GAP cover class 15 is described as 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

Table 2. Sampling rates in the habitat class 1 and 2 strata in the Minnesota ring-necked duck breeding pair survey area, June 2004-2005.

	~Area[a]		Sample plots		Sampling rate (%)	
Ecological Classification System section	2004	2005	2004	2005	2004	2005
W & S Superior Uplands[b]	1,638	2,461	18	22	1.1	0.9
Northern Superior Uplands	1,810	4,648	13	36	0.7	0.8
N Minnesota & Ontario Peatlands	1,817	2,737	26	35	1.4	1.3
N Minnesota Drift & Lake Plains	5,048	8,383	78	94	1.5	1.1
Minnesota & NE Iowa Morainal	3,510	4,033	50	35	1.4	0.9
Lake Agassiz, Aspen Parklands	316	363	15	8	4.7	2.2

[a]Number of Public Land Survey sections in habitat classes 1 and 2.
[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 2004-2005.

				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

[a]Survey conducted 6 – 17 June.

[b]Survey conducted 12 – 24 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 2004-2005.

	2004			2005		
Ecological Classification System section	Plots	Mean pairs/plot	SE	Plots	Mean pairs/plot	SE
W & S Superior Uplands[a]	18	0.167	0.122	22	0.181	0.179[b]
Northern Superior Uplands	13	0.566	0.396	36	0.252	0.118
N Minnesota & Ontario Peatlands	26	0.465	0.381[b]	35	0.087	0.045[b]
N Minnesota Drift & Lake Plains	78	0.707	0.155	94	0.416	0.138
Minnesota & NE Iowa Morainal	50	0.797	0.298	35	0.228	0.010
Lake Agassiz, Aspen Parklands	15	2.959	0.948	8	3.403	1.365[b]

[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 2004-2005.

	2004				2005			
Ecological Classification System section	Pairs	LCL[a]	UCL[a]	CV(%)	Pairs	LCL	UCL	CV(%)
W & S Superior Uplands[b]	273	0	626	74.1	444	0	1,207	99.5[c]
Northern Superior Uplands	1,025	0	2,311	69.9	1,169	244	2,095	46.8
N Minnesota & Ontario Peatlands	845	0	2,030	82.0[c]	239	20	457	54.1[c]
N Minnesota Drift & Lake Plains	3,567	2,278	4,856	21.7	3,490	1,577	5,404	33.0
Minnesota & NE Iowa Morainal	2,799	1,041	4,556	37.4	918	241	1,595	43.6
Lake Agassiz, Aspen Parklands	935	405	1,465	32.0	1,235	273	2,198	40.1[c]

[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 2004-2005.

	2004				2005			
Ecological Classification System section	Birds	LCL[a]	UCL[a]	CV(%)	Birds	LCL	UCL	CV(%)
W & S Superior Uplands[b]	546	0	1,252	74.1	889	0	2,415	99.5[c]
Northern Superior Uplands	2,049	0	4,622	69.9	2,339	488	4,190	46.8
N Minnesota & Ontario Peatlands	2,183	0	5,385	85.7[c]	477	40	915	54.1[c]
N Minnesota Drift & Lake Plains	7,849	5,015	10,682	21.7	6,981	3,154	10,808	33.0
Minnesota & NE Iowa Morainal	5,597	2,082	9,113	37.4	4,122	187	8,057	56.4
Lake Agassiz, Aspen Parklands	2,097	856	3,339	33.4	2,471	545	4,396	40.1[c]

[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, 2004-2005.

		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]

[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. Time required to complete the Minnesota ring-necked duck breeding pair survey, June 2004-2005.

			Time (min)[a]			
Year	# 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

[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.

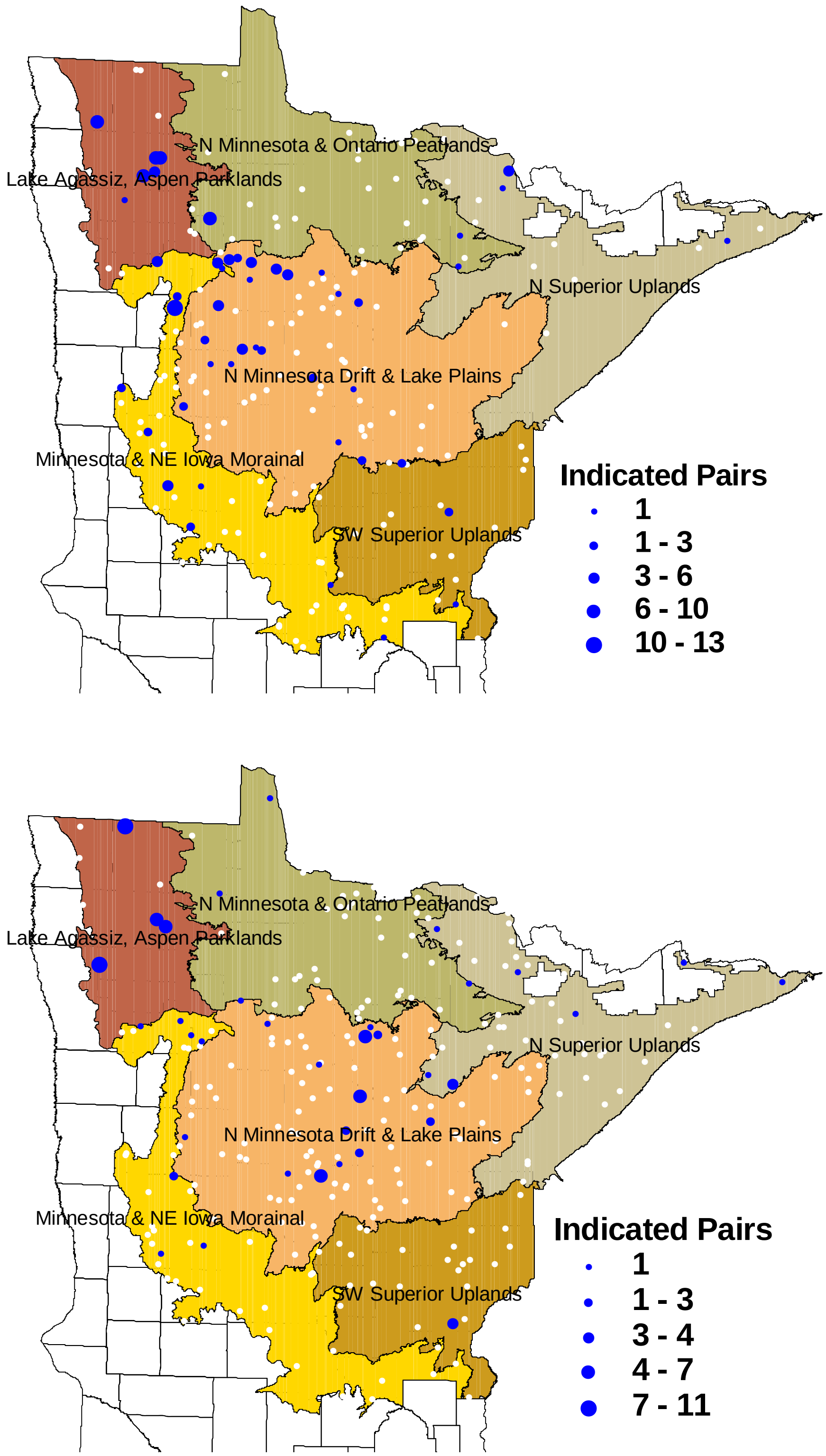

Figure 1. Plot locations and numbers of indicated breeding pairs of ring-necked ducks observed on survey plots in the Minnesota survey area in June 2004 (top) and 2005 (bottom). White circles indicate plots where no indicated pairs were seen.

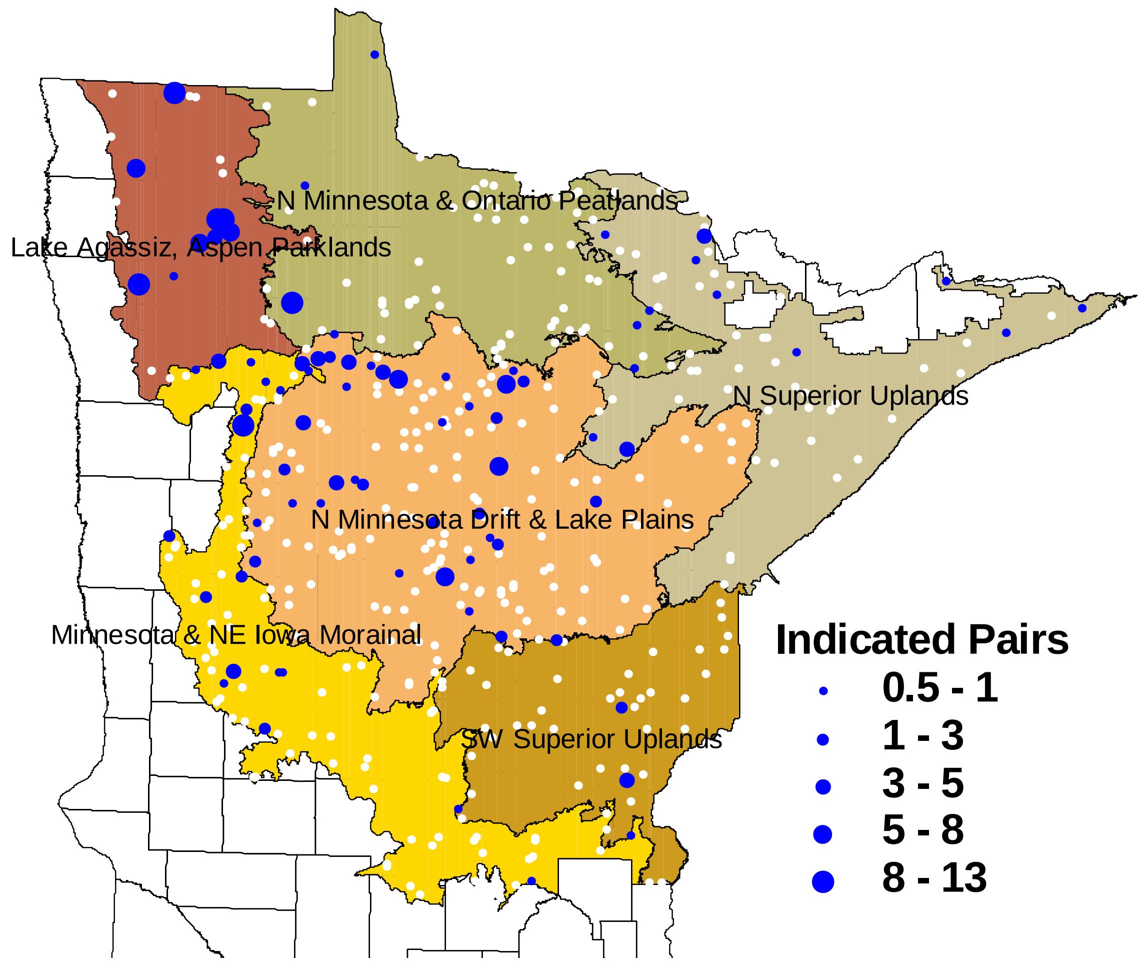

Figure 2. Plot locations and numbers of indicated breeding pairs of ring-necked ducks observed on survey plots in the Minnesota survey area, June 2004-2005.

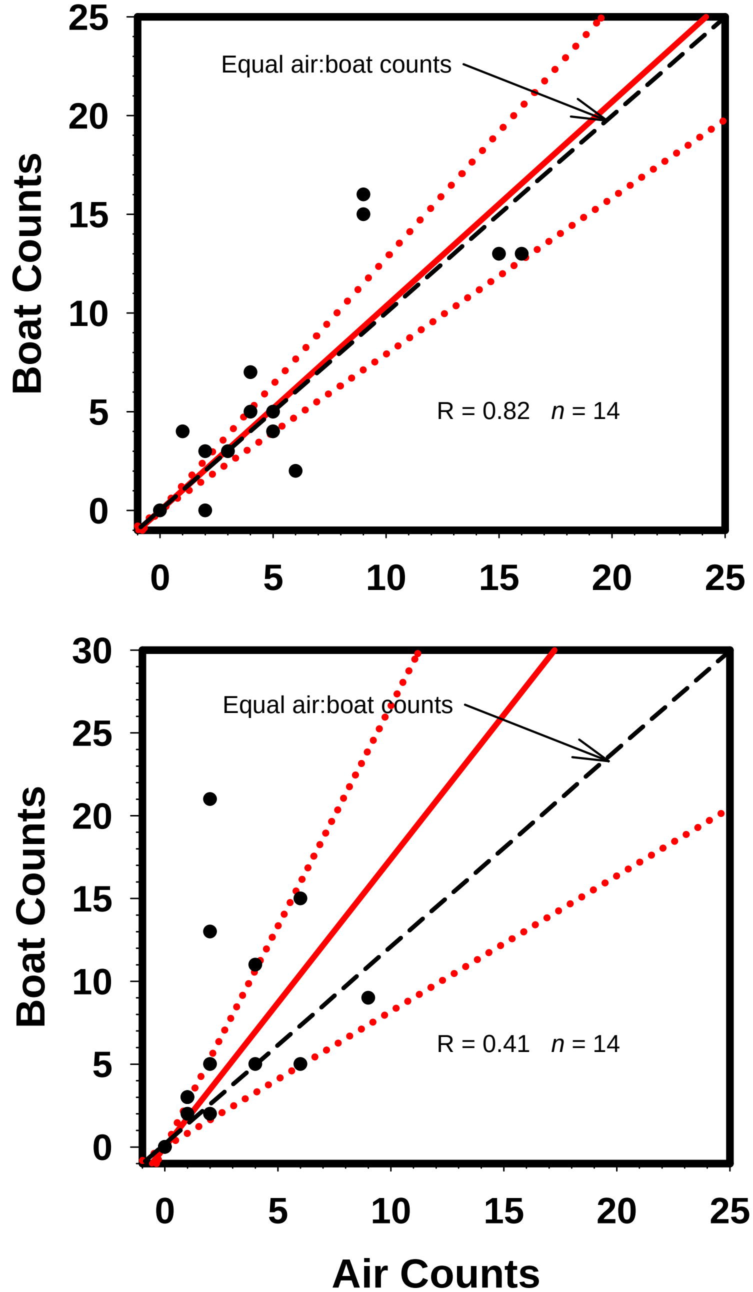

Figure 3. Regression lines and 95% confidence intervals comparing the numbers of indicated ring-necked duck breeding pairs counted from a boat and from the air on 14 lakes comprising the *Bemidji Area Ring-necked Duck Survey*, June 2004 (top) and 2005 (bottom).

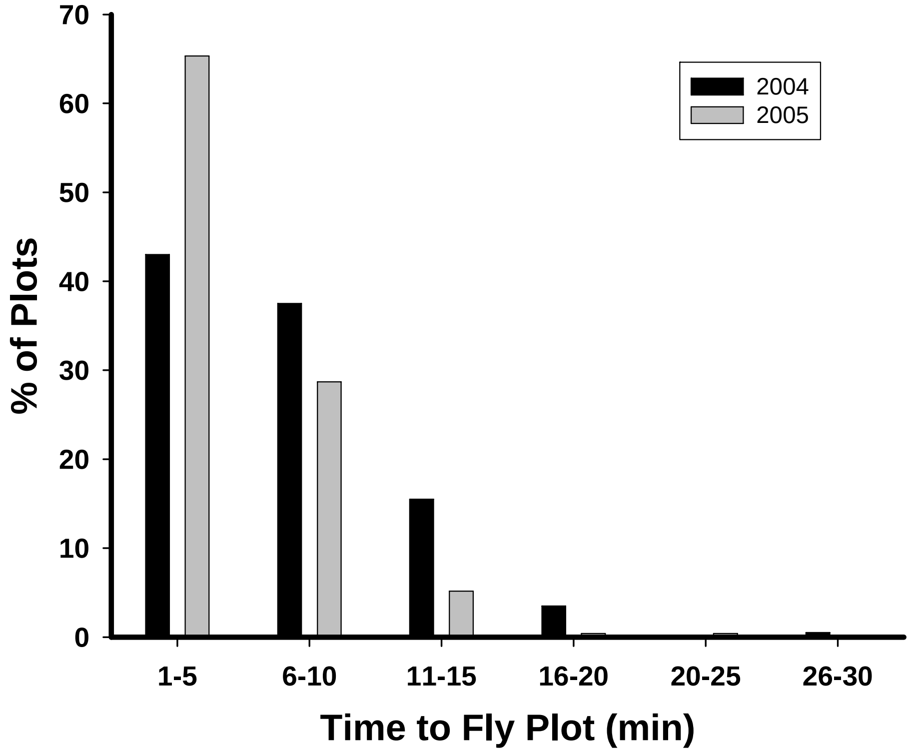

Figure 4. Time required (all observers) to survey individual ring-necked duck breeding pair plots in the Minnesota survey, June 2004 – 2005.

COST EFFECTIVENESS OF SINGLE- VS. DOUBLE-CYLINDER OVER-WATER NEST STRUCTURES[1]

Michael C. Zicus, David P. Rave, and John R. Fieberg

Abstract: Minnesota waterfowl management plans prescribe widespread deployment of mallard (*Anas platyrhynchos*) nest structures. We compared 53 single- and 57 double-cylinder structures from 1996 – 2003 because managers used both structure types but were uncertain about their respective cost effectiveness. More nests occurred in doubles, but numbers of successful nests and hatched ducklings were comparable for both types. Nest success in singles and doubles was 92.8% and 79.4%, respectively, with nest abandonment being >4.5 times greater in doubles. Structure damage occurred only at ice out and was greater for doubles. However, relative risk of failure for double- vs. single-cylinder structures was similar (1.26; 95% confidence interval = 0.91 – .75) and increased with size of the open-water area containing the structure. Modeling indicated ~95% of recruits from nest structures were additional recruits. A case history approach indicated doubles produced an additional recruit for $23.11 vs. $23.25 for singles. However, these estimates were sensitive to assumptions used to apportion costs between structure types and ignored structure placement influences. Placement affected cost effectiveness significantly with structures placed in open-water areas >10 ha being more cost-effective. Results also suggested singles might be more effective than doubles when placement is considered. Lower nest abandonment alone might make single-cylinder structures the better choice.

[1] Abstract of paper in press in the Wildlife Society Bulletin. 2006. Volume 34.

LAKE CHRISTINA RECLAMATION: ECOSYSTEM CONSEQUENCES OF BIOMANIPULATION

Mark A. Hanson, Joseph Allen[1], Deborah Buitron[1], Malcolm G. Butler[1], Todd Call, Thomas Carlson, Nicole Hansel-Welch, Katie Haws, Melissa Konsti[1], Dan McEwen[1], Gary Nuechterlein[1], and Kyle D. Zimmer[2]

SUMMARY OF FINDINGS

We assessed early patterns in fish community characteristics, limnological features, and wildlife use of Lake Christina following the 2003 rotenone treatment. Following rotenone application, fish were reduced, but not eliminated, and a diverse population comprised of planktivorous, benthivorous, and piscivorous species was again present by 2006. However, dramatic improvements in water transparency, increased density of large-bodied zooplankton (*Daphnia* spp.), and increases in occurrence of submerged aquatic plants (especially *Chara* spp.) were also evident and, collectively, these results seem to indicate the onset of a shift back to the clear-water state.

INTRODUCTION

Lake Christina, a 1,619-ha shallow lake in Douglas County in west-central Minnesota, is nationally recognized as a critical staging area for migrating canvasbacks, and also is a breeding location for a number of unique nongame bird species. Since the 1950s, the lake has alternated between ecological extremes, sometimes characterized by favorable conditions, and at other times of little use as waterfowl habitat. Sustained high water and dense populations of undesirable fish species are believed to be associated with shifts toward high turbidity and other unfavorable limnological characteristics, along with extreme habitat deterioration for waterfowl and other wildlife. Following obvious trends of habitat deterioration, the lake was “reclaimed” in 1965 and 1987 via chemical removal of fish. Extensive scientific monitoring was conducted in association with the 1987 treatment. Limnological and waterfowl-use data were gathered before and after the 1987 fish kill to assess the nature and causal mechanisms associated with observed changes. Dramatic improvements in water quality features, extensive development of submerged aquatic plants, and increased fall use by migrating ducks followed the 1987 reclamation (Hanson and Butler 1994, Hansel-Welch et al. 2003). Research before and after treatment contributed to improved understanding of ecology and management potential of shallow lakes in North America. Unfortunately, data gathering efforts at Lake Christina subsequently dwindled, more or less at the same time as habitat quality and suitability for wildlife again declined. During 2000-2003, water clarity, distribution of submerged macrophytes, and fall use by migrating ducks all indicated that the lake had again stabilized in a deteriorated condition characterized by poor water quality, a sparse community of submerged macrophytes, and limited suitability for diving ducks and other wildlife species. Fish were removed from Lake Christina using rotenone during October 2003 to stimulate a limnological shift to more favorable habitat conditions. Here, we summarize responses of fishes, limnological features, and wildlife use during 2004-05, the first two years following the fish removal. Our objectives were to evaluate broad ecosystem-level responses of the lake to the 2003 fish removal, with special emphasis on patterns of recruitment by fish that either survived the rotenone treatment, or immigrated into the lake following the fish kill. Here, we report preliminary patterns in fish populations, seasonal water transparency, abundance of large filter-feeding zooplankton (*Daphnia* spp.),

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occurrence of submerged macrophytes, fall use by waterfowl, and nesting efforts by colonial waterbirds.

METHODS

We used a variety of techniques to collect data summarized in this report. Adult, juvenile, and larval fish were sampled from May – August using gill and trap nets, beach seines, minnow traps, ichthyoplankton push nets, and boom electrofishing. Water transparency was determined using a Secchi disk, and by measuring vertical light attenuation in the water column. Zooplankton were collected using a vertical column sampler. Submerged macrophytes were sampled using weighted plant rakes. Fall use by ducks and geese was assessed as numbers observed during aerial and ground counts during late September – mid-November. Western grebe (*Aechmophorus occidentalis*) nests were counted during weekly surveys using kayaks. Methods used in collecting these and other data are discussed in greater detail in Hanson et al. (2006).

RESULTS AND DISCUSSION

We believe conditions observed during 2004-05 indicated that Lake Christina has entered a period of transition, and is tending back toward the clear-water state. Our results indicated presence of a persistent fish community during spring 2004, approximately 6 months after the October 2003 rotenone treatment. Recruitment by remnant fishes was very strong (Figure 1) and, by 2005, a diverse fish community was again present and included benthivorous, planktivorous, and piscivorous species (Figure 2). Disappointing, but not unexpected, was evidence of rapid recovery by bullheads, carp, and fathead minnows during the 2 years immediately following the rotenone treatment. Data gathered during 2004-05 also contain strong signals indicating a shift towards more favorable ecological conditions (as described by Scheffer et al. 1998). While changes in abundance of large-bodied herbivorous zooplankton (*Daphnia* spp., Figure 3) were equivocal, concomitant lake-wide trends toward higher water transparency during spring periods (Figure 4), and changes in abundance and composition of submerged aquatic plants (Figure 5) are consistent with outcomes lake managers had hoped to achieve, and with patterns observed following the 1987 rotenone treatment. One of the most encouraging signals observed following the 2003 rotenone treatment was the sharp increase in *Chara* spp. during 2004, the first post-treatment year (Figure 5B). Sharp increases in *Chara* spp. often portend major ecological shifts towards a clear-water state in shallow lakes and a similar trend was also observed within a year following the 1987 rotenone treatment at Lake Christina. Fall use by migrating ducks, coots, and Canada Geese (*Branta Canadensis*) also increased during 2004, a pattern also similar to that observed during 1988-1989 (Figure 6). Finally, we emphasize that even if the over-all lake response is ultimately similar to that observed following the 1987 treatment (and induces a transition to the clear-water state), more dramatic, sustained improvements in water transparency may not be evident until 2006, or even later. Non-target effects of rotenone in shallow lakes and wetlands may be considerable, but are rarely considered in lake rehabilitation studies. For example, Lake Christina has supported breeding western grebes since the late 1960s and a large population was observed using the lake during 2003. Availability of small prey fishes is considered crucial for successful recruitment of western grebes because adults fly infrequently other than during migration. During 2004, and following the 2003 rotenone treatment, adult western grebes returned to Lake Christina, but quickly abandoned traditional nesting areas and left the lake, presumably due to absence of suitable prey. By 2005, western grebes returned in large numbers and over 300 nests were identified and monitored. This may indicate that non-target effects of rotenone on some colonial waterbirds should be expected, but are short-term in that breeding waterbird populations return in response to recruitment of young fishes.

