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

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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.

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ECOLOGY AND POPULATION DYNAMICS OF BLACK BEARS IN MINNESOTA

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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).

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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.

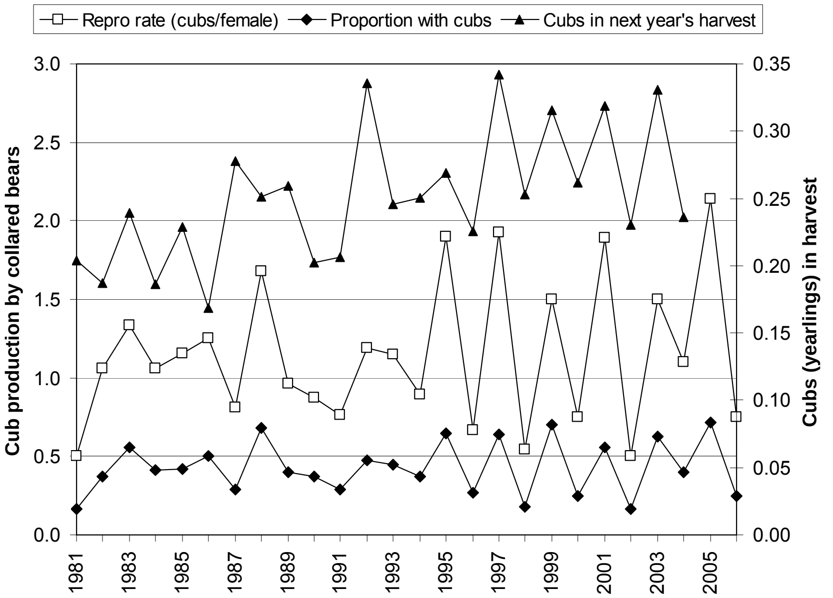


Figure 1. Comparison of reproductive data obtained from collared bears on the Chippewa National Forest (CNF) to the age structure of the statewide harvest. The strong reproductive synchrony observed among the collared bears in the CNF since 1995 (which was observed as well among collared bears at Camp 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).

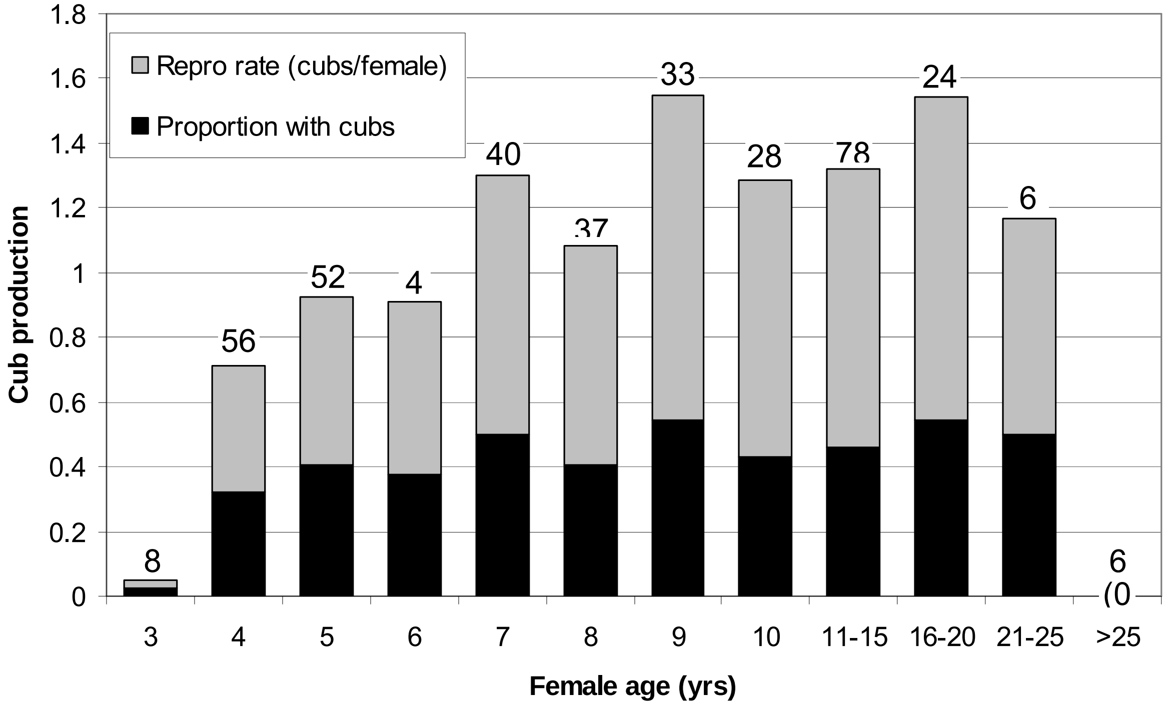


Figure 2. Age-specific cub production of bears in the Chippewa National Forest (central Minnesota) measured as the proportion of females with cubs during March den visits, 1982–2006, and cubs (M+F) per female. Sample sizes shown above bars represent bear-years (bears x years). However, only 2 individuals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

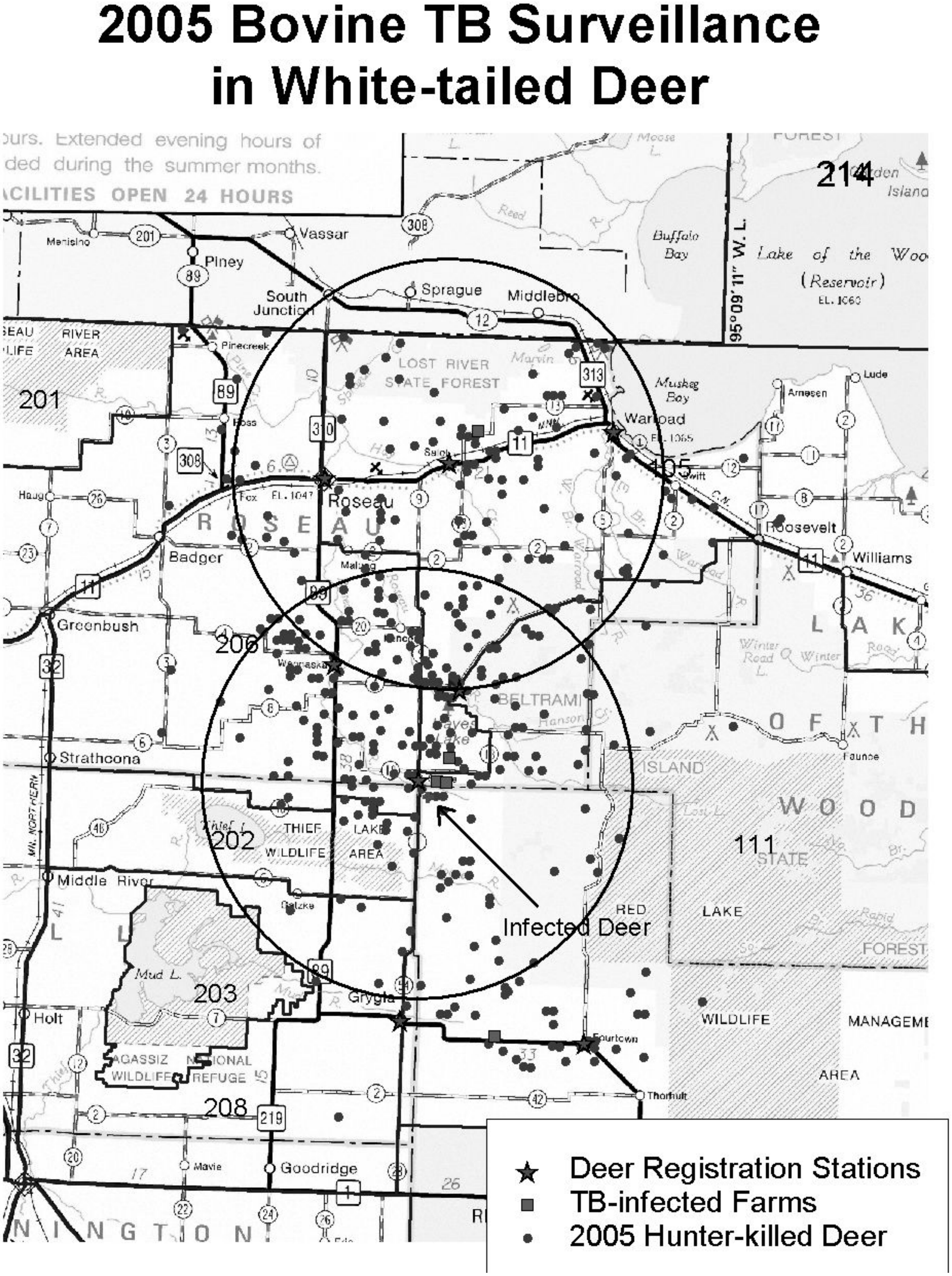
From January–April 2006, an additional 89 deer were harvested under shooting permits that were issued to landowners of bovine-TB infected cattle herds or their fence-line contacts. Given our theory of this disease originating in wild deer as a spill-over from infected cattle, it was highly likely that 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