Comparison among historical relationships has great potential to help researchers identify signals of transition, thus indicating if and when lake-wide changes are underway. Lake managers have continuing needs to identify limnological signals useful for anticipating periods of rapid change, especially when the lake is entering transition to the turbid-water state. This would facilitate better use of less drastic measures to maintain a clear-water state. For example, since 1999, environmental signs showed evidence that the lake was probably transitioning towards the turbid state. In retrospect, we know that this was true. For example, TP:chl a ratios may be important indicators of the ecological state of this and other shallow lakes (Dokulil and Teubner 2003), and researchers may benefit from monitoring trends relative to the 3:1 threshold (Figure 7). Alternatively, based on results of indicator species analyses, concern may be justified when high counts of small cladocerans such as *Bosmina* spp. consistently occur. Additionally, it may be possible to use the importance values of *Chara* to monitor whether the lake is stable or in transition. If *Chara* spp. shows sharp lake-wide declines, as it did during the period of 1999-2001, then perhaps the onset of a period of deterioration and a shift to the turbid state may be anticipated.

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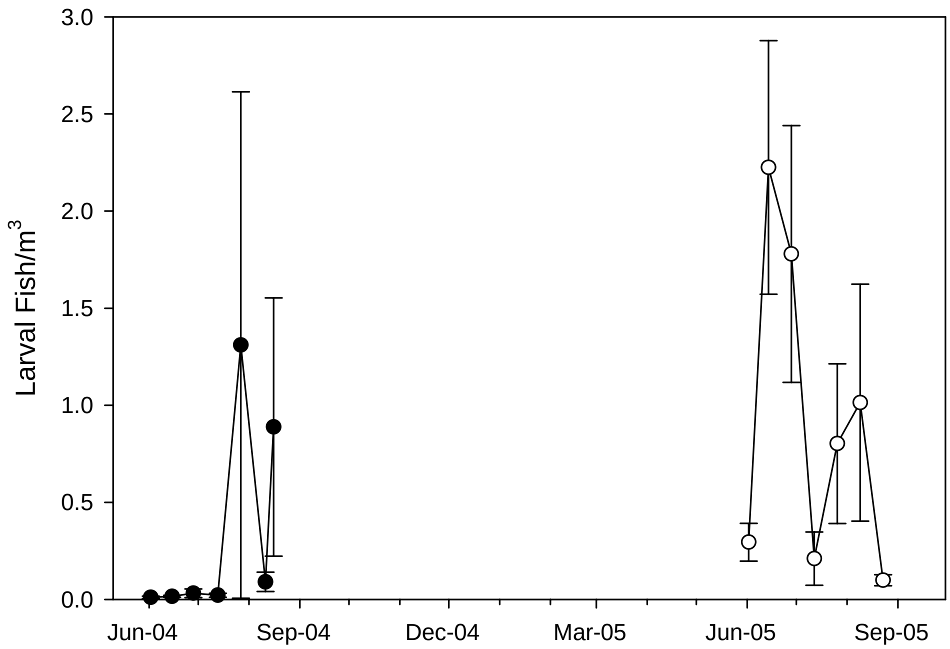

Figure 1. Larval fish tow results for 2004 and 2005 (average larval fish/m^3). Vertical bars indicate +/- 1 standard error.

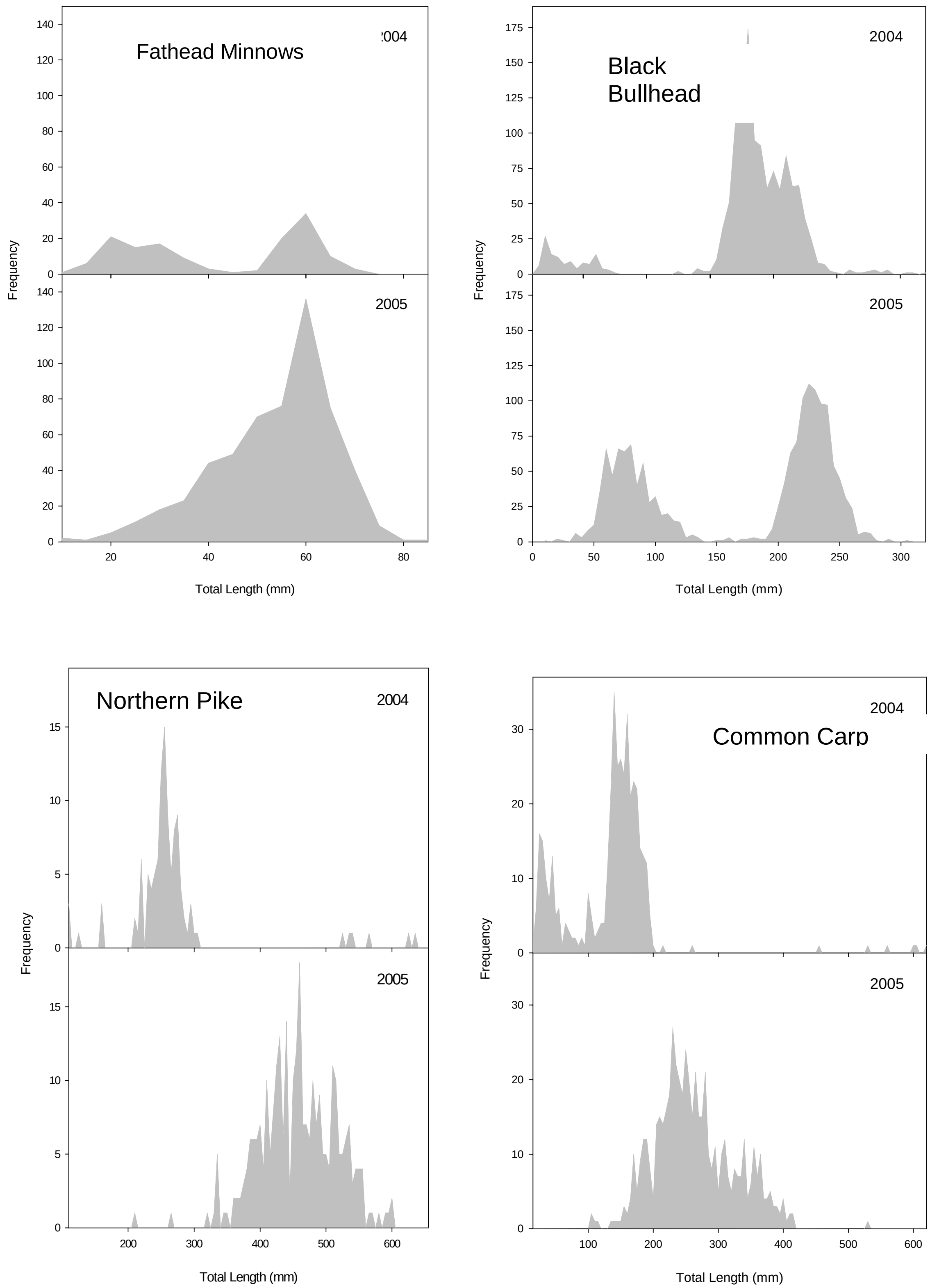

Figure 2. Length distribution of common fishes captured using beach seines at Lake Christina during 2004-05.

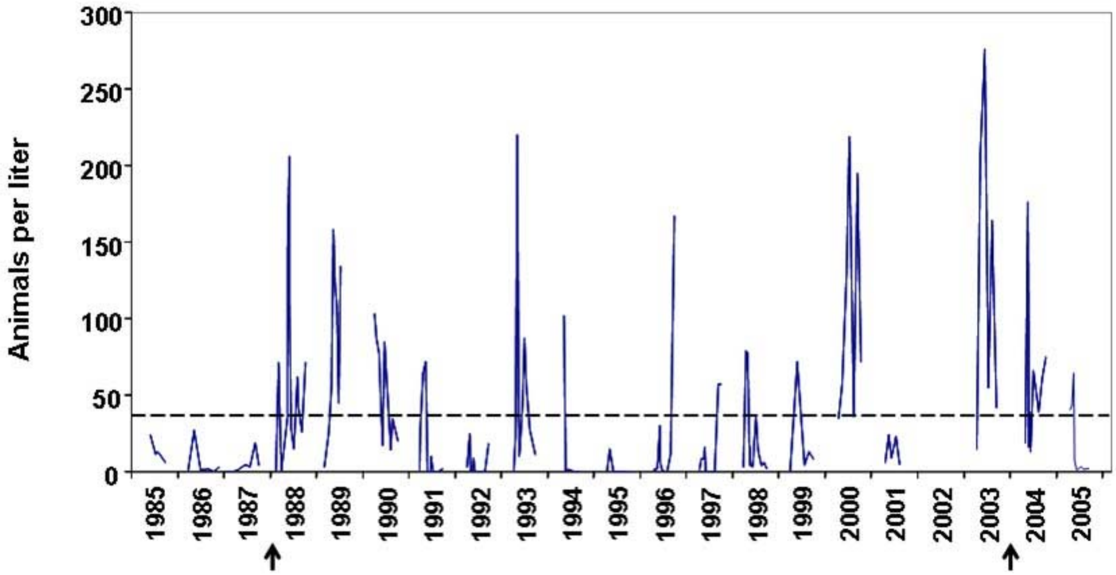

Figure 3. Mean lake-wide *Daphnia* spp. density (no./liter) in Lake Christina, 1985-2005. Arrows indicate rotenone treatments in October of 1987 and 2003. Dashed line indicates long-term mean over the 21-year record.

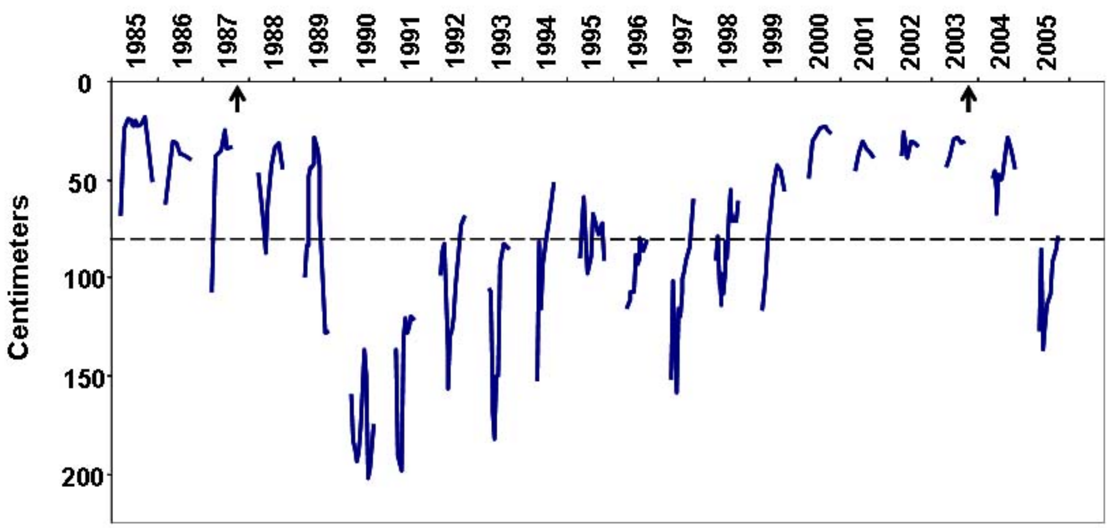

Figure 4. Mean lake-wide secchi depth (cm) in Lake Christina, 1985-2005. Arrows indicate rotenone treatments in October of 1987 and 2003. Dashed line indicates long-term mean over the 21-year record.

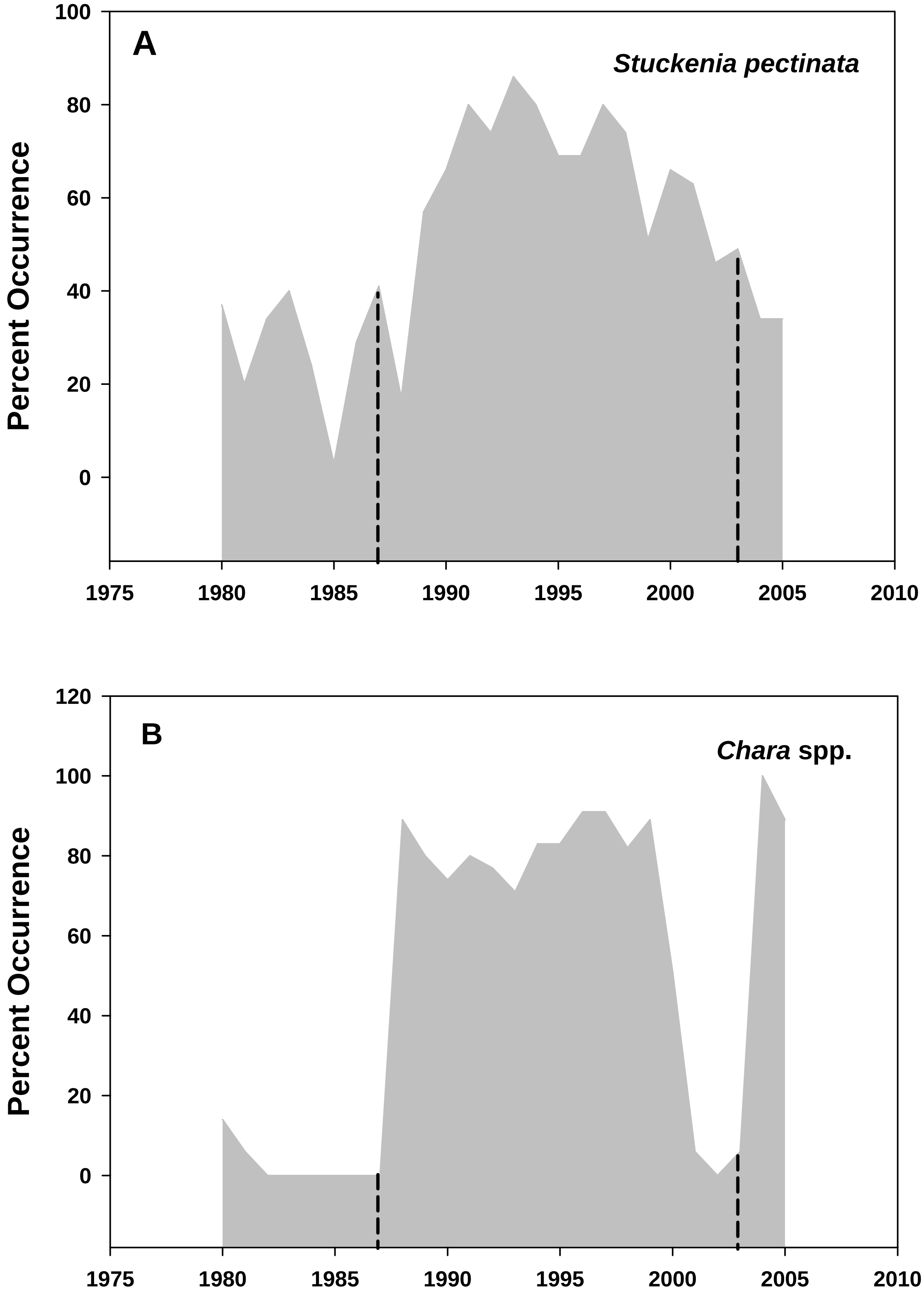

Figure 5. Results of submerged aquatic plant surveys at Lake Christina during 1980-2005. Plotted values indicate percent occurrence of 2 species (*Stuckenia pectinata* (A), and *Chara* spp. (B)), sampled at 35 locations around the lake. Hatched lines indicate timing of rotenone treatments (1987, 2003).

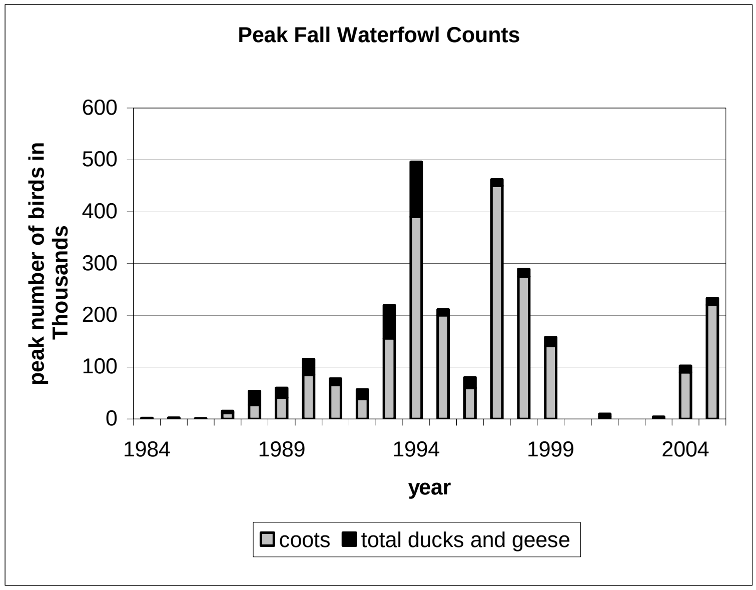

Figure 6. Annual peak waterfowl estimates for Lake Christina during 1984-2005.

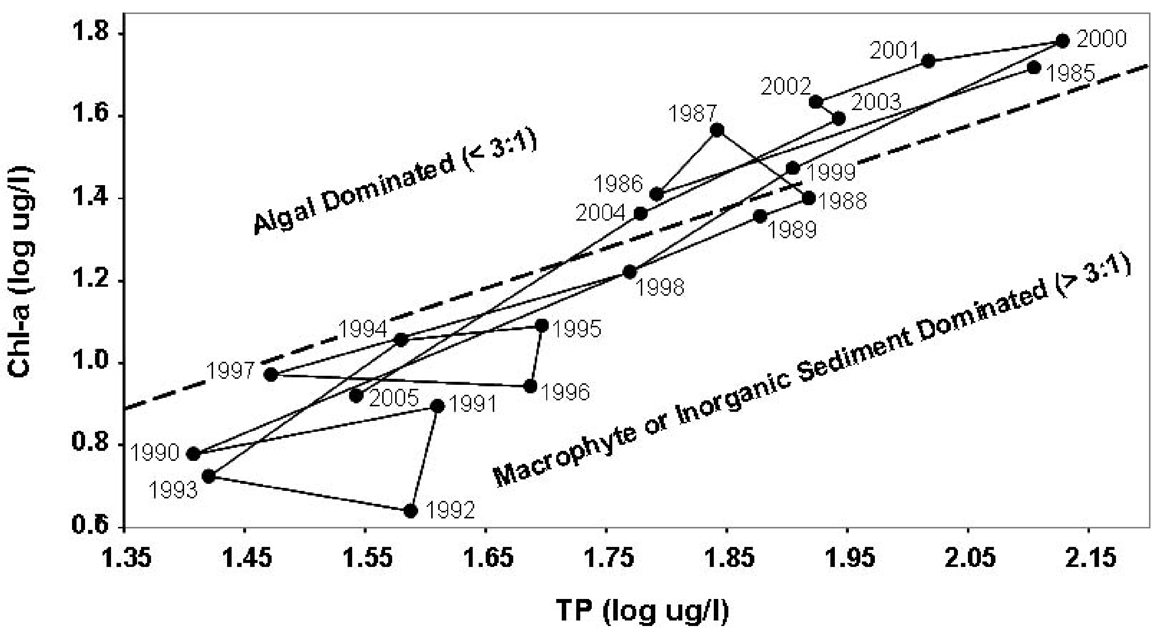

Figure 7. Chronological history of total phosphorus (TP):chlorophyll *a* (chl-*a*) ratios in Lake Christina, 1985-2005. Dashed line indicates 3:1 ratio of TP:chl-*a*; shallow lakes with values below this line often exhibit characteristics of the clear-water state.

RELATIONSHIPS AMONG LANDSCAPE FEATURES, FISH ASSEMBLAGES, AND SUBMERGED MACROPHYTE COMMUNITIES IN PRAIRIE WETLANDS

Mark A. Hanson, Brian R. Herwig, Kyle D. Zimmer[1], and Jerry A. Younk

SUMMARY OF FINDINGS

We are assessing fish community patterns and influences of site- and landscape-level variables on fish assemblages and various ecological features of prairie wetlands in two areas in western Minnesota (generally Polk and Grant County areas). Fish populations during the first year of the study (2005) 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. In general, landscape-scale variables were not useful in predicting presence of fish populations in study wetlands, but fish communities tended to reflect influences of wetland size, depth, and presence of piscivorous fish species. Biomass of planktivorous fish was not related to abundance (mass) of submerged macrophytes in our study wetlands. In contrast, biomass of benthivorous fish was negatively related to mass of submerged macrophytes in Grant, but not Polk County wetland sites. We believe this indicates presence of a strong interaction between benthivorous fish and ambient nutrient concentrations, perhaps indicating greater potential for macrophyte loss with introduction of benthivorous fish in Grant County wetlands. These results are preliminary and similar data will be gathered in 2006.

INTRODUCTION

Fish communities exert strong, but variable, influences on ecological properties of deep prairie wetlands and shallow lakes. For example, previous research has shown that dense populations of fathead minnows *(pimephales promelas)* have key structuring influences on invertebrate populations and wetland community characteristics (Zimmer et al. 2002), although additions of piscivores (e.g., walleye fry) may negate those effects (Herwig et al. 2004). Less is known about ecological roles of benthivorous fishes, but their presence is often associated with turbid conditions.

Winter hypoxia and isolation are believed to be major constraints on wetland fish communities throughout the Prairie Pothole Region (PPR) of North America. Recently, some authors have suggested that distribution of fishes has increased among PPR wetlands due to anthropogenic activities and, perhaps, climate extremes. However, the distribution and community characteristics of wetland fishes across the PPR are poorly known. Past research has not assessed influences of both scale-dependent spatial factors and site-level environmental mechanisms that control distribution of fishes in prairie landscapes, while simultaneously evaluating influences of specific fish assemblages on wetland features.

During 2005-06, we were exploring patterns and assessing influences of spatial and site-level variables on fish communities in 73 deep wetlands and shallow lakes (wetlands) in west-central Minnesota, USA, an area along the eastern margin of the PPR (Figure 1). Two focus areas were chosen for study, with 36 and 37 sites along borders of Polk/Mahnomen (PM) and Grant/Stevens (GS) counties, respectively. Because it is widely believed that anthropogenic disturbance is greater in the GS area, including data from these regions provided a means of capturing influences of a potential land-use gradient in our spatial and environmental data. Here, we report results of preliminary analyses used to 1) identify patterns in wetland fish communities, 2) relate fish community assemblages to site- and landscape-level variables, and 3) assess potential

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relationships between biomass of planktivorous and benthivorous fish and submerged macrophytes in study wetlands.