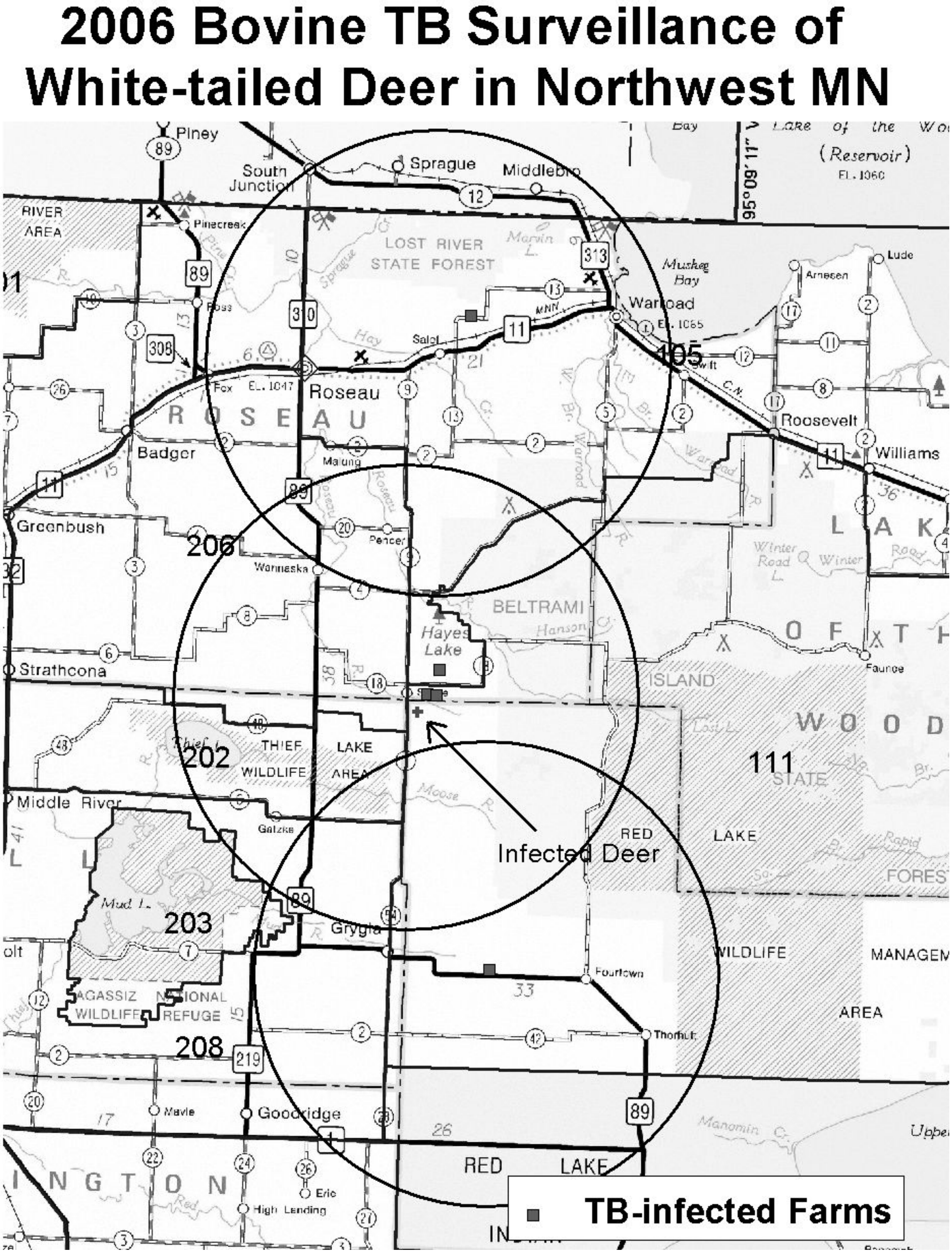
We would like to thank the students and faculty from the University of Minnesota, College of Veterinary Medicine, that assisted in our sampling efforts. Also, thanks to DNR's Jeff Lawrence, Dave Kuehn, and Carolin Humpal for volunteering their time to work at registration stations, as well as Steve Benson 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.