METHODS

We estimated presence and abundance of fish in study wetlands using a combination of mini-fyke nets, gill nets, and minnow traps. Chlorophyll *a* was estimated according to procedures followed by the Minnesota Department Agriculture chemistry lab (St. Paul, MN) and was used as an index of phytoplankton biomass. Submerged macrophytes were sampled using a weighted plant rake. Samples of fish and submerged plants were weighed on site to provide indexes to abundance. We used Principle Components Analysis (PCA), to examine potential fish assemblage patterns in preliminary data collected during 2005. We used Canonical Correspondence Analysis (CCA) to relate site- and landscape-level variables to patterns in wetland fish communities. Lowess regression was used to evaluate the relationship between chlorophyll *a* (natural log) and biomass of submerged aquatic plants (natural log [n+1]) sampled in wetland study sites. Finally, we used analysis of covariance (ANCOVA) to relate biomass of planktivorous and benthivorous fish to mass of submerged plants.

RESULTS AND DISCUSSION

Fish were more widespread and fish communities were more complex than expected (Figure 2). PCA of fish abundance data indicated four distinct fish community types, including: 1) fishless, 2) minnow-only, 3) multi-species communities with black bullheads, and 4) multi-species communities including piscivores, where minnows were strongly suppressed. Observed fish community patterns reflected strong gradients of piscivory as well as wetland depth and size. For GS sites, CCA identified two significant environmental variables ($p<0.05$): maximum wetland depth and surface area. For PM sites, CCA identified only maximum depth as a significant source of variance. Our results indicated that piscivory is an important mechanism structuring fish communities in these wetlands, but also that smaller, shallower wetlands tended to have relatively simple fish communities and were often dominated by planktivorous species such as fathead minnows.

Because submerged macrophytes and planktonic algae reflect broad ecological properties of wetlands and shallow lakes (Scheffer 1998), we also assessed influences of fish communities and ambient nutrient levels on abundance of submerged macrophytes and algae during 2005. Nutrient levels were generally much higher in the GS wetland sites. Either submerged macrophytes or planktonic algae dominated wetlands in both study regions. As chlorophyll *a* increased from 5 to 50 ppb, submerged macrophytes declined 71-fold (Figure 3). Frequency of algal dominance (chlorophyll *a* > 19ppb) differed between areas, with 31 of 37 wetlands algal-dominated in the GS, compared to 8 of 35 sites in the PM region (Figure 4). Planktivore and macrophyte abundance were not related in either study area. However, benthivore and macrophyte abundance were negatively related in the GS, although no similar relationship was detected in the PM region (Figure 5). Our results indicated that macrophyte abundance was much more strongly influenced by benthivores than by planktivores, but the strength of benthivore influences depended upon ambient nutrient levels in this landscape.

Ducks depend upon quality wetland and shallow lake habitats throughout the PPR. Certain fish communities have the potential to reduce ecological integrity of wetlands, limiting suitability of these areas for breeding and migrating ducks. Wetland managers need tools useful for predicting ecological consequences of practices that increase connectivity and permanence of wetlands and shallow lakes throughout the PPR. Our results should aid in development of models useful for predicting both fish presence and community types in PPR wetlands, and for assessing potential ecological implications of specific fish assemblages in wetland habitats.

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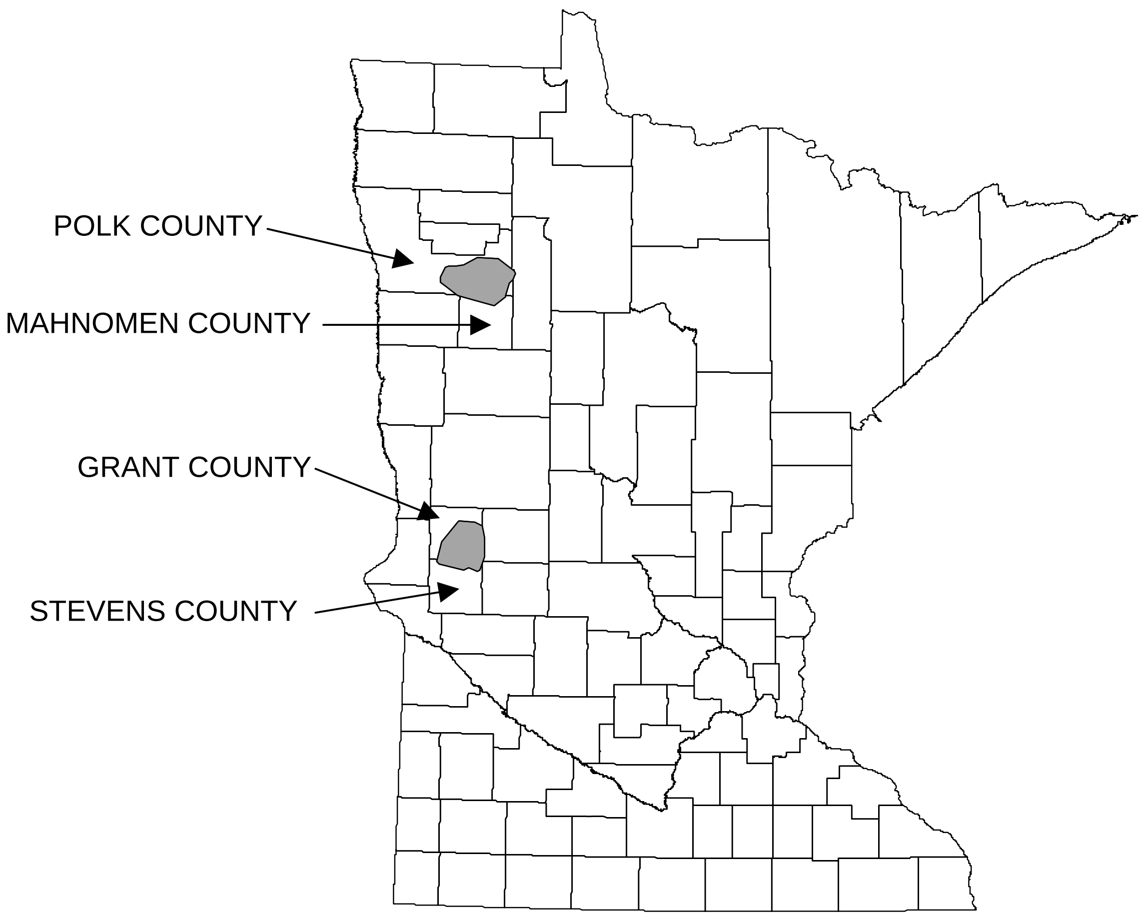

Figure 1. Locations of study focus areas, each defined by a polygon drawn around the outermost 1-mile buffers surrounding each of the study sites.

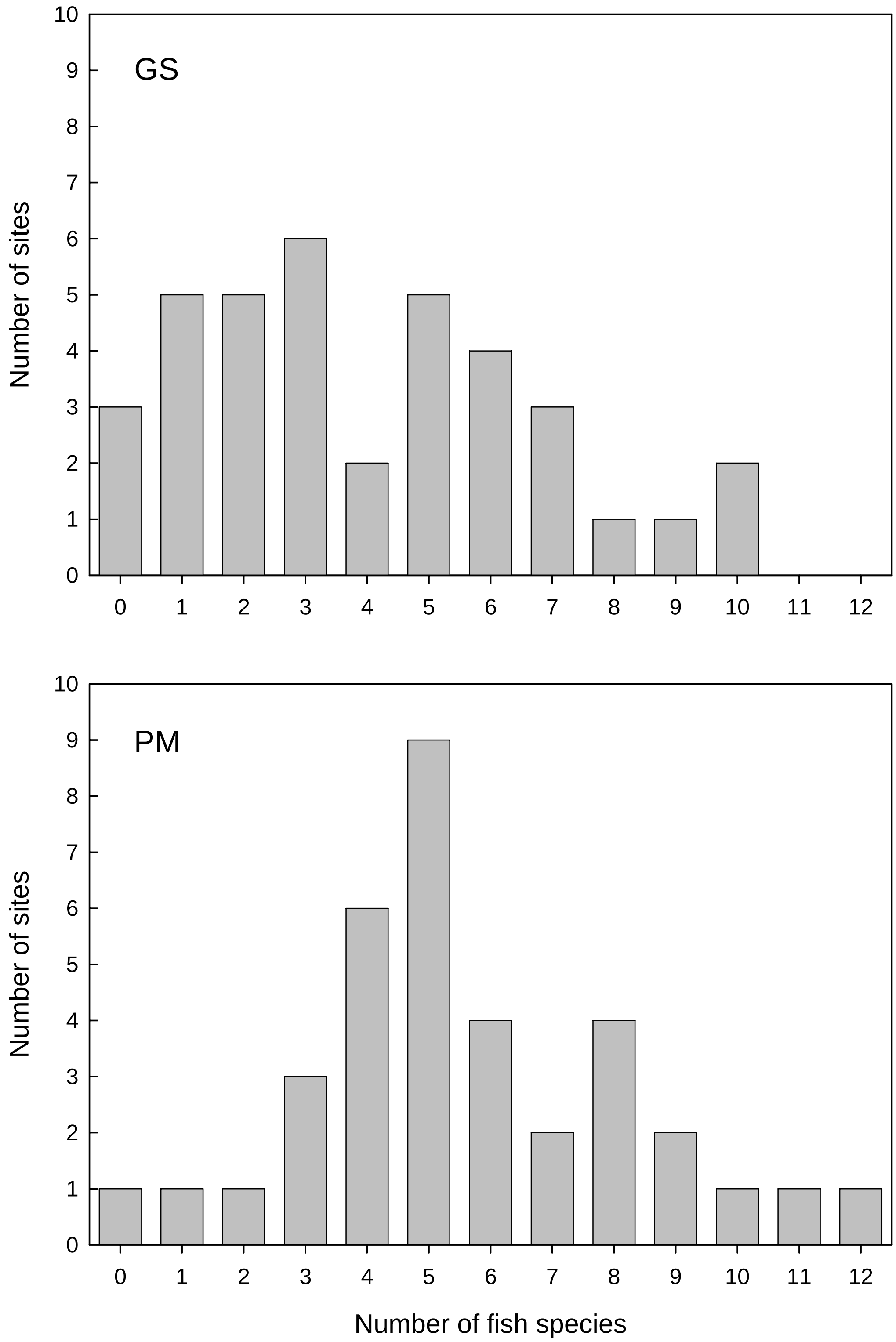

Figure 2. Frequency distribution showing fish species richness across study sites located within the Polk/Mahnomen (PM) and Grant/Stevens (GS) focus areas.

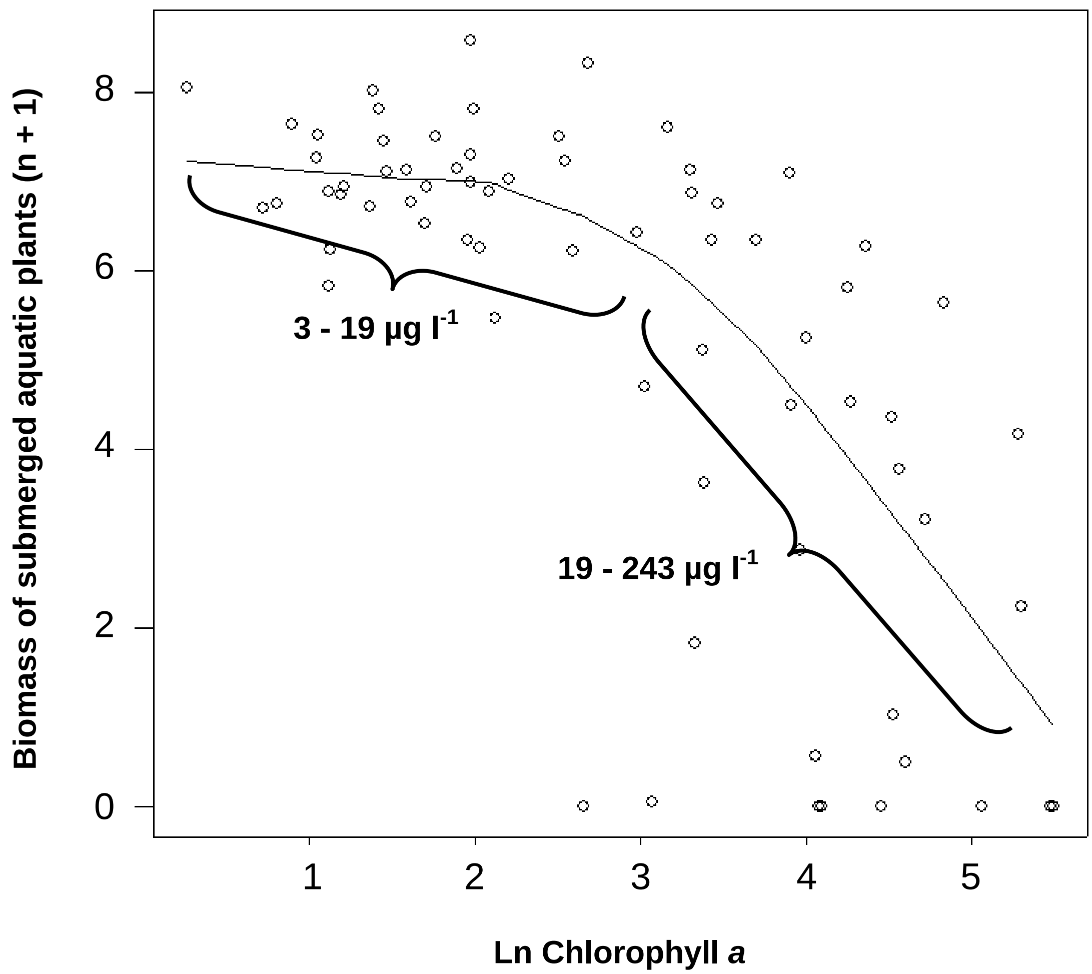

Figure 3. Lowess regression relationship between chlorophyll *a* (natural log) and biomass of submerged aquatic plants (natural log [n+1]) sampled in wetland study sites during July and August 2005.

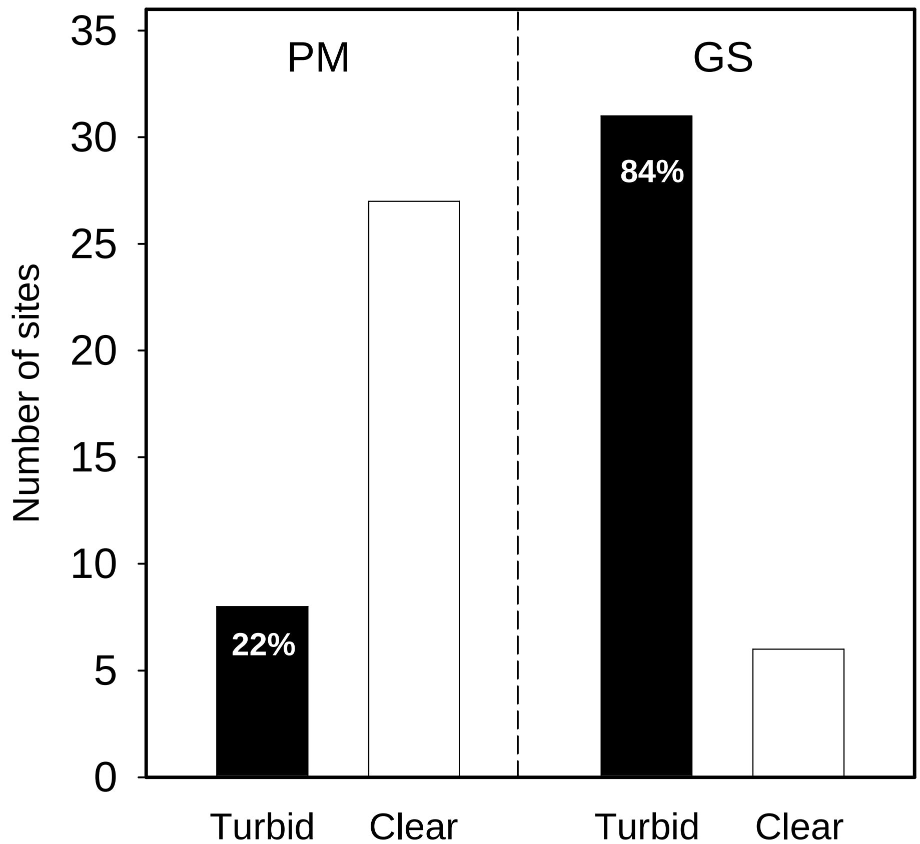

Figure 4. Proportion of turbid vs. clear wetland sites (based on threshold of 19 µg l^{-1}) sampled in Polk/Mahnomen (PM) and Grant/Stevens (GS) focus areas during 2005.

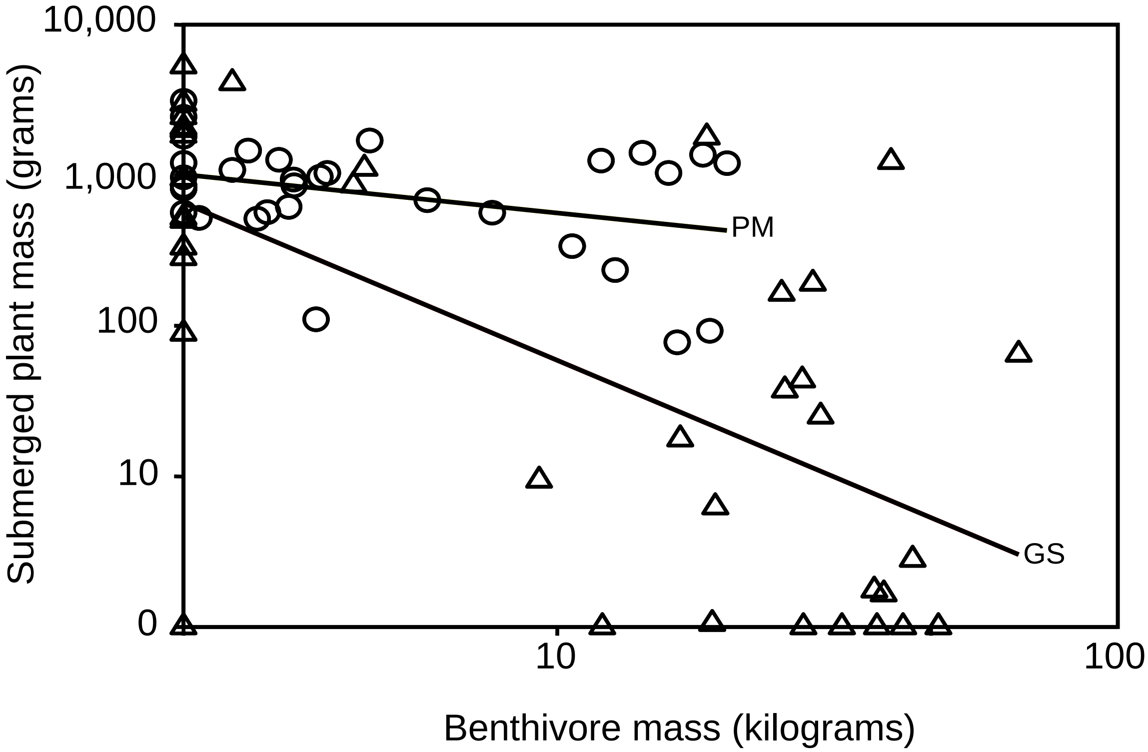

Figure 5. Relationships among submerged aquatic plants and biomass of benthivorous fishes sampled in Polk/Mahnomen (PM) and Grant/Stevens (GS) focus areas during 2005; open circles = PM, Open triangles = GS. Lines indicate relationships fitted separately to data from PM and GS focus areas using ANCOVA. Note that the slope of best-fit lines differed from 0 ($P<0.01$; $R^2=0.43$) for GS sites, but not for PM sites ($P=0.34$; $R^2=0.13$).

SEASONAL FOREST WETLANDS: CHARACTERISTICS AND INFLUENCES

Shane Bowe[1], Mark A. Hanson, Matt Bischof[1], and Rick Koch[1]

SUMMARY OF FINDINGS

Seasonal forest wetlands are abundant and broadly distributed throughout aspen-dominated landscapes in Minnesota's Laurentian Mixed Forest. Interest in seasonal wetlands has increased in recent years due to increased awareness of their ecological significance, and because these habitats are often influenced by silviculture activities. It is evident that site-level characteristics of seasonal wetlands are functionally linked to adjacent forested uplands. Forest wetlands receive major energy inputs through deposition of leaf-litter from the adjacent forest (Oertli 1993). Clear-cut timber harvest may have unexpected consequences for adjacent wetlands including modified vegetation and hydrology, increased sedimentation, reduced evapotranspiration, and soil desiccation. It is likely that biotic communities and physical attributes of small wetlands are often altered, but to date, relationships between silvicultural activities and small wetlands are poorly known, and little information is available to guide forest and wildlife managers who are interested in conserving integrity of small riparian areas.

INTRODUCTION

Since 1999, we have studied 24 small, seasonally-flooded ($\leq$ 0.6 ha) wetlands in aspen-dominated landscapes of the Buena Vista and Paul Bunyan State Forests in north central Minnesota. Study wetlands are assigned to one of three "age-class" levels of treatment, or identified as controls based upon adjacent forest (stand) age-since-harvest using natural breaks identified with Arcview. We blocked study sites based on proximity to account for local influences of soils, landforms, or other geophysical features. We assigned study wetlands to clusters, each comprised of 4 adjacent wetlands (1 in each of 4 treatment groups) located within the same general state forest area. Each state forest (hence subsection of the Ecological Classification System [ECS], Almedinger and Hanson 1998) contained three clusters of four wetlands, including one control, 2 effect/recovery sites, and 1 clearcut treatment site (total of 12 sites per state forest). Control sites were those with no adjacent forest harvesting during the past 59+ years. Treatment sites included one 59+ year area that was harvested during the winter of 2000-2001 (clearcut treatment), and 2 effect/recovery sites consisting of wetlands in stands harvested 10-34 (young-age) and 35-58 (mid-age) years before present. Overall, our design included 6 replicate sites within these 4 age-class treatments. Data gathering and analyses associated with this initial phase of the research are well underway. These analyses will assess in more detail wetland characteristics and potential changes observed during 2001-2005, the initial period following clear-cutting in adjacent uplands (winter 2000/2001). Here, we report on preliminary analyses of invertebrate-community responses, and related environmental changes including leaf litter and duration of ponding (hydroperiod), both attributes likely to be influenced by timber harvest. Our objectives were to: 1) characterize community features and identify site-level environmental characteristics of seasonal wetland habitats in the Laurentian Forest, and 2) evaluate initial responses of aquatic invertebrate communities and other wetland features to clear-cut timber harvest.

METHODS

We sampled aquatic invertebrates using surface-associated activity traps (SAT; Hanson et al. 2000) deployed for 24

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hrs at random locations near the margin of each wetland. Five traps were used concurrently in each wetland. Aquatic macroinvertebrates were sampled during open-water periods, at approximately 3-week intervals during May, June, and July 1999-05. Vertical and horizontal leaf litter traps were implemented at one cluster (4 wetlands) in each forest during 2004-05. Leaf litter was collected every two weeks during September-mid- November. Litter samples were dried at 60˚C for 24 hrs, weighed and combusted at 450˚C for 4 hrs to determine organic matter content (ash-free dry weight). Site-level measurements of vertical distance to groundwater were obtained using networks of piezometers and monitoring wells (following methods of Sprecher 2000). Single wells were established in the deepest portion of all wetlands during 2004 to assess approximate distance to upper limits of groundwater. Additionally, during 2005, piezometer nests were deployed at 8 wetland sites to more accurately characterize relationships between groundwater movements and wetlands. Wetland maximum depth was recorded weekly from spring thaw until surface water disappeared, and every two weeks thereafter until frozen. Additional measurements were made at these wetland sites during 1999-2005 (Ossman 2001).

Invertebrate data were analyzed to identify potential patterns using Nonmetric Multidimensional Scaling (NMS). We used NMS to compare invertebrate community structure by ordination of site scores based on a dissimilarity matrix. Significance of patterns in our site scores were further assessed using Multi-response Permutation Procedures (MRPP; McCune and Grace 2002). Leaf litter data were assessed graphically and using independent samples t-tests (Green and Salkind 2005). Hydroperiod data (days of continuous inundation) were compared among treatments graphically and using ANOVA (Green and Salkind 2005). Results presented here are preliminary; interpretations are likely to change following additional data analysis.