2005 Bovine TB Surveillance in White-tailed Deer

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



2006 Bovine TB Surveillance of White-tailed Deer in Northwest MN

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

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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 (1991̃2002) objectives were to examine (1) serum progesterone as an indicator of pregnancy in free-ranging white-tailed deer (0.5̃15.5 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 0.5̃15.5 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 87.5̃100.0% 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 2.6̃15.6 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

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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 ($P \le 0.05$) 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^{o} C (0^{o} 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^{o} C or 19.4^{o} 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 the1950s1960s, 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 79 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 ($\geq$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

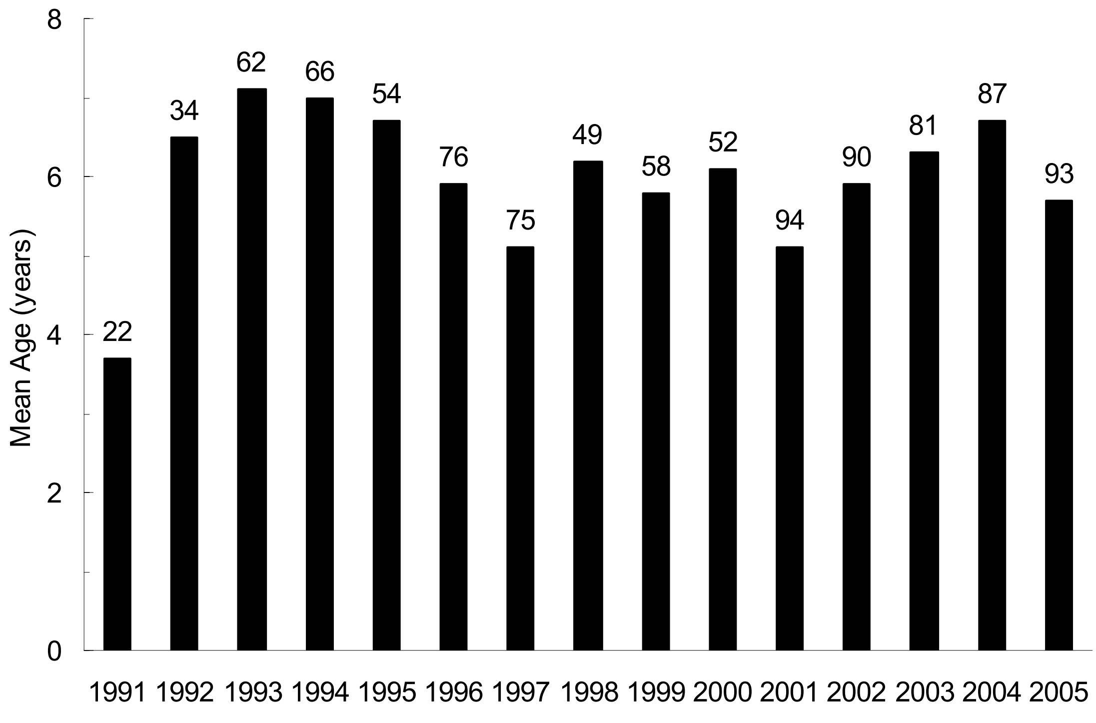


Figure. 1. Mean age of radiocollared female white-tailed deer among years, north-central Minnesota, 1 January 1991–31 December 2005. (Sample sizes are above bars.)