RESULTS AND DISCUSSION

Invertebrate community composition showed a notable shift during the first 5 years following clear-cut timber harvests (Winter 2000-01). For example, during 2005, invertebrate communities from clear-cut sites exhibited higher within-group similarity than did non-clear-cut sites (Figure 1). Clear-cut invertebrate communities comprised a distinct group that was clustered based on dissimilarity with uncut wetland sites. Similar contrasts between invertebrate communities of clear-cut and other treatment sites also were observed during 2001-2004. Our results reflect patterns of change in wetland invertebrate communities, apparently in response to clear-cut timber harvest in adjacent uplands. Comparison of NMS site scores using MRPP indicated that dissimilarity between invertebrate communities of clear-cut wetlands and other treatment groups was greater than expected by chance ($T = -1.8$; $P<0.05$). This is not surprising given the widely held view that biological processes and communities in small, seasonal wetlands are functionally linked to adjacent upland areas (Palik et al. 2001).

Clear-cut harvesting modifies wetland hydroperiods (Verry 1997, Roy et al. 2000), leaf-litter inputs, light availability at the wetland surface, and water temperature, among other things. We observed obvious differences in wetland hydroperiods among our forest-age treatments; clear-cut wetlands maintained standing water longer than did all other groups ($F_{(3,20)} = 3.14$; $P<0.05$). During 2004, on average, study wetlands embedded in clear-cut harvests remained flooded approximately 45 days longer than did sites in old-growth aspen stands (Figure 2). Following adjacent clear-cut harvest, litter inputs to our wetland sites diminished ($T=3.02$; $P<0.05$; Figure 3), concurrent with sharp decreases in canopy closure.

Observed patterns in invertebrates are consistent with Church (2006) who also reported changes in similar communities following clear-cutting adjacent to seasonal ponds in north central Minnesota. Changes in invertebrates probably reflect cumulative influences of shifts in site-level environmental characteristics during periods immediately following clear-cutting. Oertli (1993) suggested that leaf litter constitutes the major source of energy for macroinvertebrate production in small wetlands, thus reductions in leaf litter inputs to our sites are likely associated with observed changes in invertebrates. Batzer et al. (2004) reported weak associations between wetland invertebrate communities and hydroperiods in seasonal ponds in north central Minnesota. Relationships between hydrology of small seasonal wetlands and clear-cut timber harvest are poorly understood. Some previous research indicates that tree removal has the potential to elevate water tables (Verry 1997, Roy et al. 2000) and modify local hydrology (Roy et al. 2000). Other unanticipated ecological responses to timber harvest are also possible. For example, extending hydroperiods of small forest wetlands may allow vertebrate and invertebrate predators to persist and disrupt natural community dynamics. Hence, other animals including amphibians and early arriving birds and waterfowl, may face added competition for food resources before larger water bodies become ice-free. We expect that subsequent data and analyses will provide better characterization of these wetlands and help clarify specific relationships between wetland communities and clearcut timber harvest.

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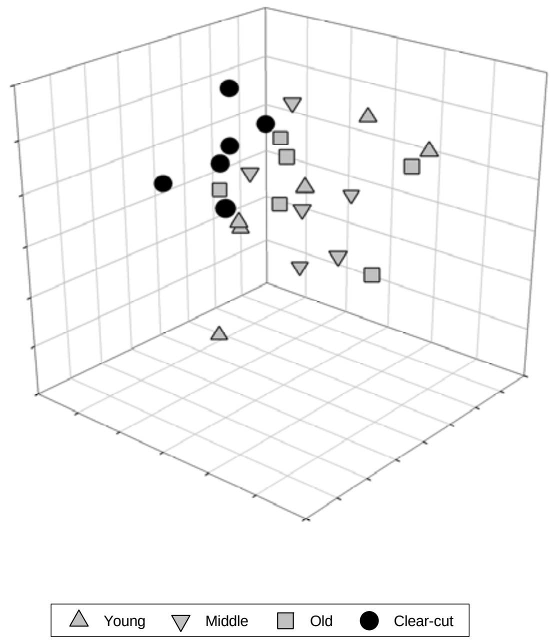

Figure 1. A three-dimensional NMS ordination of site scores based on dissimilarity in invertebrate communities among wetland study sites during 2005. Distances between plotted site scores illustrate extent of dissimilarity in invertebrate species composition. Symbols represent age-structure characteristics of adjacent uplands.

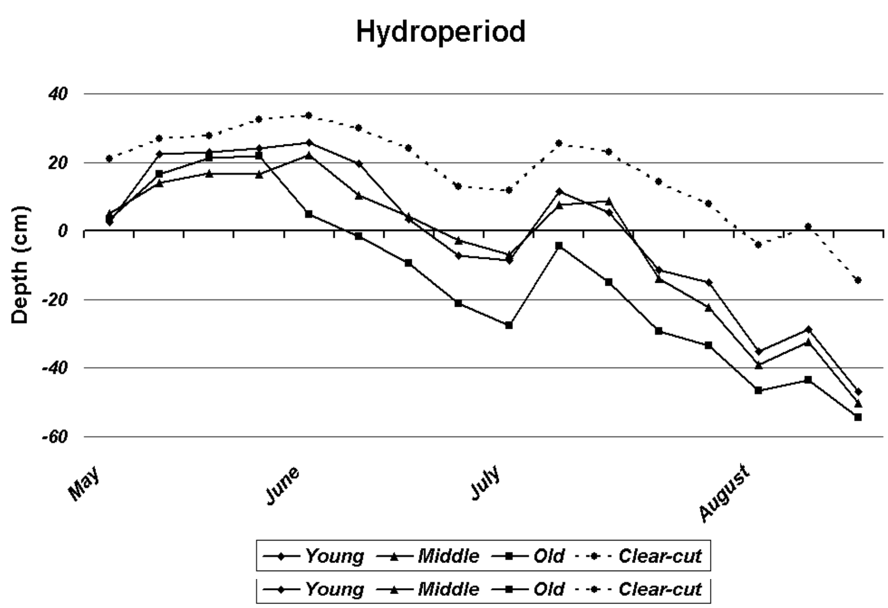

Figure 2. Average maximum depth (standing waters) and distance to groundwater for 24 wetland sites during 2004. A value of 0 cm indicates lack of standing water within the deepest portion of the wetland basin; negative values reflect approximate distance to upper limits of groundwater.

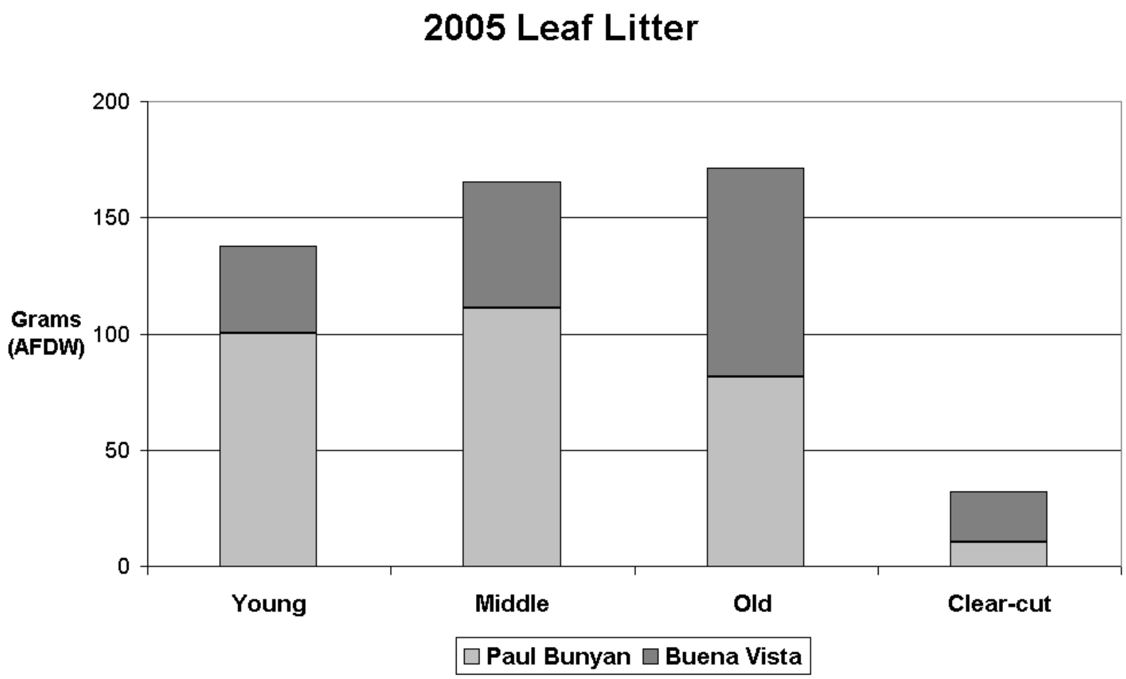

Figure 3. Total grams (Ash Free Dry Weight) of leaf litter collected from 4 sites in the Paul Bunyan State Forest and 4 sites in the Buena Vista State Forest during 2005.

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LANDOWNER ATTITUDES AND PERCEPTIONS REGARDING WILDLIFE BENEFITS OF THE CONSERVATION RESERVE PROGRAM (CRP)

Martin D. Mitchell[1], Richard O. Kimmel, Roxanne M. Franke[2], and N. Nicole Moritz[1]

SUMMARY OF FINDINGS

Landowner perceptions of farmland programs are important in their successful implementation. Our purpose was to survey landowners who were participating in the CRP and those who were non-participants in 1997 and 2006 to determine: 1) if there were differences in how each group perceived the CRP and its associated environmental impacts, and 2) if these perceptions change from 1997 to 2006. We found that all landowners had a dramatically enhanced sense of environmental awareness regarding wildlife habitat and particularly ring-necked pheasant (*Phasianus colchicus*) populations relative to the CRP in 2006. Attitudes of landowners in south-central Minnesota generally paralleled findings of a recent USGS study that addressed perceptions of CRP participants throughout the Corn Belt, though certain qualifications applied in our findings. Finally, perceptual differences between participants and non-participants noticeably narrowed from 1997 to 2006, indicating increased awareness of the intended conservation benefits of the CRP.

INTRODUCTION

Agricultural programs are dependent on both government legislation from which the programs originate and landowners who implement these programs. Landowner acceptance of agricultural programs is paramount for success. In the 1960s, there were high sign-ups indicating strong landowner interest for annual set-aside programs (Berner 1988). Concurrently, there was reduced interest in the Cropland Conversion Program of 1962 and the Cropland Adjustment Program of 1965, which were multi-year land retirement programs designed after the popular Soil Bank Conservation Reserve (Berner 1988, Kimmel & Berner 1998). A multi-year land retirement option was not available again until the Conservation Reserve Program (CRP) was authorized in 1985 and reauthorized in 1996 (Kimmel & Berner 1998). In Minnesota, a sign-up of 0.76 million ha (1.9 million acres) of CRP during the 1980's demonstrated the landowner interest in this program. Currently, almost 0.72 million ha (1.8 million acres) are enrolled in Minnesota (USDA 2006).

Several studies have described characteristics of CRP participants (e.g., Force and Bills 1989, Hatley et al. 1989, Mortensen et al. 1989). Miller and Bromley (1989) evaluated interest of CRP participants in improving wildlife habitat and stressed improved communication between farmers and wildlife professionals. Kurzejeski et al. (1992) found that when wildlife information was available, landowner participation in wildlife conservation measures increased.

More recent studies have focused on CRP's socio-economic effects and its perceptions of the program on the physical environment. Leistritz et al (2002) examined the socio-economic impacts of CRP in 6 different agricultural sub-regions of North Dakota. This study centered on surveying CRP participants and community leaders from the agri-business sector who were not participants in CRP. In another North Dakota study, Bangsund et al (2004) modeled the effects of enhanced hunting relative to the opportunity costs of CRP participants. Finally, the USGS (2003) conducted a national survey of CRP participants to determine their perceptions of wildlife, vegetation, and the general impacts and

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impressions of the CRP on the rural landscape.

The purpose of our investigation was to survey landowners in the Corn Belt region of south-central Minnesota to better understand their attitudes and perceptions about CRP, and its impact on wildlife abundance, and to see how such attitudes have changed or remained constant over the past 10 years.

METHODS

In 1997 we surveyed landowners in south-central Minnesota with questions regarding land ownership, enrollment in CRP, opinions on whether CRP improved habitat for wildlife, and factors influencing land-use decisions (Kimmel et al. 1997). A 25-question, 6-page survey was first mailed to 308 landowners on April 18, 1997. Using plat books, we selected landowners who owned land located on study areas used for an on-going investigation of avian population responses in the CRP (Haroldson, in press). Since 1990, we have monitored abundance of ring-necked pheasants, gray partridge (*Perdix perdix*), and meadowlarks (*Sturnella* spp.) on these study areas (Kimmel et al. 1992).

In February 2006, we prepared a similar, but smaller 14-question, survey that was implemented by telephone interview to 60 landowners located in south-central Minnesota. We attempted, whenever possible, to include the same landowners from the 1997 survey sample. With both studies, we divided the landowners on an approximate 50/50 ratio into CRP participants and non-participants to identify differences in perceptions between these two groups.

RESULTS

Following 4 mailings, 2 postcard reminders (after the 1st and 2nd mailings), and follow up phone calls, 219 of the 308 surveys were returned. The final survey mailings and phone reminders were conducted in July 1997. Undeliverable surveys and deceased landowners accounted for 44 unreturned surveys. The response rate for deliverable surveys (n=264) was 83.0%. Our telephone-based survey in February 2006 had a 100% compliance rate with 31 CRP participants (52%) and 29 non-participants (48%) comprising the final sample.

In 1997, land enrolled in CRP per farm averaged of 32.8 ha (81.9 acres) between 1985-1997. In 2006, this figure dropped to 14.8 ha (37 acres). Landowners with CRP owned an average of 156 ha (390 acres) in 1997 and 160 ha (399 acres) in 2006. Landowners without land enrolled in CRP owned an average of 112 ha (280 acres) both in 1997 and 2006.

In 1997, the most common answers for not enrolling eligible lands into CRP related to higher potential income from crops than CRP payments (68%) and increased crop prices (56%). In 2006, the most common reply for non-participation was ineligibility (41%) followed by the opportunity costs of growing crops (28%).

Landowners with CRP in 1997 indicated they enrolled land because of: a) concern for soil erosion (73%); b) provision of wildlife habitat (67%); c) most profitable use of land (52%); d) low risk associated with payments (36%); and e) easiest way to meet conservation compliance (36%). Personal retirement (15%), and reduced labor (15%) were secondary factors. Most landowners (73%) indicated their selection of a cover crop for CRP land was to benefit wildlife. In 2006, landowners indicated erosion (36%), conservation/buffer strips (33%), and wildlife (29%) as the most popular factors for program participation.

In 1997, 35% of landowners with CRP and 27% of landowners without CRP indicated wildlife was an important consideration in their choice of farming practices. By contrast, in 2006 94% of the participants considered wildlife as important when selecting a farming practice. As for 2006 non-participants, we found 67% considered wildlife as important when selecting a farming practice.

Most landowners with CRP in 1997 (93.7%) indicated that CRP improved pheasant habitat in the vicinity of their farm. The majority of landowners without CRP (70.5%) also indicated improved pheasant populations. A majority of all landowners (52%) indicated CRP improved habitat for white-tailed deer (*Odocoileus virginianus*) and gray partridge (*Perdix perdix*). Fewer landowners (38%) indicated CRP improved habitat for meadowlarks.

For 2006, 98% of all respondents agreed with the statement: "the CRP has improved the overall wildlife habitat in Minnesota." Moreover, 92% of those surveyed answered they agreed with the statement: "The CRP has improved wildlife habitat in your area." There were no significant differences between participants and non-participants. Again, pheasants (85%) and white-tailed deer (34%) were the two major perceived beneficiaries.

DISCUSSION

Landownership amounts between participants and non-participants did not change between 1997 and 2006. In 1997, the most common reasons for not enrolling were directly related to anomalously high prices for corn and soybeans. In 2006, ineligibility was the leading factor. This occurred after USDA tightened the criteria for CRP eligibility and made the program more competitive for the receipt of rental payments. On the national level, these changes favored the Great Plains states within the prairie pothole region.

The average size of CRP fields declined from 33 to 15 ha (82 to 37 acres) in south-central Minnesota. Interestingly, statewide aggregate acreage in 2006 was only about 40,000 ha (100,000 acres) below the late 1980s peak. However, CRP lands are presently more concentrated in the Red River valley in northwestern Minnesota (Lopez et al. 2000).

The most significant changes in landowner perception between 1997 and 2006 concern wildlife perceptions. In 1997 approximately one-third of the CRP participants indicated wildlife was important in farming considerations, increasing to 94% in 2006. A similar pronounced increase from 27% in 1997 to 67% in 2006 occurred with non-participants as well. This change is indicative of heightened environmental awareness of the CRP especially for and appreciation for pheasants and to a lesser extent, white tailed deer, but not for nongame species such as meadowlarks. Interestingly, meadowlarks have been found to sustain increased populations in areas with CRP grasslands (Kimmel et al. 1992).

Our findings paralleled a national study conducted by the USGS (2003) that examined CRP participants and their environmental perceptions of the program. This study found that in the Corn Belt 73% of landowners agreed that CRP had positive changes on wildlife and 59% agreed the program provided additional opportunities to view wildlife. Our 2006 survey found that 92% of our respondents (participants and non-participants) agreed with the statement that CRP "improved wildlife" in the local area and 98% to Minnesota at-large.

The USGS (2003) found that CRP was often viewed by participants as a source of weeds (33%) and attracted unwanted permissions for hunting (23%). Our 2006 survey found only 3% of all surveyed "strongly agreed" with these criteria, although approximately 30% "agreed" at a more moderate level. Consequently, landowners in south-central Minnesota mirrors the Corn Belt regional findings yet the intensity of these negative attributes is dissimilar.

The USGS (2003) survey also found that about 14% of the participants felt CRP added to an unkempt appearance. In our 2006 survey, the participants matched the USGS (2003) regional finding. However, approximately 25% of our non-participants felt CRP fostered an unkempt farm appearance. It is possible that the latter could be due to ignorance. Non-participants may recognize a CRP field as unorderly relative to the virtually manicured appearance of heavily cultivated corn and soybeans that dominate the regional landscape. Unlike, Reinvest in Minnesota (RIM) lands, CRP fields are typically not denoted by signage advertising the program.

Leistritz et al (2002) found that non-participants, (i.e. local leaders, agri-business professionals) in North Dakota felt CRP drained money from local economies because land taken out of production does not require the same amount of purchased inputs (fertilizers, insecticides, etc.) as cropland, and encouraged population loss through retirement and relocation elsewhere. Although we did not survey "local leaders" as defined by Leistritz et al. (2002), the majority of our non-CRP participants in 1997 (52%) felt the CRP was at least somewhat important in stabilizing rural incomes. In 2006, about 65% of our non-participants said the CRP was financially good for farmers. As for retirement and its perceived impact on population loss, our 1997 survey found retirement to be inconsequential when making a CRP decision. We did not survey for this criterion in 2006.

In summary, our most significant findings were: 1) in 2006, 98% of all landowners found that CRP benefited wildlife in Minnesota and that pheasants were the major beneficiaries, and 2) more landowners in 2006 than in 1997 considered wildlife populations when making farm related decisions. Our survey results paralleled the USGS (2003) regional findings, but with some qualifications. Overall, both the non-CRP and CRP participants find the CRP to be a popular program. Approximately 56% of those surveyed in 2006 would change absolutely nothing if given the chance to re-authorize the CRP, while the remaining 44% recommended only minor changes.

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ESTIMATING WHITE-TAILED DEER ABUNDANCE USING AERIAL QUADRAT SURVEYS

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SUMMARY OF FINDINGS

We estimated white-tailed deer (*Odocoileus virginianus)* abundance in select permit areas using stratified and 2-dimensional systematic quadrat surveys to evaluate the impact of deer season regulation changes on deer population levels and to recalibrate population models. Precision was similar between sampling designs when an adequate number of animals was observed. When few animals were observed, and their distribution was aggregated into relatively few clusters, precision of stratified surveys was poor. Understanding deer distribution across the landscape is critical to selecting an appropriate sampling design and obtaining accurate and precise abundance estimates.

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 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).

In Minnesota, white-tailed deer populations exceed management goals in many permit areas (PAs). A conventional approach of increasing the bag limit within the established hunting season framework has failed to reduce deer densities. As a result, the Department of Natural Resources is currently 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). In addition, wildlife managers in Minnesota's farmland zone have expressed concern regarding the accuracy of deer population estimates derived from simulation modeling (Osborn et al. 2003). Because population estimates are subject to drift as model input errors accumulate over time, periodic model recalibration is recommended (Grund and Woolf 2004). The objective of this study is to provide independent estimates of deer abundance in select PAs. These data will be used to evaluate the impact of deer season regulation changes on deer abundance and to recalibrate population models.

METHODS

We estimated deer populations in PAs using a quadrat-based, aerial survey design. Quadrat surveys have been used successfully 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. In PAs where the local wildlife manager had prior knowledge about deer abundance and distribution, we employed a stratified, random sampling design, with quadrats stratified into 2 abundance classes (high, low). In other areas, we employed a 2-dimensional systematic sampling design (Cressie 1993, D'Orazio 2003). Systematic designs are typically easier to implement, maximize sample distribution, and often result in estimates that are more precise than those obtained using 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. 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.

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However, a Cessna 182 airplane was used in 3 PAs dominated by intensive row-crop agriculture. To improve visibility, we completed surveys after leaf-drop by deciduous vegetation, and when snow cover measured at least 15 cm. A pilot and 2 observers searched for deer within each quadrat until they were confident all animals had been observed. We used a moving-map software program (DNR Survey) coupled to the aircraft global positioning system receiver to identify quadrat boundaries, guide quadrat navigation, and log deer locations and aircraft flight path. We estimated deer abundance from stratified surveys using SAS Proc SURVEYMEANS (SAS 1999) and from systematic surveys using formulas from D'Orazio (2003).

RESULTS AND DISCUSSION

We completed 5 surveys during January-February 2005, and 10 surveys during January-March 2006 (Table 1). Survey results from Carlos Avery Wildlife Management Area (PA 235) and St. Croix State Park will not be reported here because sampling design varied from that reported previously to account for the small geographic size of these 2 units.

Fixed-wing surveys were conducted in PAs 252, 421, and 423. In the latter 2 areas, population estimates were substantially lower than expected, based on long-term deer harvest rates. 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 low using fixed-wing aircraft, and/or 5) kill locations from hunter-killed deer were incorrect.

In terms of precision and relative error, systematic and stratified designs appear to provide similar results, with the exception of PAs 421, 423, and 201 (Table 1). In PA 421, all high strata quadrats were surveyed, resulting in a sampling variance of zero. In addition, because few deer were observed in low strata quadrats, sampling variance was low and, therefore, overall precision of the population estimate was high. 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 very poor. Few deer were observed during either survey (144 and 56, respectively). Most quadrats contained no deer, and nearly all observations occurred within 1 or 2 quadrats.

Clearly, understanding deer distribution across the landscape is critical to selecting an appropriate sampling design and obtaining accurate and precise abundance estimates. Over the next several months, we plan to complete survey analysis and make recommendations for next year's sampling protocol. Analysis will include *post-hoc* evaluation of habitat features present in quadrats containing deer. 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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Sampling design	Year	Permit area	Sampling rate (%)	Population estimate		CV (%)	Error (%)[a]	Density Estimate	
				N	90% CI			Mean	90% CI
Systematic	2005	252	16	2,999	2,034 – 3,969	19.5	32.2	2.9	2.0 – 3.8
		257	16	2,575	1,851 – 3,290	16.9	28.1	6.2	4.5 – 8.0
	2006	204	16	3,432	2,464 – 4,401	17.0	28.2	4.8	3.4 – 6.1
		209	17	6,205	5,033 – 7,383	11.4	18.9	9.7	7.9 – 11.6
		210	17	3,976	3,150 – 4,803	12.5	20.8	6.5	5.1 – 7.8
		256	17	4,670	3,441 – 5,899	15.9	26.3	7.1	5.3 – 9.0
		236	16	6,774	5,406 – 8,140	12.1	20.2	18.2	14.5 – 21.9
Stratified	2005	206	20	2,486	1,921 – 3,051	13.7	22.5	5.3	4.1 – 6.5
		342	20	3,322	2,726 – 3,918	10.8	17.7	9.5	7.8 –11.2
		421	20	631	599 – 663	3.0	5.0	0.8	0.8 – 0.9
	2006	201	20	274	100 – 449	37.6	61.9	1.7	0.6 – 2.8
		420	20	2,000	1,349 – 2,652	19.7	32.3	3.1	2.1 – 4.1
		423	20	472	179 – 764	37.4	61.5	0.9	0.3 – 1.4

Table 1. Deer population and density estimates derived from aerial surveys in Minnesota, 2005-2006.

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

EVALUATING ALTERNATIVE REGULATIONS FOR MANAGING WHITE-TAILED DEER IN MINNESOTA—A PROGRESS REPORT

Marrett D. Grund, Louis Cornicelli, David C. Fulton[1], Brian S. Haroldson, Emily J. Dunbar, Sonja A. Christensen[2], and Michelle L. Imes

SUMMARY OF FINDINGS

The increasing number of white-tailed deer (*Odocoileus virginianus*) in many deer permit areas of Minnesota is posing significant challenges to wildlife managers. Our primary objectives in this investigation are to: 1) quantify impacts of 3 alternative deer harvest regulations have on age and sex structures of hunter-killed deer and deer populations, and 2) measure hunter and landowner attitudes regarding alternative deer harvest regulations. We outline methods employed and progress made during the first year of the alternative deer management project. Over the past year, we accomplished all objectives defined in the project proposal and anticipate continued success during the upcoming year.

The increasing number of white-tailed deer (*Odocoileus virginianus*) in many deer permit areas of Minnesota is posing significant challenges to wildlife managers. Supply of antlerless permits offered to hunters exceeds demand, and desired annual antlerless harvests are frequently not achieved. In Minnesota, the primary approach for managing overabundant deer is through allocating bonus permits, which allows hunters to take 1-4 additional antlerless deer. Minnesota Department of Natural Resource (DNR) harvest data from the 2005 hunting season suggest bonus permits are not being used efficiently under the existing seasonal framework. During 2005, 72% of successful hunters killed 1 deer, 21% of successful hunters killed 2 deer, and 7% of hunters killed >2 deer. Allowing hunters to harvest >1 deer has little impact on the total numerical harvest, because the regulation only affects about 1 out of 4 successful hunters.

Alternative harvest strategies that emphasize harvesting antlerless deer during the hunting season may increase both number and proportion of adult females in the overall harvest. Increased harvest of adult females would reduce deer densities in areas where traditional harvest strategies using bonus permits have not been successful. Our primary objectives were to: 1) quantify impacts of 3 alternative deer harvest regulations on age and sex structures of hunter-killed deer and deer populations, and 2) measure hunter and landowner attitudes regarding alternative deer harvest regulations.

STUDY AREAS

For the most part, this study is being conducted in Minnesota's transition zone. The transition zone is a loosely defined region between Minnesota's forest and farmland zones. The zone extends from northwest to southeast Minnesota and primarily encompasses hunting zones 2 and 3. Virtually all deer permit areas in hunting zones 2 and 3 allowed bonus tags in 2005. We originally proposed 3 blocks of deer permit areas to evaluate an early antlerless-only hunting season (Figure 1). However, an early antlerless-only season has not yet been adopted by DNR in the central study area. We are currently evaluating earn-a-buck and antler-point restriction regulations in 7 state parks distributed throughout Minnesota.

METHODS

General Hunter Survey

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At the time of license purchase, hunters were asked where they intend to hunt most often and those data were retained in an electronic license system (ELS) database. We spatially-stratified our study area into 4 groups (Figure 2), which were based primarily on the Minnesota ecological classification system. Hunters were selected at random from the ELS database. In total, 1,500 surveys were sent to hunters in each of the 4 groups, yielding a total sample size of 6,000.

The survey contained 4 sections. The first section contained questions to assess 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 experiences. In the final section, hunters were presented with 5 scenarios related to Minnesota deer management. In total, there were 7 choices within each scenario but hunters were only given 3 choices (at random), which they were asked to rank (preference 1, 2, 3). While each choice was assigned at random, the same number of total choices was 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 option of not hunting or moving to another area were offered as a choices.

The initial mailing was conducted on 15 October 2005. Second and third mailings to non-respondents were conducted on 15 November 2005 and 15 December 2005, respectively.

Check Station Operations

In Minnesota, successful hunters were required to register each deer harvested within 24 hours of the close of the deer-hunting season. Based on historical registration data and in consultation with DNR Area Wildlife Managers, we identified 40 registration stations most likely to register the maximum number of deer within or near 1 of our study areas. We trained approximately 150 college students and DNR staff to sex and estimate age classes of deer (fawn, yearling, adult) based on tooth replacement and wear (Severinghaus 1949) from jaws viewed *in situ*. Primary incisors were removed from all deer having bicuspid third premolars so that age-at-death could be estimated by year using cementum annuli techniques. Antler characteristic data were also obtained from antlered deer.

Study Area Hunter Survey

Hunters participating in 1 of our treatment hunts were identified through the ELS database. We also identified hunters declaring to hunt in nearby deer permit areas to serve as a control group. Identical to the aforementioned choice survey, hunters were randomly selected from this population to be surveyed. Sample sizes differed among treatment groups and were dependent on numbers of hunters participating within a particular hunting regulation. A total of 3,629 hunters were randomly selected to receive this survey.

The survey contained 3 sections. The first section contained questions to determine where hunters hunted in each hunting season. The next section of questions was designed to determine hunting techniques, hunter behavior, and hunting motivation. The final section of questions focused on hunting experiences and support for deer hunting regulations after the hunter had experienced hunting under the regulation.

The initial mailing was conducted on 6 March 2006. Second and third mailings to non-respondents are planned for April and May 2006.

Deer Population Monitoring

Aerial Surveys.--Deer populations were estimated from the air using helicopter quadrat surveys. Each deer permit area was divided into 2.6-km^2

quadrats (sections from Public Land Survey data). A twenty percent sample of these quadrats was surveyed using either a 2-dimensional systematic random sampling design (Cressie 1993, D'Orazio 2003), or a stratified random sampling design. Surveys were conducted when approximately 100% of the ground was covered with snow and was anticipated to last several days. Complete snow cover improved visibility and ensured that enough time was available to allow the survey to be completed. Quadrats were flown until observers were confident they had seen all deer within each quadrat. Density estimates were calculated using standard formulas (Hayek and Buzas 1997).

Ground Surveys.--Deer populations were estimated from the ground using spotlight quadrat surveys. Similar to aerial surveys, deer permit areas were partitioned using Public Land Survey Data and 20% of the quadrats were selected using a stratified random sampling design. Roads adjacent to selected quadrats served as transects for ground surveys. The field season for conducting ground surveys is 1 April 2006 through 15 May 2006, or until all selected quadrats are searched. The surveys began approximately 30 minutes prior to sunset and continued for approximately 4 hours. During surveys, 2 observers searched for deer using hand-held spotlights while a pickup truck traveled at speeds of 24–32 km/hour. Observers determined distance to centers of deer clusters (i.e., groups) with a laser range finder, and determined angles to centers of clusters using a prismatic compass. Geographic positioning system (GPS) units were used to facilitate locating transects in the field and to monitor locations of observers throughout the survey. Clusters were separated using nearest neighbor criterion (LaGory 1986), location of deer, and their behavior. In general, a group of deer behaving similarly in close proximity to each other e.g., traveling together in a field) was considered a cluster.

Vegetation Surveys

Vegetation sampling was conducted in Itasca State Park, MN from 14 July – 21 July 2005. Itasca State Park was divided into a 16 x 16 grid. Three sampling plot arrays were selected using a random number generator. Each sampling plot array contained a 50-m^2 subplot and 4, 1-m^2 subplots nested within the 250-m^2 plot (Figure 3). Plots were permanently marked by hammering 0.6-m pieces of rebar at the center and at each corner of the 250-m^2 sampling plot, at each corner of the 50-m^2 subplot, and at 1 pair of diagonal corners of each 1-m^2 subplots.

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. At each corner of the 250-m^2 plot, all trees (> 1.5-m tall and/or between 2.54 and 12.7 cm dbh) within a 6-m radius of the permanent marker were identified to species, and height and dbh recorded. Trees were also recorded as dead or alive.

At each permanent marker of the 50-m^2 subplot, trees and shrubs (> 1.5-m) were sampled within a 2-m radius. A tally of living and dead trees, according to species and height classes, was recorded. A count of shrubs according to species and height classes was also recorded.

In each 1-m^2 subplot, percent cover of all woody and herbaceous species (<2.54 dbh and <1.5-m tall) was recorded using Daubinmier cover classes. We also recorded percent cover of bryophytes and lichens, tree seedlings, and rock and litter. The height of each woody or herbaceous plant was also recorded. An estimate of understory cover was measured using a density board and recording the number of squares obscured at eye level in each cardinal direction. Litter depth was measured and recorded. Percent overstory cover was estimated using a spherical densitometer at the center of the subplot and a densitometer at 5 5-m intervals along transects in each cardinal direction from the subplot center. Browsing intensity was recorded for each plant and was based upon percent of stems browsed and height of plant. The number of sterile and flowering or fruiting Canada mayflower (*Maianthemum canadense*) was also recorded. Photographs were taken above each plot and also in each cardinal direction to record forest structure.

RESULTS

General Hunter Survey

After 3 mailings, we achieved a response rate of approximately 60% (Table 1). Analysis is planned for May 2006 with results available in summer 2006.

Check Station Operations

Staff examined 3,492 male and 2,230 female deer at registration stations during fall 2005. Including both genders, there were 1,322 deer aged as fawns. Antler characteristic data were recorded from 2,625 deer. We sent 2,448 primary incisors to Matson's Laboratory (Milltown, Mont., U.S.) for cementum annuli aging.

Study Area Hunter Survey

The initial mailing of the survey was underway in April 2006. No data were available at the time of this writing.

Aerial Surveys

Results from the aerial survey can be reviewed in Haroldson et al. (2005).

Ground Surveys

Ground surveys began on 1 April 2006 in northwestern deer permit areas and on 3 April 2006 in the north-metro deer permit areas (Figure 1). Only 13% of the surveys were complete when this report was written. Therefore, no results are presented.

Vegetation Surveys

Data obtained from vegetation surveys have been entered into a database. No analyses have been performed because these data will be collected and analyzed across years.

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Table 1. Survey mailing dates and return rates associated with the general hunting survey conducted in fall 2005, Minnesota.

Mailing	Date	Total Returned	Response Rate
First	15 Oct 2005	1,543	26.5
Second	15 Nov 2005	2,542	43.7
Third	15 Dec 2005	3,331	59.8

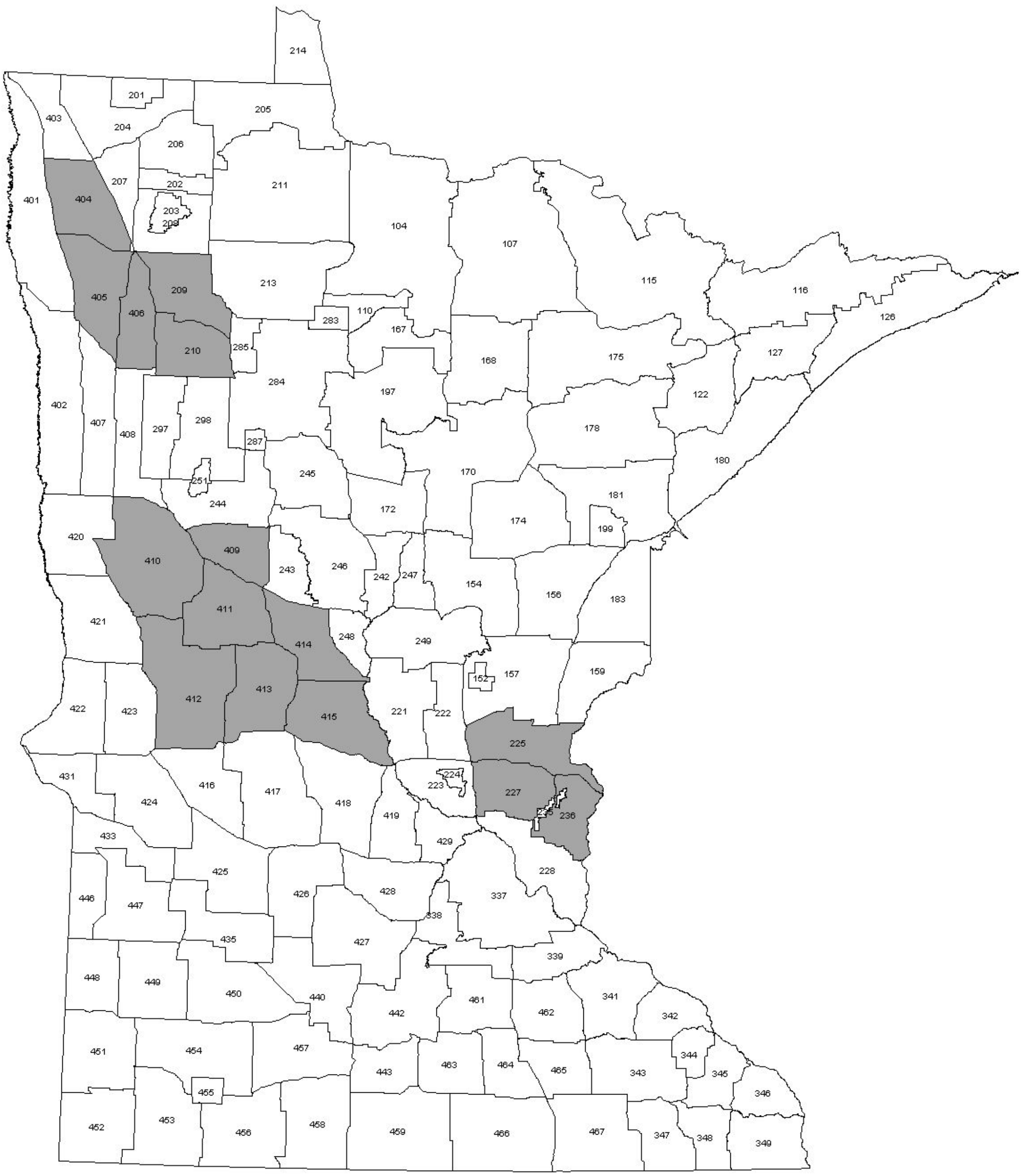

Figure 1. Blocks of deer permit areas where October antlerless-only seasons were proposed for evaluation as part of the alternative deer management project, Minnesota, 2005-2010.

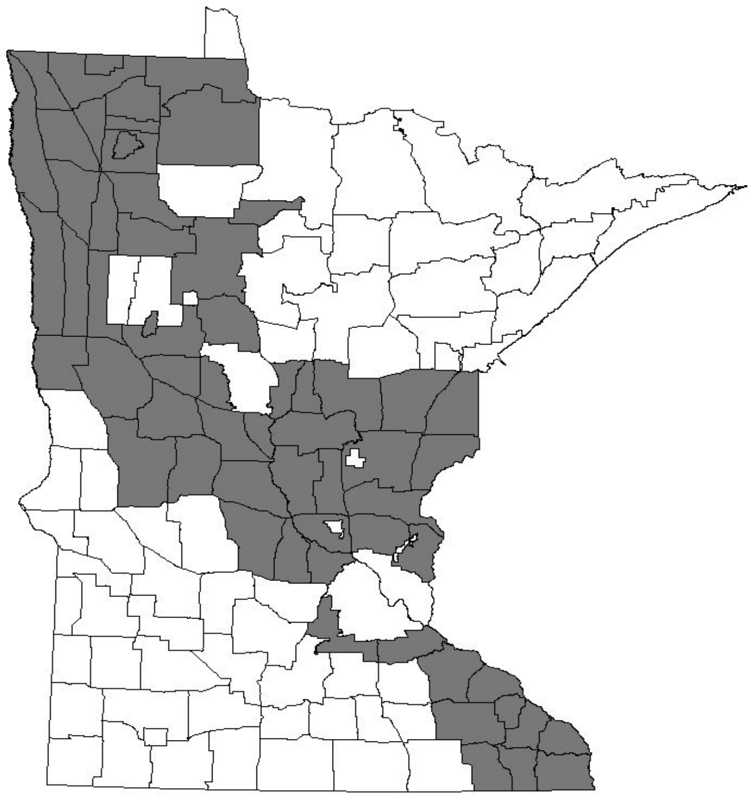

Figure 2. Surveys were sent to hunters declaring to hunt in shaded deer permit areas for the general hunter survey, Minnesota, 2005.

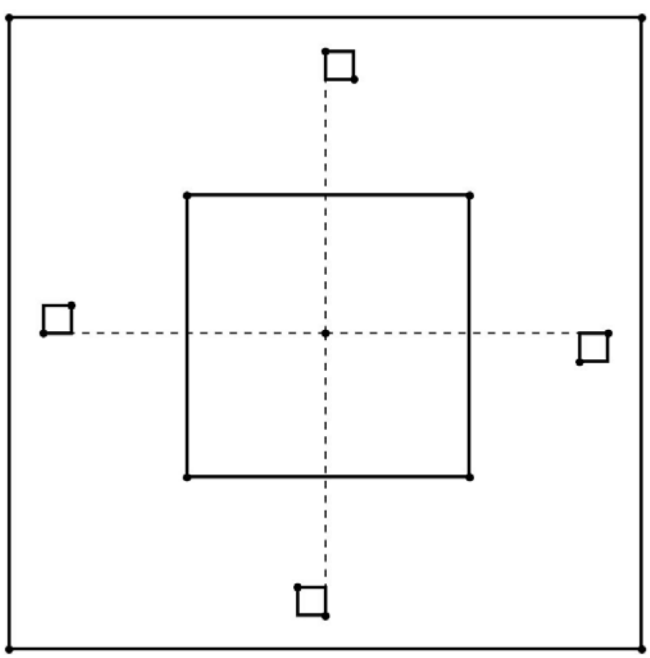

Figure 3. Design of a sampling plot array used at Itasca State Park, Minnesota, 2005. Dots indicate locations of 17 permanent markers.

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 is 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 2003-2005, 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 at least 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.

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 often inundated by snow. 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.

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 (Figure 1).

We estimated amounts of winter cover, winter food, and reproductive cover on each study area by cover mapping to a GIS from 2003 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, 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°C [32°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. GPS receivers were used to record the time and position of observers 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 318 patches of winter cover on the 36 study areas and surrounding 3.2-km (2-mile) buffers. Severity of winter weather was low during all 4 winters (2002-06) 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 2005 surveys

Observers completed all 360 scheduled surveys (10 repetitions on 36 study areas) during the spring 2005 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, or frost). Over all regions, 91% of the surveys were started with at least light dew present, which was much greater than 2004 (78%) and 2003 (84%). However, only 60% of surveys were started under clear to partly cloudy skies (<60% cloud cover), and only 38% reported wind speeds <6 km/hour (4 miles/hour). Seven percent of surveys were started on mornings with wind >16 km/hour (10 miles/hour), and 11% were started with temperatures <0^oC (32^oF). Among regions, Glenwood experienced the least dew (17% of surveys started with no dew), the most wind (16% of surveys started with wind speed >16 km/hour [10 miles/hour]), and the greatest cloud cover (50% of surveys started with cloud cover ≥60%).

Pheasants were observed on all 36 study areas during spring 2005, but abundance indices varied widely among areas from 15.0–293.7 pheasants observed per route (Table 1). Over all study areas, the mean pheasant index was 104.9 birds/route, a nonsignificant change from spring 2004 (Table 2). Total pheasants/route varied among regions from 57.3 in the Faribault region to 167.6 in the Windom region (Table 2). Compared to 2004, total indices changed significantly only in the Faribault region, where they decreased 28% (95% CI: –3 to –53%).

Hens were relatively abundant among study areas in spring 2005. The overall hen index averaged 58.3/route, a nonsignificant change from 2004 (Table 2). Among regions, the hen index ranged from 23.8/route in Faribault to 102.6/route near Windom. Hen indices were not significantly different from 2004 in any region (Table 2). The observed hen:rooster ratio varied from 0.3 to 2.9 among study areas (Table 1). Fewer hens than roosters were observed on 3 study areas in the Marshall region, 4 areas in Glenwood, and 7 areas in Faribault.

Summer 2005 surveys

Observers completed 359 of the 360 surveys during the summer 2005 season. Weather conditions during the summer surveys ranged from excellent (calm, clear sky, heavy dew) to poor (light or no dew, overcast sky, or rain). Over all regions, 81% of the surveys were started with medium-heavy dew present, which was lower than 2004 (87%) and equal to 2003 (81%). Sixty-six percent were started under clear skies (<30% cloud cover), and 69% reported wind <6 km/hour (4 miles/hour). In comparison, 91% of the statewide August Roadside Surveys were started under medium-heavy dew conditions, 84% under clear skies, and 71% with winds <6 km/hour (4 miles/hour). The less desirable weather conditions reported in this study probably reflects the limited availability of 10 suitable survey days within the 31-day period.

Pheasants were observed on all 36 study areas during 2005, but abundance indices varied widely from 2.5–372.3 pheasants observed per route (Table 3). Over all study areas, the mean pheasant population index was 150.9 birds/route, an 82% (95% CI: 49–115%) increase from 2004. Total pheasant indices varied among regions from 90.5 birds/route in the Faribault region to 190.5 birds/route in Marshall (Table 4). Compared to 2004, total indices increased significantly in the Marshall, Glenwood, and Faribault regions, but not Windom (Table 4).

The overall hen index (26.3 hens/route) increased 63% (95% CI: 15–111%) from last year, and varied among regions from 14.8 in the Faribault region to 37.4 near Windom (Table 4). Hen indices increased 64% (95% CI: 5–123%) in the Glenwood region, but were not significantly higher than 2004 in the Marshall, Faribault, or Windom regions (Table 4). In contrast, overall and regional cock indices fell to their lowest levels in the 3-year study (Table 4), but declines from last year were significant only in the Windom (95% CI: –23 to –53%) and Faribault regions (95% CI: –8 to –52%). The observed hen:rooster ratio varied from 0.0 to 8.0 among study areas (Table 3), and averaged 2.8 overall. Fewer hens than roosters were observed on 1 study area in the Glenwood and Windom regions and 2 areas in the Faribault region.

The 2005 overall brood index (23.6 broods/route) increased 102% (95% CI: 63–141%) from 2004, with regional indices ranging from 12.6 in Faribault to 35.0 in Marshall (Table 4). Regional brood indices increased significantly in all regions except Windom (Table 4). Mean brood size averaged 5.1 chicks/brood overall, but varied among regions (4.2 in Marshall, 6.1 in Glenwood, 5.0 in Windom, and 5.5 in Faribault). Mean brood size in 2005 increased over that in 2004 in the Glenwood and Faribault regions, declined in Marshall, and was unchanged in Windom (Table 4). On average, 55.3 broods were observed for every 100 hens counted during spring surveys, a 207% (95% CI: 127–287%) increase from last year. This brood recruitment index (broods/100 spring hens) varied among regions from 30.2 in Windom to 77.2 in Marshall. Brood recruitment indices increased significantly in all regions except Windom (Table 4).