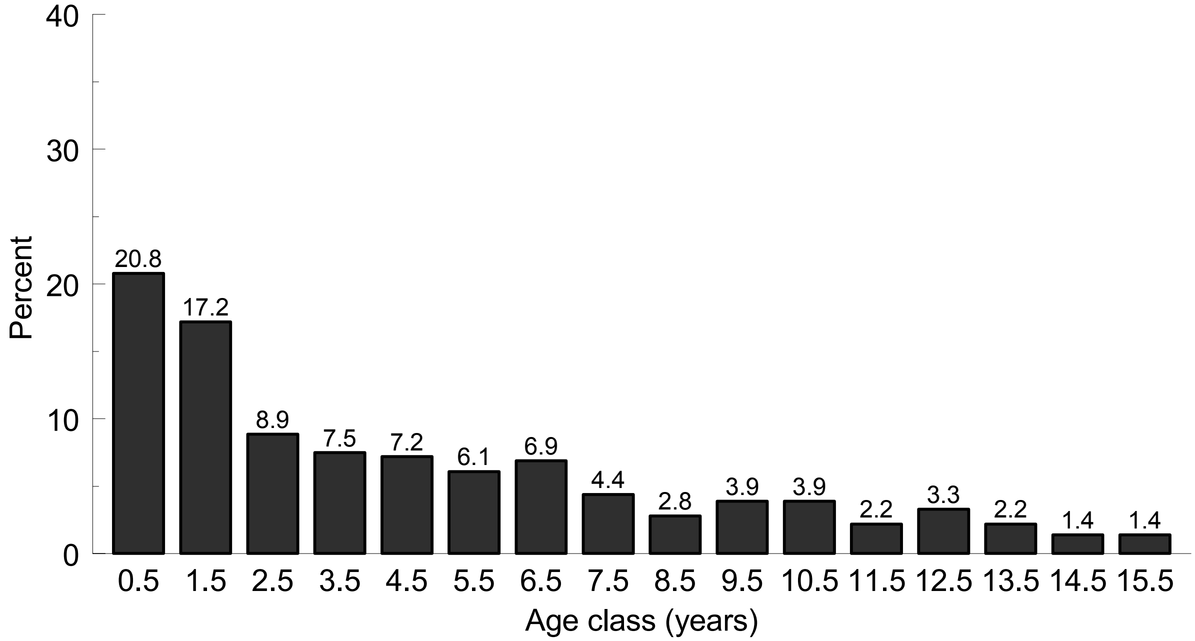


Figure 2. Age distribution of radiocollared female white-tailed deer (pooled across study sites), north-central Minnesota, 1 January 1991–31 December 2005.