Habitat associations

The mean pheasant index (total pheasants/route averaged over summer 2003–2005) was positively related to the cover index in all regions except Glenwood (Figure 2). Cover index explained 42% of the variation in pheasant indices in the Marshall region, 34% in Windom, 13% in Faribault, and 0% in Glenwood.

DISCUSSION

A high spring hen population in 2005 was expected given the mild winter of 2004-05 (the 4th consecutive mild winter). Furthermore, warm weather during the reproductive period was apparently conducive for increased nest success as the proportion of spring hens in 2005 that successfully recruited a brood into the summer population was twice that of 2004. Furthermore, average brood size increased significantly. Thus, the summer 2005 pheasant index was 82% above the 2004 index.

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 (2006-07) is severe, we may consider extending the study. However, the potential loss of two-thirds of CRP contracts expiring during 2007-09 will confound our ability to estimate winter cover needs.

We plan to continue to survey pheasant populations during spring and summer 2006-07. 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 2005.

			Birds/route[a]			
Region	Study area	n	Total	Cocks	Hens	F:M ratio
Marshall	1	10	133.3	61.0	72.3	1.2
	2	10	103.3	53.8	49.6	0.9
	3	10	184.5	88.8	95.6	1.1
	4	10	172.0	55.5	116.5	2.1
	5	10	45.8	25.0	20.8	0.8
	6	10	164.2	57.5	106.6	1.9
	7	10	85.5	35.9	49.5	1.4
	8	10	71.3	36.6	34.7	0.9
	9	10	33.3	14.9	18.4	1.2
Glenwood	10	10	47.0	28.0	19.0	0.7
	11	10	43.2	19.5	23.7	1.2
	12	10	142.9	72.9	70.0	1.0
	13	10	61.7	33.0	28.7	0.9
	14	10	66.7	32.5	34.2	1.1
	15	10	205.6	91.2	114.4	1.3
	16	10	56.2	35.2	21.0	0.6
	17	10	22.3	14.0	8.3	0.6
	18	10	114.8	35.6	79.2	2.2
Windom	19	10	293.7	75.3	218.4	2.9
	20	10	232.0	113.4	118.6	1.0
	21	10	120.1	44.6	75.5	1.7
	22	10	225.6	93.9	131.8	1.4
	23	10	228.7	105.9	122.8	1.2
	24	10	119.0	43.5	75.5	1.7
	25	10	130.8	43.0	87.9	2.0
	26	10	110.5	43.9	66.7	1.5
	27	10	47.8	21.7	26.1	1.2
Faribault	28	10	118.9	52.8	66.0	1.3
	29	10	92.2	54.4	37.9	0.7
	30	10	32.3	18.5	13.7	0.7
	31	10	65.7	49.0	16.7	0.3
	32	10	66.1	35.6	30.5	0.9
	33	10	42.2	31.9	10.3	0.3
	34	10	48.2	30.3	18.0	0.6
	35	10	34.8	21.4	13.4	0.6
	36	10	15.0	7.5	7.5	1.0

[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 2003–2005.

			Birds/route[a]			% change	
Region	Group	n	2003	2004	2005	2004-2005	95% CI
Marshall	Total pheasants	9	87.2	116.3	110.4	8	±35
	Cocks	9	43.1	47.4	47.7	11	±33
	Hens	9	44.1	68.9	62.7	8	±44
Glenwood	Total pheasants	9	100.9	113.0	84.5	-10	±30
	Cocks	9	48.7	47.2	40.2	3	±36
	Hens	9	52.2	65.9	44.3	-20	±28
Windom	Total pheasants	9	162.3	179.7	167.6	3	±23
	Cocks	9	69.4	75.8	65.0	-11	±16
	Hens	9	92.9	103.9	102.6	19	±37
Faribault	Total pheasants	9	70.3	86.0	57.3	-28	±25
	Cocks	9	37.1	47.1	33.5	-28	±16
	Hens	9	33.2	38.8	23.8	-18	±46
All	Total pheasants	36	105.2	123.8	104.9	-7	±13
	Cocks	36	49.6	54.4	46.6	-6	±12
	Hens	36	55.6	69.4	58.3	-3	±18

[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 2003–2005.

			Birds/route[a]			% change	
Region	Group	n	2003	2004	2005	2004-2005	95% CI
Marshall	Total pheasants	9	142.6	114.9	190.5	119	±95
	Cocks		12.7	13.5	10.5	15	±61
	Hens		25.6	20.5	32.3	168	±190
	Broods		22.3	16.8	35.0	172	±122
	Chicks/brood		4.6	4.8	4.2	–10	±7
	Broods/100 spring hens		59.9	29.8	77.2	260	±246
Glenwood	Total pheasants	9	139.9	57.9	135.7	140	±87
	Cocks		9.2	8.3	8.0	24	±48
	Hens		23.5	12.3	20.7	64	±59
	Broods		20.2	8.3	17.2	122	±103
	Chicks/brood		5.0	4.1	6.1	38	±18
	Broods/100 spring hens		44.7	14.7	42.8	240	±146
Windom	Total pheasants	9	283.5	180.1	187.0	9	±38
	Cocks		25.9	23.6	13.8	–38	±15
	Hens		50.9	36.3	37.4	3	±32
	Broods		36.2	24.2	29.4	29	±48
	Chicks/brood		5.4	5.0	4.6	–8	±11
	Broods/100 spring hens		47.1	29.1	30.2	35	±78
Faribault	Total pheasants	9	164.6	54.4	90.5	60	±29
	Cocks		9.5	13.0	8.0	–30	±22
	Hens		23.6	13.1	14.8	16	±24
	Broods		23.6	6.8	12.6	85	±20
	Chicks per brood		5.5	5.0	5.5	23	±22
	Broods/100 spring hens		85.4	18.6	71.0	293	±175
All	Total pheasants	36	182.6	101.8	150.9	82	±33
	Cocks		14.3	14.6	10.1	–7	±19
	Hens		30.9	20.5	26.3	63	±48
	Broods		25.6	14.0	23.6	102	±39
	Chicks/brood		5.1	4.7	5.1	10	±9
	Broods/100 spring hens		59.3	23.1	55.3	207	±80

[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 2005.

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	174.8	13.1	27.5	2.1	134.2	29.7	4.5	108.2	41.1
	2	9	189.8	6.9	34.7	5.0	148.1	39.8	3.7	114.7	80.3
	3	10	101.9	14.6	18.4	1.3	68.9	14.6	4.7	78.9	15.2
	4	10	258.0	9.5	56.5	5.9	192.0	46.0	4.2	81.4	39.5
	5	10	156.7	12.1	27.1	2.2	117.5	35.0	3.4	129.2	168.0
	6	10	302.8	8.5	50.9	6.0	243.4	55.7	4.4	109.3	52.2
	7	10	145.5	6.4	25.5	4.0	113.6	27.3	4.2	107.1	55.0
	8	10	274.0	14.0	32.0	2.3	228.0	48.0	4.8	150.0	138.5
	9	10	111.4	9.6	18.4	1.9	83.3	19.3	4.3	104.8	104.8
Glenwood	10	10	133.0	3.0	15.0	5.0	115.0	14.0	8.2	93.3	73.7
	11	10	53.4	8.5	10.2	1.2	34.7	7.6	4.6	75.0	32.1
	12	10	167.6	5.7	28.6	5.0	133.3	21.9	6.1	76.7	31.3
	13	10	113.9	6.1	17.4	2.9	90.4	17.4	5.2	100.0	60.6
	14	10	201.8	7.5	25.9	3.5	168.4	29.8	5.6	115.3	87.2
	15	10	223.3	8.4	38.1	4.6	176.7	28.8	6.1	75.6	25.2
	16	10	65.7	11.0	11.0	1.0	43.8	8.6	5.1	78.3	40.9
	17	10	2.5	2.5	0.0	0.0	0.0	0.0	.	.	0.0
	18	10	260.2	19.4	39.8	2.0	200.9	26.9	7.5	67.4	33.9
Windom	19	10	175.8	18.4	36.3	2.0	121.1	26.3	4.6	72.5	12.0
	20	10	259.6	11.4	65.4	5.7	182.8	54.0	3.4	82.5	45.5
	21	10	202.1	9.5	43.2	4.6	149.5	33.7	4.4	78.0	44.6
	22	10	125.5	17.6	30.2	1.7	77.6	19.0	4.1	62.7	14.4
	23	10	372.3	18.8	73.3	3.9	280.2	57.4	4.9	78.4	46.8
	24	10	96.0	14.0	16.0	1.1	66.0	14.0	4.7	87.5	18.5
	25	10	180.4	11.7	32.2	2.8	136.4	22.4	6.1	69.6	25.5
	26	10	249.1	14.9	36.0	2.4	198.2	34.2	5.8	95.1	51.3
	27	10	22.6	7.8	4.3	0.6	10.4	3.5	3.0	80.0	13.3
Faribault	28	10	110.4	13.2	20.8	1.6	76.4	20.8	3.7	100.0	31.4
	29	10	57.3	10.7	3.9	0.4	42.7	5.8	7.3	150.0	15.4
	30	10	95.2	4.4	12.5	2.8	78.2	11.3	6.9	90.3	82.4
	31	10	84.3	11.8	16.7	1.4	55.9	11.8	4.8	70.6	70.6
	32	10	82.2	5.1	20.3	4.0	56.8	15.3	3.7	75.0	50.0
	33	10	179.9	3.5	28.2	8.0	148.1	23.8	6.2	84.4	230.2
	34	10	163.2	11.8	22.4	1.9	128.9	20.2	6.4	90.2	112.2
	35	10	20.4	6.2	2.7	0.4	11.5	1.8	6.5	66.7	13.2
	36	10	21.7	5.0	5.8	1.2	10.8	2.5	4.3	42.9	33.3

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

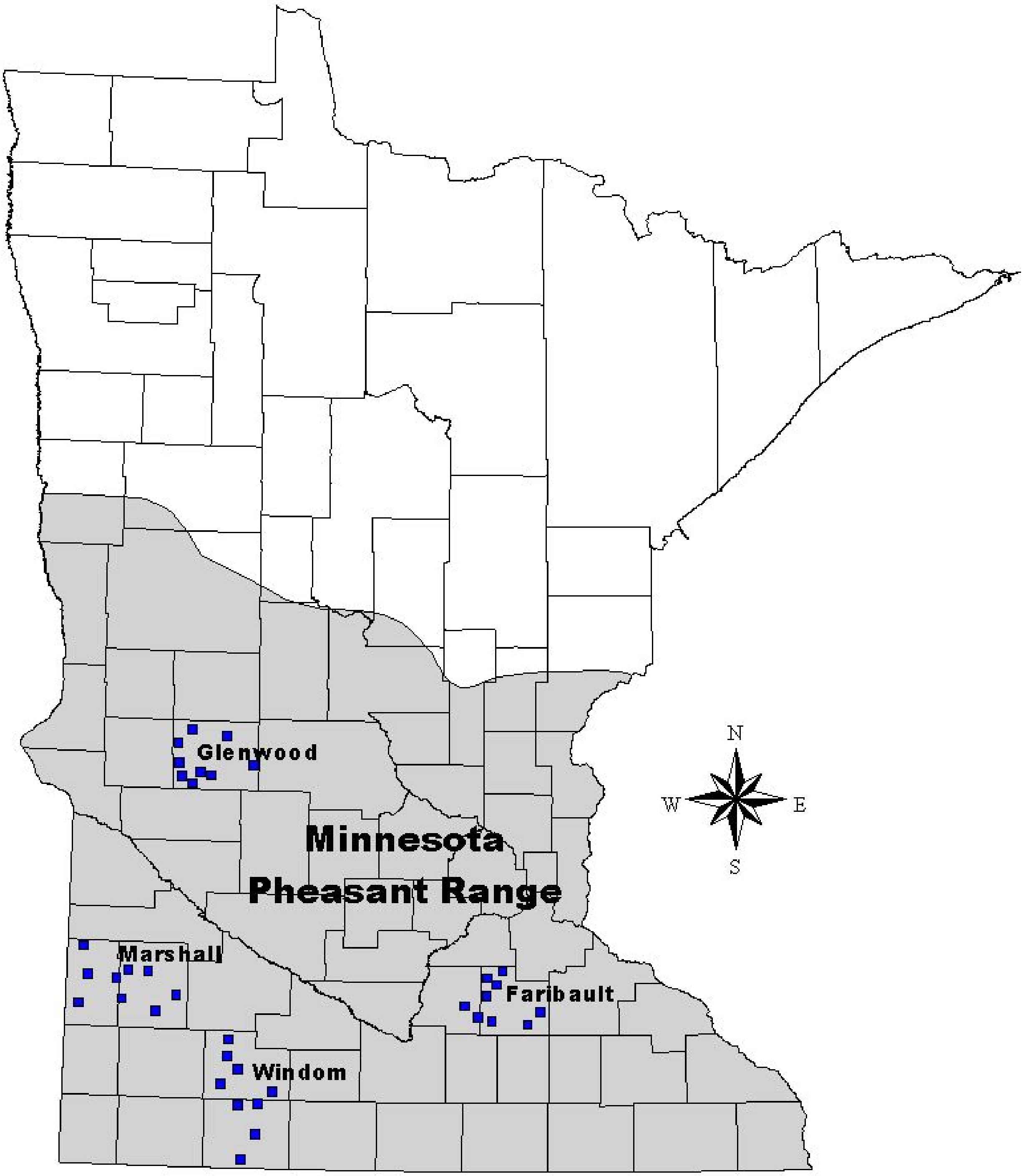

Figure 1. Locations of winter-habitat study areas within Minnesota's pheasant range, 2003-2005.

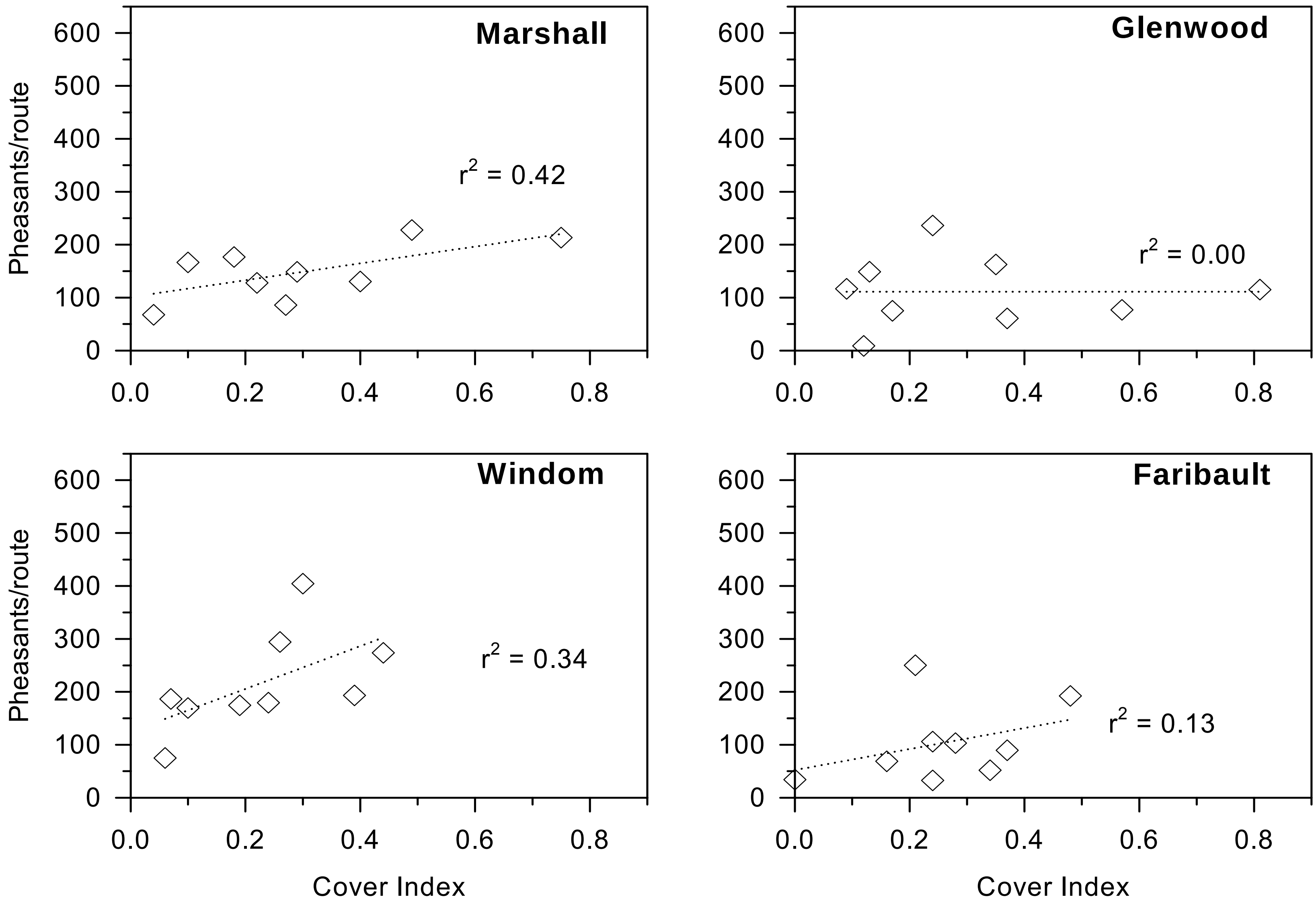

Figure 2. Relationship between relative pheasant abundance (pheasants counted/route) and amount of habitat (cover index) on 9 study areas in 4 regions in Minnesota during summer 2003-05. Route length was standardized to 161 km (100 miles).

2005 MINNESOTA SPRING TURKEY HUNTER AND LANDOWNER SURVEY

Allison M. Boies[1], Sharon L. Goetz, Richard O. Kimmel, and John D. Krenz

SUMMARY OF FINDINGS

Increased spring wild turkey (*Meleagris gallopavo)* hunter densities have resulted in concerns regarding hunt quality, hunter safety, and landowner tolerance of turkey hunters. This study assesses hunter satisfaction and landowner attitudes at current spring turkey hunter densities in Minnesota. A spring turkey hunter and landowner survey was conducted in 10 hunting permit areas (PAs) during the 2005 season to evaluate hunter satisfaction and landowner attitudes about turkey hunters at varying hunter densities. Spring 2005 surveys showed overall landowner attitudes were positive, and most hunters found it easy to gain access to private land. Interference by hunters or other individuals was infrequent. Based on hunter satisfaction and landowner attitudes, 2005 results showed hunt quality was high at a hunter density of 0.63 hunters/km^2 (1.62 hunters/mi^2) of huntable habitat. After completion of the spring 2006 hunter and landowner survey in 10 additional PAs, we will conduct further analysis to determine the relationship between hunter density, landowner attitudes, and hunter interference.

INTRODUCTION

It is important to carefully allocate permit numbers to ensure hunter safety, limit hunter access problems, ensure landowner and hunter satisfaction, maintain hunt quality, and best manage the wild turkey population. Kimmel (2001) noted that season management strategies in Minnesota initially restricted numbers of hunting permits to protect developing wild turkey populations. Currently, permit numbers are restricted to ensure hunt quality. Interference and hunting access are the most important factors that define a high-quality hunt (Smith et al. 1992). Dingman (2006) found that current hunter interference levels were shown to not significantly affect hunter satisfaction. Managers in southeastern Minnesota have expressed concern that increasing hunter densities would impact landowner tolerance of turkey hunters, which could lead to hunting access issues (G. Nelson, Minnesota Department of Natural Resources, personal communication).

For the spring 2005 turkey hunting season PA 343 had the highest hunter density at 0.63 hunters/km^2 of huntable habitat (forested areas with a 50 m buffer; 0.95 hunters/km^2 of forested habitat). Kubisiak et al. (1995) found that increasing hunter densities in southeastern Wisconsin to 1.16 hunters/km^2 (3.0 hunters/mi^2) of forested habitat had little impact on either hunters or landowners. Subsequently, Wisconsin Department of Natural Resources has increased hunter densities to 2.3 hunters/km^2 (>6 hunters/mi^2) of forested habitat in some areas (K. Warnke, Wisconsin Department of Natural Resources, personal communication). Hunter interest groups, in particular the Minnesota Chapter of the National Wild Turkey Federation, are aware of higher turkey hunter densities in Wisconsin and are requesting that the Minnesota Department of Natural Resources increase spring wild turkey hunting permit numbers. The goal of the first year of this 2-year study was to collect data to evaluate hunter access, safety, interference, and hunt quality on 10 PAs. Data from this survey will be used to determine relationships between hunter density and other variables such as hunter interference and landowner attitudes.

METHODS

Permit Area Selection

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We selected 10 PAs that had a range of hunter densities for the 2005 hunter and landowner surveys (Figure 1). These PAs included the highest hunter densities found in Minnesota during the spring 2005 turkey hunting season. Sampling criteria required selected PAs to contain more than 15 permits per hunting time period, be located in south-central or southeastern Minnesota, and contain a range of hunter densities.

Hunter and Landownctioner Selection

Hunters were randomly selected using Minnesota's Electronic License System database of spring turkey hunting permit recipients. Hunters were only sampled from the first 6 hunting time periods due to an unrestricted archery turkey hunting season during the last 2 time periods.

A sample of landowners was drawn from each selected PA using a database developed from county tax parcel data. Criteria for surveyed landowners included: ownership of at least 100 acres of land that intersects huntable turkey habitat, parcels located outside of city limits, and exclusion of non-agricultural businesses and organizations. Each parcel was evaluated with ArcView (Environmental Systems Research Institute, Redlands, CA, USA). County parcel shapefiles (taxpayer address, parcel size, and parcel location) were obtained from county tax role data. A huntable turkey habitat shapefile was used to determine location of wild turkey habitat in each selected PA (Ramseth 2004). A city limit shapefile that identifies subdivisions and limits was also obtained for each county. The shapefiles were all projected in UTM zone 15 coordinate system (Manual 1) from Lambert Conformal Conic.

County parcel shapefiles were queried to eliminate parcels of land that were less than 100 acres in size or that fell within city limits. Parcels of land that intersected the shapefile of huntable turkey habitat in each PA were selected in ArcView. The resulting database file was then exported to Microsoft Excel and queried by name and address to eliminate duplicate records, government entities, and out-of-state mailing addresses.

Survey Methodology

The hunter survey instrument evaluated hunter satisfaction at varying hunter densities. The survey consisted of questions regarding hunter success, access, satisfaction, number of days hunted, time period, and interference from other hunters (Appendix A). For the spring 2005 wild turkey hunter survey, 2,144 surveys were mailed to a sample of turkey hunt permit holders in 10 PAs (Figure 1). The selected hunters were mailed a survey and return envelope on the last day of the last time period of the spring turkey hunting season, (27 May 2005). A second and third mailing were then sent to non-respondents at 3-week intervals (20 June 2005 and 12 July 2005).

The landowner survey instrument evaluated landowner attitudes about hunters at various hunter densities. The survey contained questions regarding landowner attitudes about allowing access for spring turkey hunting, trespass, and the number of hunters requesting permission (Appendix B). For the spring 2005 landowner survey, 2,077 surveys were mailed 5 days after the close of the hunting season to landowners in 10 PAs randomly picked from all landowners meeting selection criteria. Selected landowners were sent a survey and a return envelope on 1 June 2005. Three additional mailings were sent to non-respondents at 4 and 5-week intervals (29 June 2005, 5 August 2005, and 3 September 2005).

RESULTS

We received a response rate of 74% for the hunter survey. The average number of turkeys seen by hunters was 21.6. The average number of turkeys shot at was 0.8. Hunters were more successful at harvesting turkeys in the morning (81%) than in the afternoon (19%). A total of 38% of hunters were successful at harvesting a turkey.

The majority of hunters hunted on private land (75%) and of these hunters, an average of 0.66 landowners refused access. Access to a hunting location was reported as either extremely easy (42%) or somewhat easy (38%) for the majority of hunters (Figure 2). Overall, 98% of hunters felt other hunters did not put them in danger at any time while hunting.

Overall, 91% (1,403) of hunters saw 0-2 hunters that were not part of their own hunting group. The rate of interference from other hunters was 13% (Figure 3), and 10% from non-hunters. Interference rates from other hunters in all the PAs were below 21% (Table 1). Eighty-four percent (1,261) of turkey hunters rated hunt quality average or above average (Figure 4).

We received a response rate of 64% for the landowner survey. The top 2 reasons for landownership were farming and preserving the land for the future. Ninety-seven percent of landowners reported they did not lease their land for spring turkey hunting. Overall, 65% of landowners reported seeing turkeys on their land in the past year.

Ninety-five percent of landowners did not personally hunt turkeys on their land during spring 2005. Overall, 36% of landowners were asked for permission to hunt their land by each of the following groups: family (450), acquaintances (415), and strangers (310; Figure 5). Thirty-one percent of landowners did not allow any hunters to hunt their land from the following groups: family (388), acquaintances (358), and strangers (208; Figure 6). Landowners who allowed 1 or more hunters on their property were more likely to allow friends or family (38.4%) compared to acquaintances (37.8%) or strangers (19.3%; Figure 6).

The majority (71%) of landowners reported that the number of hunters asking permission to hunt stayed the same over the past 5 years (Figure 7). Landowners most often (67%) neither agreed nor disagreed that there were too many hunters wanting to hunt their land (Figure 8). Seventy-six percent of landowners did not have hunter trespass problems on their land during the spring hunting season. Overall, 70% of landowners did not post their land to control hunter access.

DISCUSSION

Hunter access was not indicated as a problem for turkey hunters in 10 PAs during the 2005 spring season in Minnesota. Most hunters used private land for hunting and the majority found access to be easy. Hunter requests for hunting access were rarely denied. Hunters saw few individuals while hunting, and interference rates were low, which likely led to greater hunter safety and satisfaction. Hunt quality ratings were high.

Landowner attitudes about spring wild turkey hunters were positive. Trespassing issues were very low and posting land was not used to control hunting. Landowner perception of hunter density did not indicate they felt too many hunters were asking for permission to hunt. The majority of landowners did not feel that hunter density had increased over the past 5 years.

The data indicated that hunters were not concerned with access issues, interference rates, and safety. Landowner attitudes about hunters were positive and indicated that landowners did not feel pressured by hunters requesting access. The study indicated hunter satisfaction and landowner tolerance of hunters was positive in all the sampled PAs including PA 343, which had the highest hunter density in Minnesota in spring 2005. Thus, hunter density during the spring turkey season does not appear to be an issue for hunters or landowners at current levels, even in 2 PAs that had permit increases of $\geq$ 25% for the 2005 hunt.

In the second year of the project we will survey spring turkey hunters and landowners during the 2006 season in 10 additional PAs (Figure 1). The 2005 and 2006 landowner and hunter survey results will be used to determine impacts of hunter density and other variables on hunter interference and landowner tolerance of hunters. We will compare hunter interference and landowner attitude responses at varying hunter densities. This study will help to allocate permits at levels that will ensure a quality spring wild turkey hunt.

ACKNOWLEDGEMENTS

We would like to thank J. Krenz and J. Guidice for statistical and study design consulting, T. Klinkner for assisting in survey mailings, and B. Abel for data entry.

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Table 1. Hunter interference rates from the 2005 spring turkey hunter survey in Minnesota.

Permit Area	Hunter Density (hunters/buffered mi^2)	Interference Rate (%)	Hunt Quality
337	0.92	0.07	7.56
339	0.87	0.15	6.94
343	1.61	0.10	7.66
344	1.51	0.20	6.32
348	1.10	0.13	6.92
349	1.62	0.15	6.64
443	0.87	0.10	6.49
463	0.29	0.05	6.66
464	0.42	0.09	6.49
466	0.43	0.13	7.09

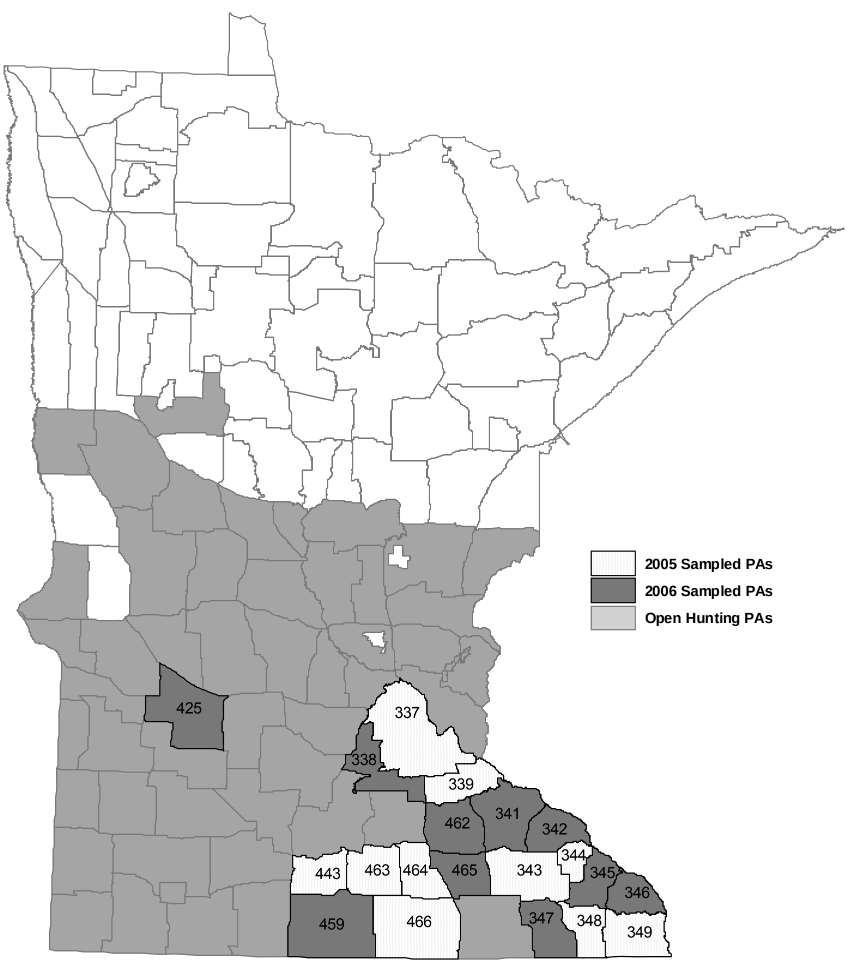

Figure 1. Permit areas (PAs) sampled during the 2005 and 2006 Minnesota spring turkey hunter and landowner survey.

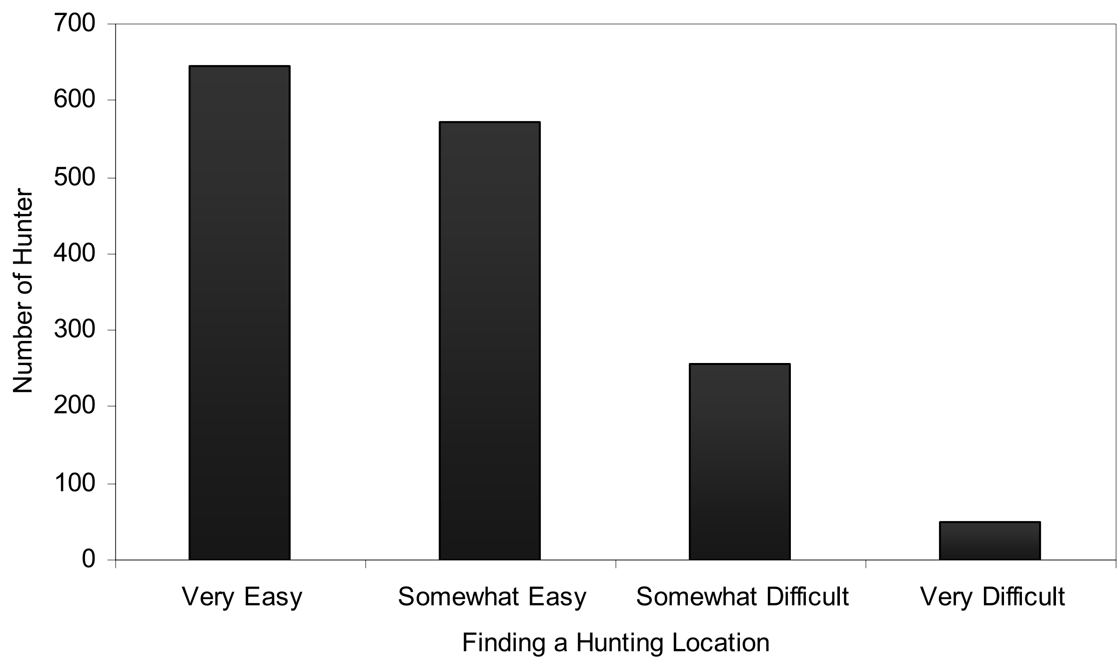

Figure 2. Difficulty ratings of finding a hunting location by Minnesota spring wild turkey hunters, April-May 2005.

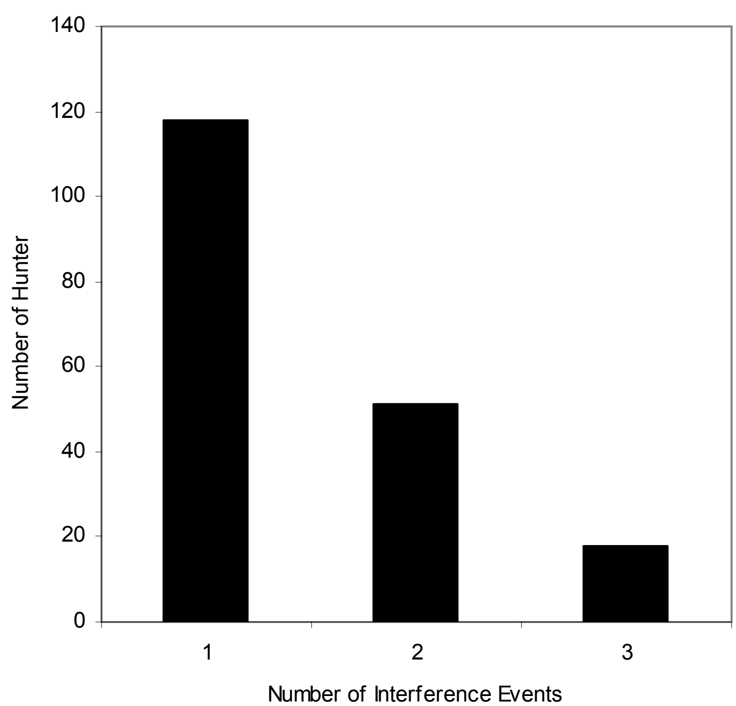

Figure 3. Number of times hunters were interfered with by other hunters while hunting during the Minnesota spring wild turkey season, April-May 2005.

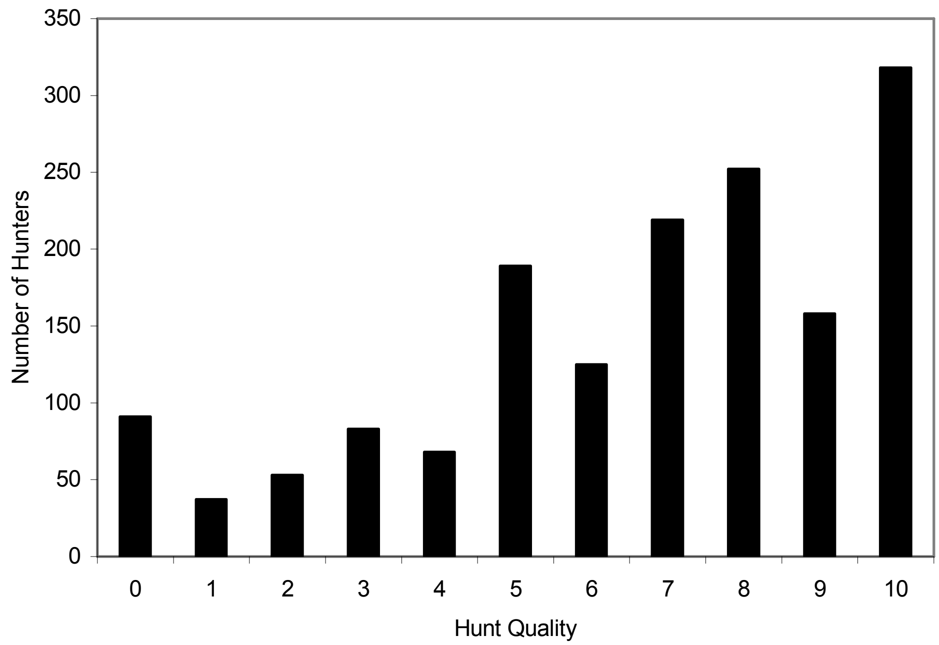

Figure 4. Hunt quality for the Minnesota spring wild turkey hunting season, April-May 2005.

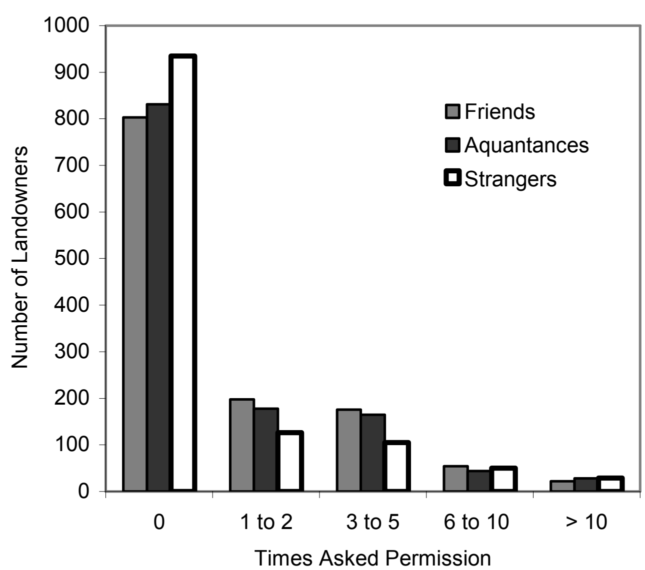

Figure 5. Number of times landowners were asked for permission to hunt their land by hunters for the Minnesota spring wild turkey season, April-May 2005

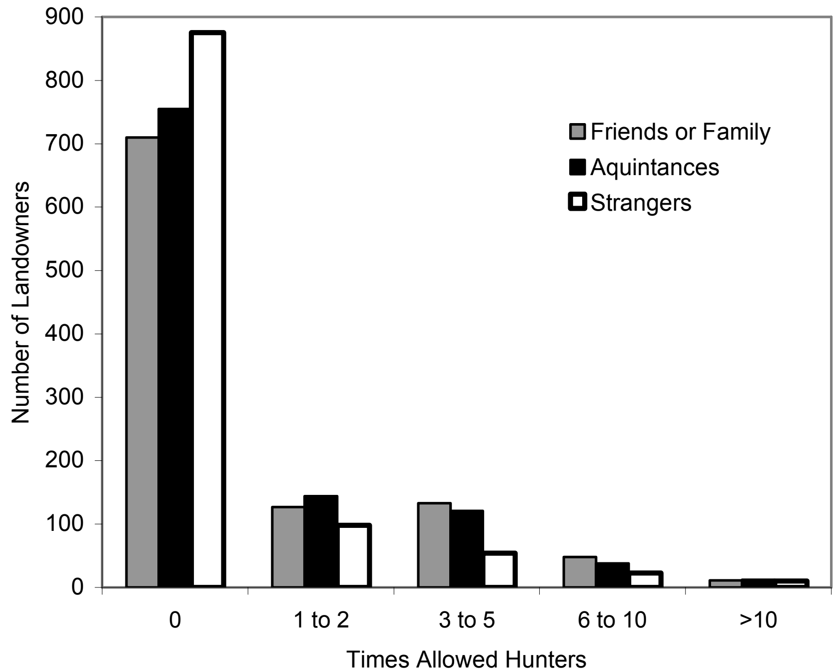

Figure 6. Number of times landowners granted hunting permission on their land during the Minnesota spring wild turkey season, April-May 2005.

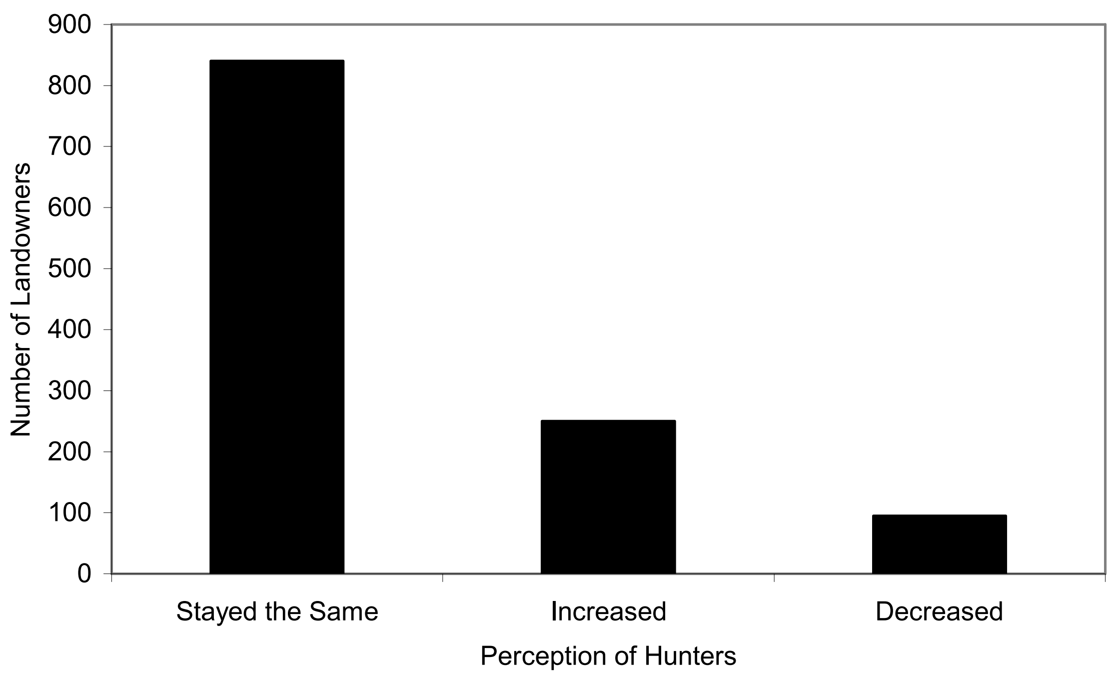

Figure 7. Landowner perception of the number of hunters requesting permission to hunt their land over the past 5 years, April-May 2005.

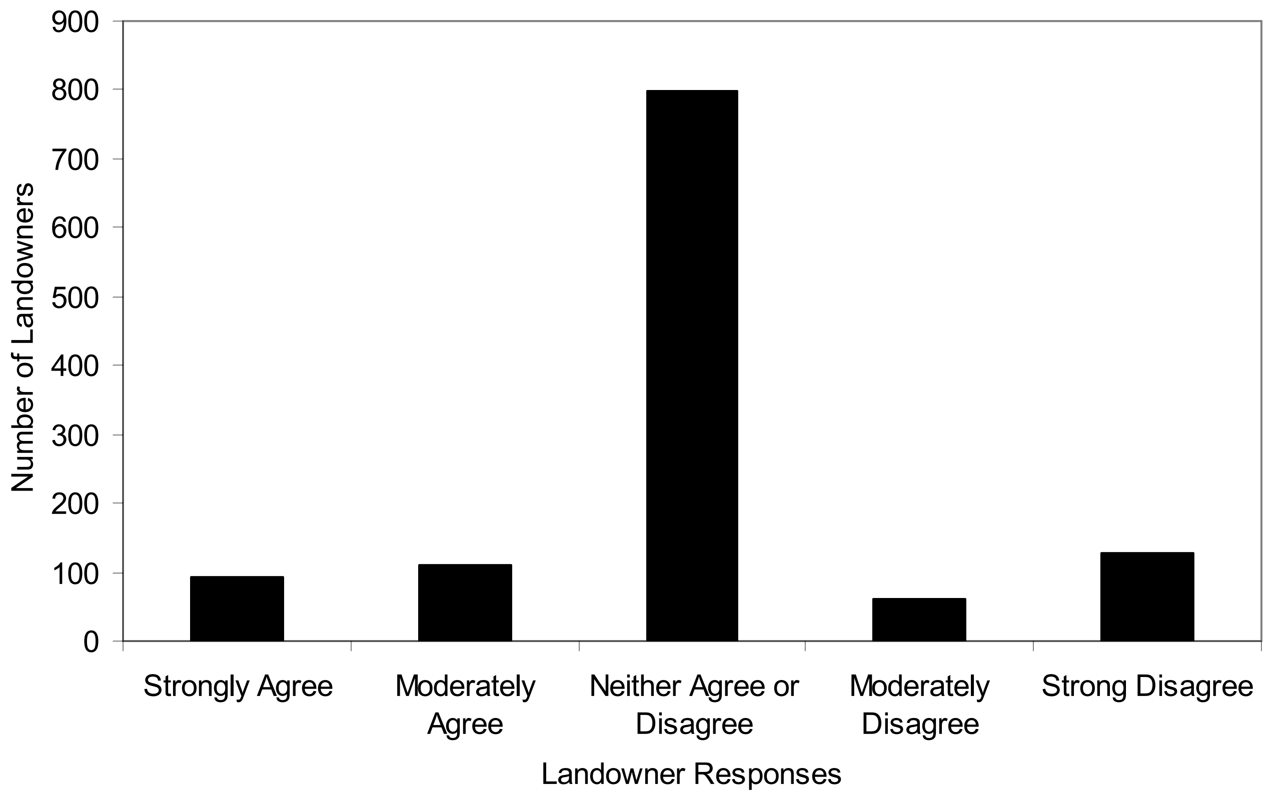

Figure 8. Landowner responses when asked if too many hunters wanted to hunt their land during the Minnesota spring wild turkey season, April-May 2005.

Appendix A. Hunter instrument for the 2005 Minnesota spring wild turkey hunting season survey.

Minnesota Spring Turkey Hunter Survey
*Please respond to all questions based on the
SPRING 2005 TURKEY SEASON.

1. Did you hunt turkeys in Minnesota during the spring 2005 season? Yes____ No*____
   *If no, you do not need to continue but please return survey.

2. Which wild turkey permit area did you hunt in? __________

3. Did you have a landowner permit or a regular lottery permit?
   Landowner____Regular Lottery____

4. Which season did you hunt? April 13-17___ April 18- 22___ April 23-27___ April 28-May 2___ May 3-7___ May 8-12___ May 13-19___ May 20-26___

5. How many days did you hunt turkeys during spring 2005? __________

6. How did you hunt turkeys in 2005? Shotgun only____ Bow Only____ Shotgun and Bow___

7. How many turkeys did you see while turkey hunting in 2005? __________

8. How many turkeys did you shoot at? __________

9. Were you successful in bagging a turkey? Yes*____ No____
   *If yes, was it killed in the morning or afternoon? AM____ PM____
   *If yes, with what weapon did you harvest your turkey? Shotgun____ Bow____

10. How difficult was it for you to find a place to hunt during the spring 2005 wild turkey hunting season? (check one answer)
    Very easy____ Somewhat easy____ Somewhat difficult____ Very difficult____

11. Did you hunt on public land or private land during the spring 2005 season?
    Public____ Private*____ Both____
    *If you hunted on private land, how many landowners turned down your request for permission? ____

12. Did you at any time feel you were put in danger by other hunters while turkey hunting?
    Yes____ No____

13. On average, how many hunters, other than members of your own party, did you see each day while you were actually in the field hunting during spring 2005?
    ____

14. How many times did hunters, other than members of your own party, interfere with your hunting during spring 2005? __________

15. How many times did people **other than hunters** interfere with your hunting during spring 2005? ___

16. Rate the quality of your turkey hunting experience during spring 2005 on a scale of 0-10 (check one number):

    Poor Quality Average Quality Excellent Quality

    0____ 1____ 2____ 3____ 4____ 5____ 6____ 7____ 8____ 9 ____ 10____

Appendix B. Landowner instrument for the 2005 Minnesota spring wild turkey hunting season survey.