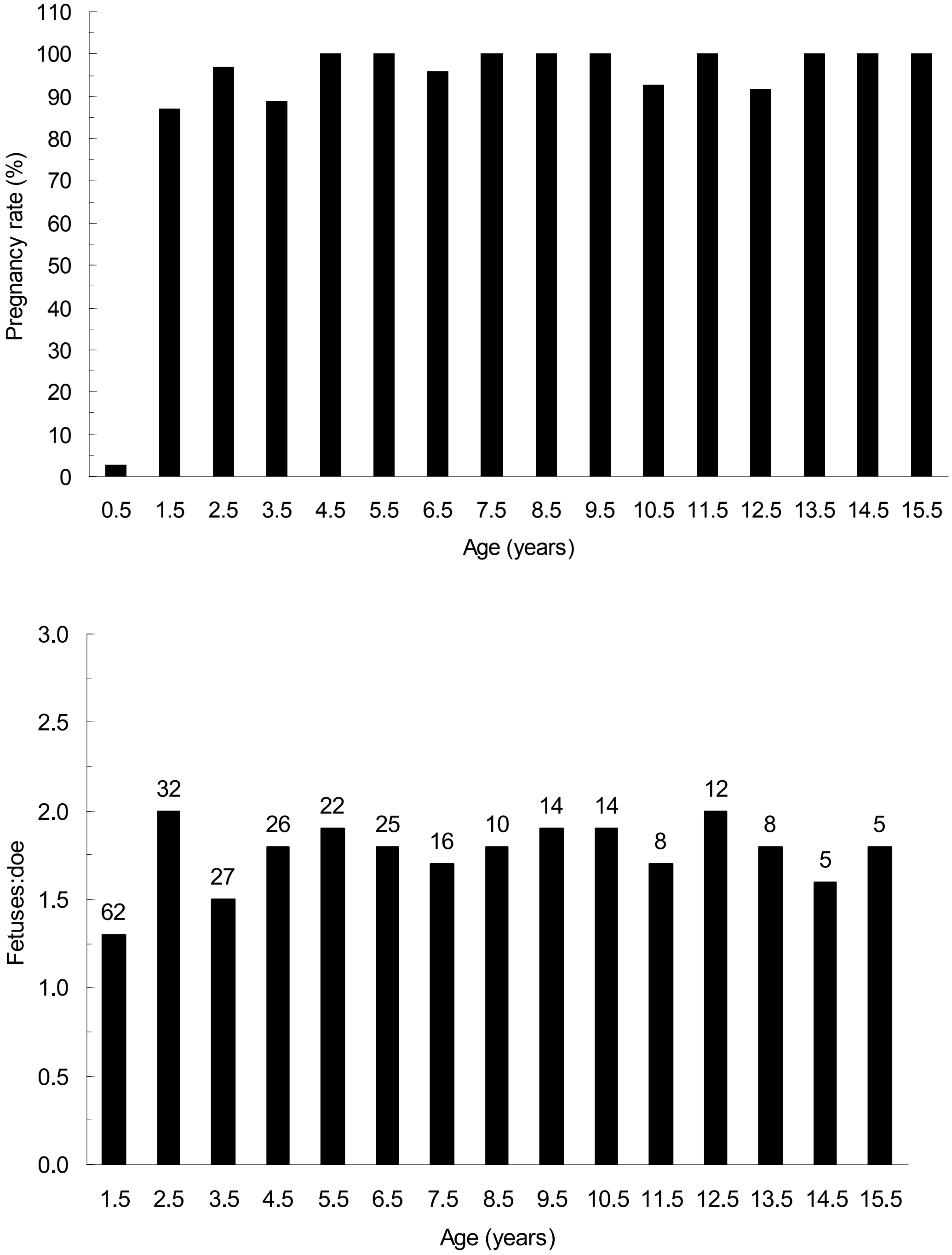


Figure 3. Age-specific pregnancy rate and fecundity (sample sizes are above bars) of radiocollared white-tailed deer (4 study sites pooled) in north-central Minnesota, winters 1991–2005.

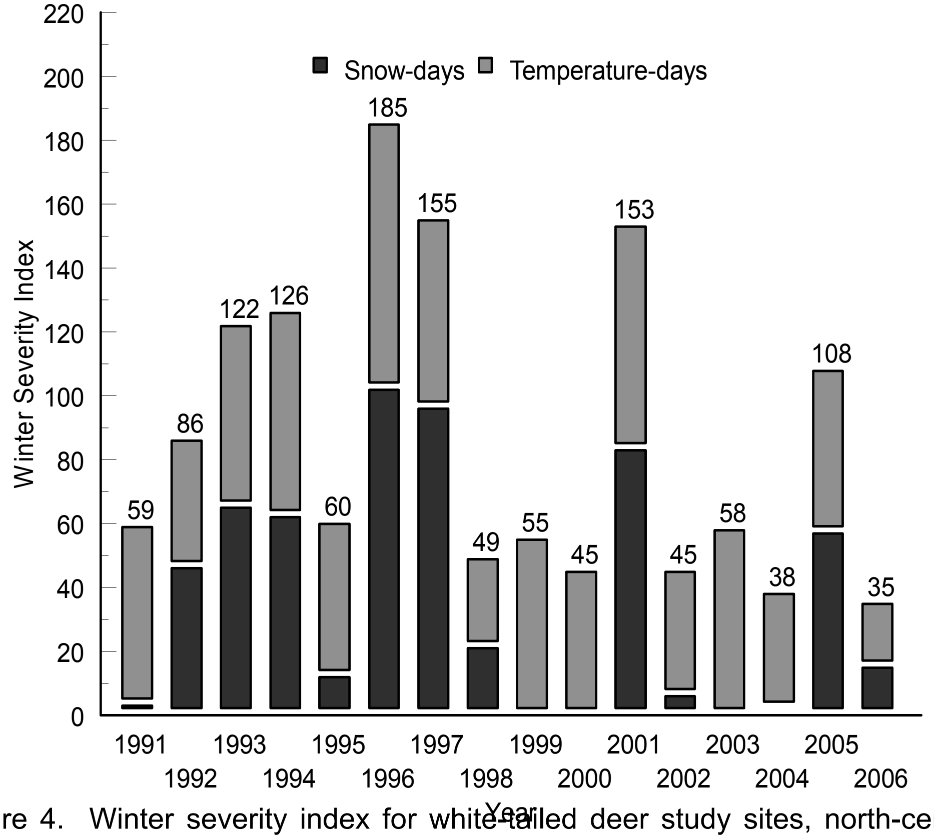


Figure 4. Winter severity index for white-tailed deer study sites, north-central Minnesota, winters 1990–1991 to 2005–2006. One point is accumulated for each day with an ambient temperature ≤ -17.8° C (temperature-day), and an additional point is accumulated for each day with snow depths ≥38.1 cm (snow-day).