Minnesota Spring Turkey Hunt Landowner Survey

*Please respond to all questions based on your land in County for the SPRING 2005 Turkey Hunting Season.

1. How many total acres of land do you own in «COUNTY» County?

   Acres Cropland__________ Acres Woodland __________ Other Acres__________

2. How long have you owned your land?

   ☐ 0-5 years ☐ 6-10 years ☐ > 10 years

3. Is your primary residence on this land?

   ☐ Yes ☐ No

4. Which of the following are reasons why you own this property? (Please check all that apply)

   ☐ I use it to make a living farming.
   ☐ I use it for non-hunting recreational purposes.
   ☐ I want to preserve the land for the future.
   ☐ I like the wildlife that lives on my land.
   ☐ I use it for hunting.
   ☐ I am using this land for investment or development.
   ☐ Other. Please specify:______________________________________

5. Do you currently lease out any of your land for farming, spring turkey hunting, or other hunting? (Please check one response for each item.)

For farming	☐ Yes	☐ No
For spring turkey hunting	☐ Yes	☐ No
For other hunting	☐ Yes	☐ No

6. Have you seen wild turkeys on your land in the past year?

   ☐ Yes ☐ No

7. Did you personally hunt wild turkeys on your land during spring 2005?

   ☐ Yes ☐ No

8. During the spring of 2005, how many turkey hunters **asked permission to hunt** on your land that were family or friends, acquaintances, or strangers? (Please check one box for each category.)

Friends or Family	☐ 0	☐ 1-2	☐ 3-5	☐ 6-10	☐ >10
Acquaintances	☐ 0	☐ 1-2	☐ 3-5	☐ 6-10	☐ >10
Strangers	☐ 0	☐ 1-2	☐ 3-5	☐ 6-10	☐ >10

9. During the spring of 2005, how many turkey hunters did you **allow to hunt** on your land that were family or friends, acquaintances, or strangers? (Please check one box for each category.)

Friends or Family	☐ 0	☐ 1-2	☐ 3-5	☐ 6-10	☐ >10
Acquaintances	☐ 0	☐ 1-2	☐ 3-5	☐ 6-10	☐ >10
Strangers	☐ 0	☐ 1-2	☐ 3-5	☐ 6-10	☐ >10

10. Over the past 5 years do you think the number of hunters requesting permission to hunt wild turkeys during the spring season on your land has increased, decreased, or stayed the same?

☐ Increased
☐ Decreased
☐ Stayed the same

11. How do you feel about the following statement: There are too many spring turkey hunters requesting permission to hunt on my land?

☐ Strongly agree
☐ Moderately agree
☐ Neither agree or disagree
☐ Moderately disagree
☐ Strongly disagree

12. How do you feel about the number of hunters requesting permission to hunt on your land?

☐ Way too many
☐ Too many
☐ Just Right
☐ Too few
☐ Way too few

13. Did you have a problem with hunters trespassing on your property during the 2005 spring turkey hunt?

☐ Yes ☐ No

14. Do you post signs on your land in an effort to control hunter access?

☐ Yes ☐ No

Please provide any additional comments.

2005 MINNESOTA SPRING WILD TURKEY ARCHER SURVEY

Sharon L. Goetz, Bryan J. Abel, and Allison M. Boies[1]

SUMMARY OF FINDINGS

The addition of an archery season during the last 2 time periods (G and H) of the 2005 spring wild turkey (*Meleagris gallopavo*) hunting season lead to concerns about potential impacts on hunter density and hunt quality. An archer survey instrument modified from the traditional spring turkey hunter survey was used to collect information on hunting pressure, hunter density, and interference rates by permit area hunted. The addition of 2,210 archers on the landscape did increase hunter density in some permit areas, however interference rates and hunt quality did not appear to be negatively impacted in the 25 permit areas open for the archery season.

INTRODUCTION

Spring wild turkey (*Meleagris gallopavo*) hunter surveys are conducted after the completion of the spring hunting season to gather hunter information, such as hunter interference rates that are used in the spring permit allocation model (Kimmel 2001). Estimates of hunt quality obtained from these surveys are used in making future spring hunting management decisions.

Beginning in 2005, resident and nonresident turkey hunters were able to purchase an archery license for the final 2 time periods (G and H) for any permit area with ≥50 permits available per time period. Both hunters unsuccessful in the lottery and those who never applied are eligible for the archery season. This survey was conducted to provide information regarding the 25 permit areas that qualified for the archery season, and potential impacts on hunter density and interference rates. Although successful lottery applicants can use a bow during the regular season, this survey focuses on archers who purchased an archery season permit.

METHODS

Hunters who purchased archery licenses were randomly selected from the Electronic License System (ELS) database of spring turkey hunting recipients. A total of 2,210 hunters purchased an archery license. The survey instrument (Appendix A), modified from precious spring wild turkey hunter surveys, was mailed to 496 archery license holders. Three survey mailings were conducted with second and third mailings were sent to non-respondents. The first mailing was sent 6 June 2005, the second on 29 June 2005, and the final on 9 August 2005.

RESULTS

Overall 366 surveys were returned for a response rate of 74%. Of the survey respondents, 332 (91%) stated they hunted the spring 2005 archery season.

All 25 permit areas open to the archery season were hunted by archery hunters, along with 3 others that were not designated for archery hunting (Permit areas 228, 235, 410; Figure 1). Permit areas 236 and 343 were each hunted by 10% of the sample (~33 hunters). Permit areas 337, 341, and 442 each accounted for 4-6% of the sample with 14, 14, and 19 hunters, respectively. All other permit areas hunted accounted for less than 4% of the sample each, and hunter numbers ranged from 1 to 11. A total of 94 hunters (28%) did not specify or entered invalid permit area information.

Spring 2005 archery hunters hunted an average of 4.2 days, saw an average of 11.3 wild turkeys, and shot at an average of 0.5 turkeys. Based on survey results, there were 48 wild turkeys registered in 13 different permit areas (Figure 2) for a success rate of 14.5%.

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Experienced bow hunters tagged a majority of the wild turkeys registered, with 46 successful hunters stating they had archery hunted big game in the past. Only 14% (54) of respondents stated they had never archery hunted prior to spring 2005.

Morning proved to be the best time to harvest a turkey with 42 turkeys shot, compared to 6 turkeys harvested in the afternoon. The majority of archers hunted private land (88%), 14% hunted both public and private land, while only 6% hunted solely on public land. Hunters spending most of their time hunting during time period G (13-19 May) shot 58% of the harvested turkeys (28); with 15 turkeys harvested by hunters focusing effort in time period H (20-26 May). Twenty-six hunters stated they hunted both time periods equally.

The majority of hunters found it was very easy (41%) or somewhat easy (36%) to find a place to hunt (Figure 3). Hunters who gained access to private land were refused by an average of 0.7 landowners.

A majority of the hunters (71%) did not see another hunter while in the field. Only 11% of spring archery hunters experienced at least one interference event. Hunt quality was rated average or above by 80% of archers (Figure 4).

DISCUSSION

The opening of an archery season, an additional spring turkey hunting opportunity, during the last 2 time periods (G and H) of the 2005 season raised concerns about potential impacts on hunter density and hunt quality, particularly in areas that already have hunter densities >0.4 hunter/km^2 (>1 hunter/mi^2) of huntable turkey habitat. Based on survey responses, hunting pressure by archers was spread evenly across seasons and time periods. Permit areas 236 and 343 had the most archers with approximately 33 individuals (10%) hunting each area. The majority of turkey hunters indicated little interference by other hunters and non-hunters, even though the addition of the archery season increased the chance of more individuals being in the woods compared to previous spring seasons. Most spring archery hunters rated the experience as average to excellent and many respondents commented that they were highly in favor of the new archery season.

At current participation levels, the archery season, although increasing hunter density in some permit areas, does not seem to have impacted hunter interference or hunt quality in eligible areas. As awareness and popularity of the new archery season grows and more individuals purchase an archery license, there is still potential for interference and hunt quality impacts in future seasons. We plan to continue to monitor impacts of the archery season on hunting pressure, hunter density, and interference rates by conducting the archery survey in spring 2006.

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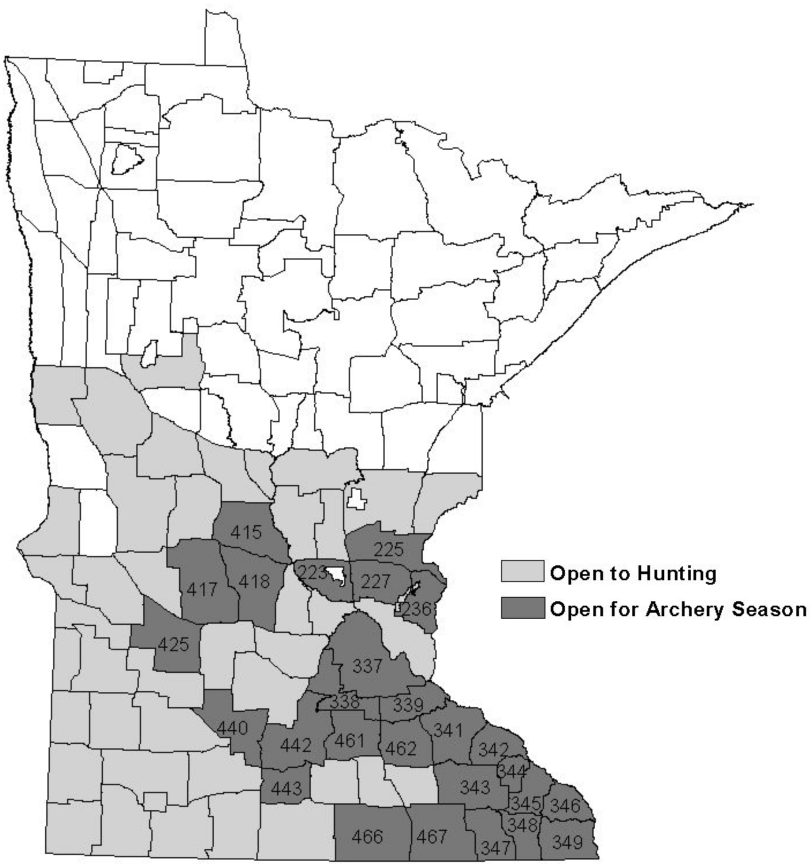

Figure 1. Permit areas open to the 2005 spring wild turkey archery season in Minnesota.

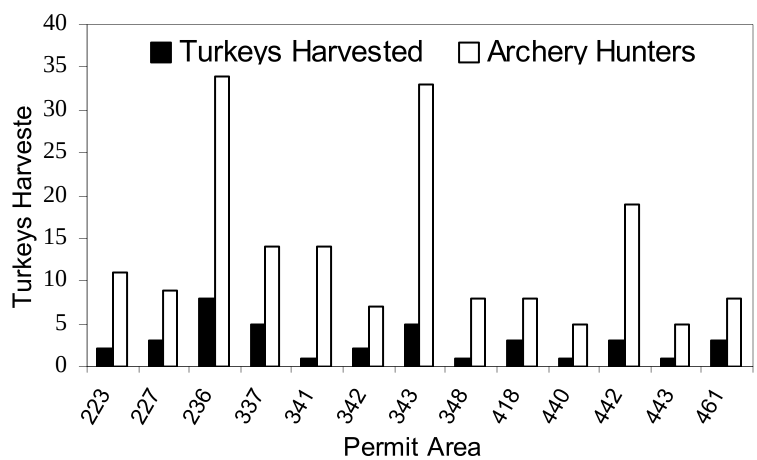

Figure 2. Turkeys harvested and the number of archery hunters by permit area for the 2005 spring archery season in Minnesota.

Figure 3. Difficulty of finding a place to hunt by 2005 spring wild turkey archery hunters in Minnesota.

Figure 4. Quality of the hunt experienced by 2005 spring wild turkey archery hunters in Minnesota.

Appendix A. Hunter instrument for the 2005 Minnesota spring wild turkey archery season survey.

Minnesota Spring Turkey Archery Survey
*Please respond to all questions based on the
SPRING 2005 TURKEY SEASON.

1. Did you hunt turkeys in Minnesota during the spring 2005 season? Yes____ No*____
   *If no, you do not need to continue but please return survey.

2. Which wild turkey permit area did you hunt the most? _______
   List all other permit areas you hunted
   ____

3. Have you bowhunted big game or wild turkeys in the past? Yes*____ No____
   *If yes, how many years have you bowhunted:

   turkey__________ deer__________ other__________

4. Which time period did you hunt the most? May 13-19___ May 20-26___

5. How many days did you hunt turkeys during spring 2005? __________

6. How many turkeys did you see while turkey hunting in 2005? __________

7. How many turkeys did you shoot at? __________

8. Were you successful in bagging a turkey? Yes*____ No____
   *If yes, was it killed in the morning or afternoon? AM______ PM_______

9. How difficult was it for you to find a place to hunt during the spring 2005 wild turkey hunting season? (check one answer)
   Very easy____ Somewhat easy____ Somewhat difficult____ Very difficult____

10. Did you hunt on public land or private land during the spring 2005 season?
    Public_____ Private*_____ Both_____
    *If you hunted on private land, how many landowners turned down your request for permission? _

11. Did you at any time feel you were put in danger by other hunters while turkey hunting?
    Yes____ No____

12. On average, how many hunters, other than members of your own party, did you see each day while you were actually in the field hunting during spring 2005? ______

13. How many times did hunters, other than members of your own party, interfere with your hunting during spring 2005? ______

14. How many times did people **other than hunters** interfere with your hunting during spring 2005? __

15. Rate the quality of your turkey hunting experience during spring 2005 on a scale of 1-10 (check one number):
    Poor Quality Average Quality Excellent Quality

    0____ 1____ 2____ 3____ 4____ 5____ 6____ 7____ 8____ 9 ____ 10____

Additional comments can be written on the back.

SURVIVAL AND HABITAT USE OF EASTERN WILD TURKEYS TRANSLOCATED TO NORTHWESTERN MINNESOTA.

Sharon L. Goetz, Brett J. Goodwin[1], and Chad J. Parent[1]

SUMMARY OF FINDINGS

Translocations of eastern wild turkeys (*Meleagris gallopavo slyvestris*) in Minnesota have increased the range as far north as a line from the St. Croix River Valley south of Duluth through the Lake Mille Lacs area and northwest to Mahnomen and Norman Counties in northwestern Minnesota. There is continued public interest for expanding wild turkey populations northward. To assess the potential for transplanting wild turkeys farther north, information on survival, habitat use, and potential depredation in agricultural areas will be explored in a 2-year research project. In winter 2006, 9 of 23 (39%) released turkeys survived the winter in Red Lake County and 7 of 22 (32%) in Pennington County.

INTRODUCTION

The current distribution of eastern wild turkeys (*Meleagris gallopavo slyvestris*) in Minnesota extends well beyond the ancestral range identified by Leopold (1931). Translocations of wild turkeys in Minnesota have increased the range from the St. Croix River Valley south of Duluth through the Lake Mille Lacs area and northwest to Mahnomen and Norman Counties in northwestern Minnesota. The Minnesota Department of Natural Resources (DNR) has had public interest for expanding wild turkey populations northward. However, additional research is needed to provide information regarding wild turkey ecology in northern habitats, the impact of winter severity on wild turkeys at the population level, and effective management techniques for northern populations.

Physiologically, wild turkeys should be able to survive northern Minnesota winters if food is available (Haroldson 1996, Haroldson et al. 1998). However, wild turkeys' ability to find food can be limited by deep snow in northern regions (Porter et al. 1983, Haroldson et al. 1998). Severe winter weather has also been associated with decreased recruitment as reduced hen body condition impacts hatching success (Porter et al. 1983). Additionally, it is becoming more apparent that wild turkeys' tolerance for human contact increases when snow conditions intensify the need for food (Kulowiec and Haufler 1985, Gillespie 2003, Moriarty and Leuth 2003). As human tolerance increases, the potential for agricultural depredations and urban turkey problems increase. Ultimately, the ecological northern limit of wild turkey distribution will likely be determined by interactions of temperature, food availability (influenced by snow cover), and habitat quality (Haroldson 1996). The objective of this 2-year study is to collect information on survival, habitat use, and potential depredation in agricultural areas before wild turkeys are transplanted into additional northwestern Minnesota counties.

STUDY AREA

We used remotely sensed data (i.e. land cover maps, aerial photos, etc.) and Geographic Information System software to identify potential wild turkey habitat in northwestern Minnesota north of the current turkey range. Landscape composition and configuration were considered in determining potential release sites that met wild turkey habitat requirements, while decreasing potential for unwanted human/turkey interactions. Landscapes with a good mix of open and forested habitats were selected, while areas where feedlots and domestic turkey farms were located were avoided. Sites that allow for future expansion of turkey

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populations were prioritized for wild turkey study areas. Two release sites were chosen, one each in Red Lake and Pennington counties (Figure 1).

The Red Lake County release site near Red Lake Falls, Minnesota (RLF) is located in the Hardwood Hills Ecological Classification System subsection. The major land use in this subsection is agriculture with upland hardwoods surrounding lakes, on beach ridges, and steep slopes. The turkey release site is near the confluence of the Red Lake and Clearwater rivers.

Forested beach ridges and wet swales are common features of the Aspen Parkland subsection where the Pennington County release site near Thief River Falls (TRF) is located. The release site will be located on a beach ridge. Beach ridges and river corridors provide opportunity for turkey expansion by following the north-south running beach rides and traveling along riparian corridors.

The average number of days per year where snow depths were greater than or equal to 30 cm (12 inches) varies from 30 to 40 days in the portions of Pennington and Red Lake county surrounding the release sites (MCWG 2005).

METHODS

Wild turkeys were captured from established flocks in Minnesota during January-March 2006 using rocket nets (Bailey 1980). Trapping was conducted by DNR trapping crews. Captured wild turkeys were weighed, aged (juvenile or adult), leg-banded, equipped with a backpack style radio-transmitter, and released within 1-3 days following capture. Transmitters (95 - 104 g, 40 cm whip antenna) have an approximate battery life of 3 years and a mortality sensitive switch (Advanced Telemetry Systems-ATS, Isanti, MN, USA). Only females were radioed because hens are easier to catch, more susceptible to winter stress, and have greater influence on recruitment to the following years population.

Radioed hens were monitored 3 to 4 times/week during winter. Winter was defined as 1 January through 31 March (Kane et al. 2003, Kassube 2005). Birds were located via triangulation from known locations on roads using ≥3 bearings for each location. When transmitters were retrieved soon after mortality signals occurred, efforts were made to determine cause of death by field sign (Thogmartin and Schaeffer 2000).

RESULTS

Fifty-nine females and 21 males were released at the 2 sites from 19 January 2006 to 2 March 2006. At the RLF site 29 radioed hens and 10 males were released, while 30 radioed hens and 9 males were released at the TRF site. Fourteen hens died within 7 days of their release, the typical censor period for wild turkeys with radio transmitters. With these individuals censored, 9 of 23 (39%) turkeys survived the winter season at the RLF site and 7 of 22 (32%) at the TRF site. Both avian and mammalian predation has been identified in addition to a turkey that was likely killed by a car collision.

We plan to release additional turkeys to fill each site to sample size during winter 2007. We will continue to monitor seasonal survival in addition to collecting data regarding wild turkey habitat use, recruitment, and landowner attitudes about wild turkeys.

ACKNOWLEDGEMENTS

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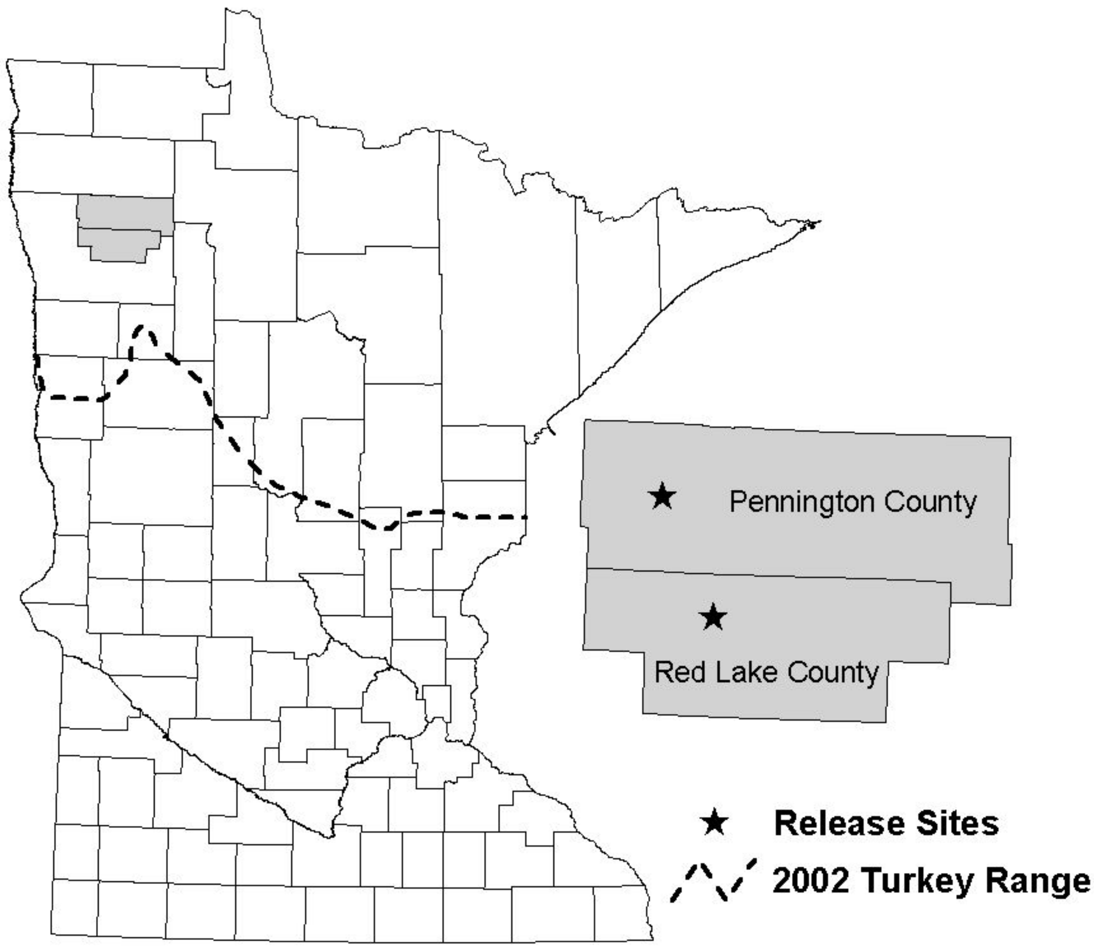

Figure 1. Wild turkey release site locations in northwestern Minnesota, January-March 2006.

PUBLICATIONS

Below are lists of scientific reports and other publications by personnel in the Wildlife Populations and Research Unit for the approximate period March 2005 through February 2006. Some titles by Unit personnel pertain to work done while employed by the DNR, but other reports are on work done elsewhere (e.g., as a graduate student, employed by another agency, while on leave of absence, etc.) An (*) before an author's name indicates that the report was listed as in press or in review in previous summaries of research project findings. Included under scientific reports are those which have been published and those accepted for publication (in press). Names in bold indicate a DNR employee.

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