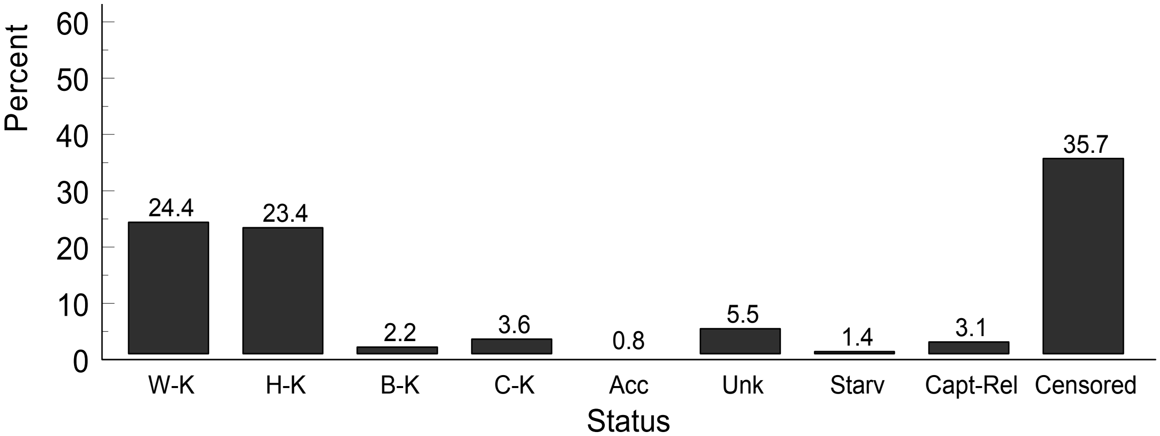
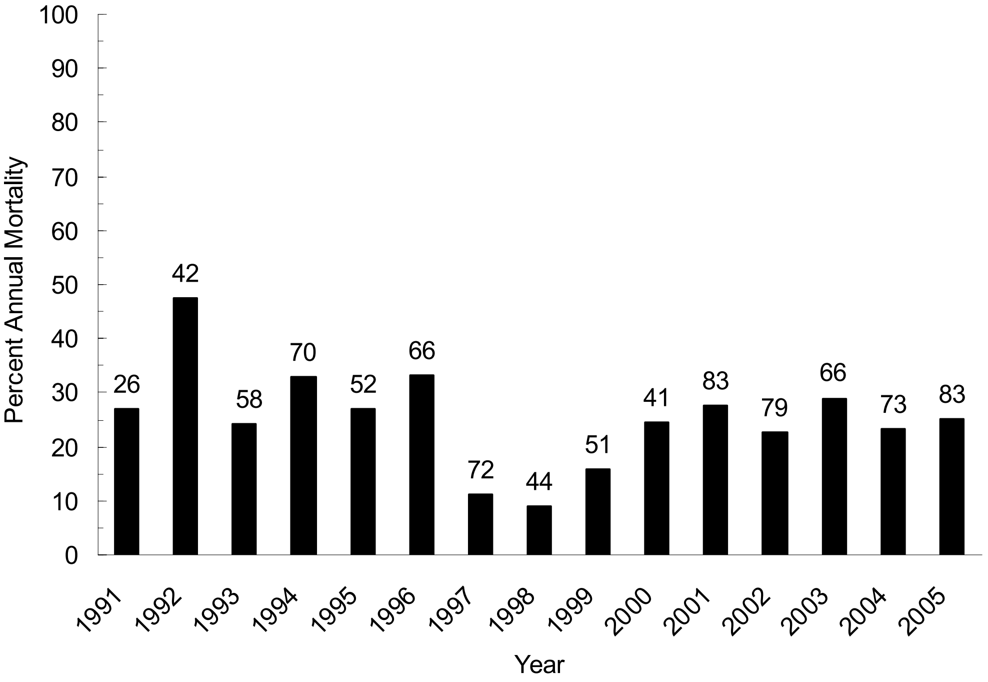


Figure 5. Status of radiocollared female deer, north-central Minnesota, 1 January 1991–31 December 2005. (W-K = wolf-kill, H-K = hunter-kill, B-K = bobcat-kill, C-K = car-kill, Acc = accidental, Unk = unknown cause, Starv = starvation, Capt-Rel = capture-related.) Censored deer include those that were 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).



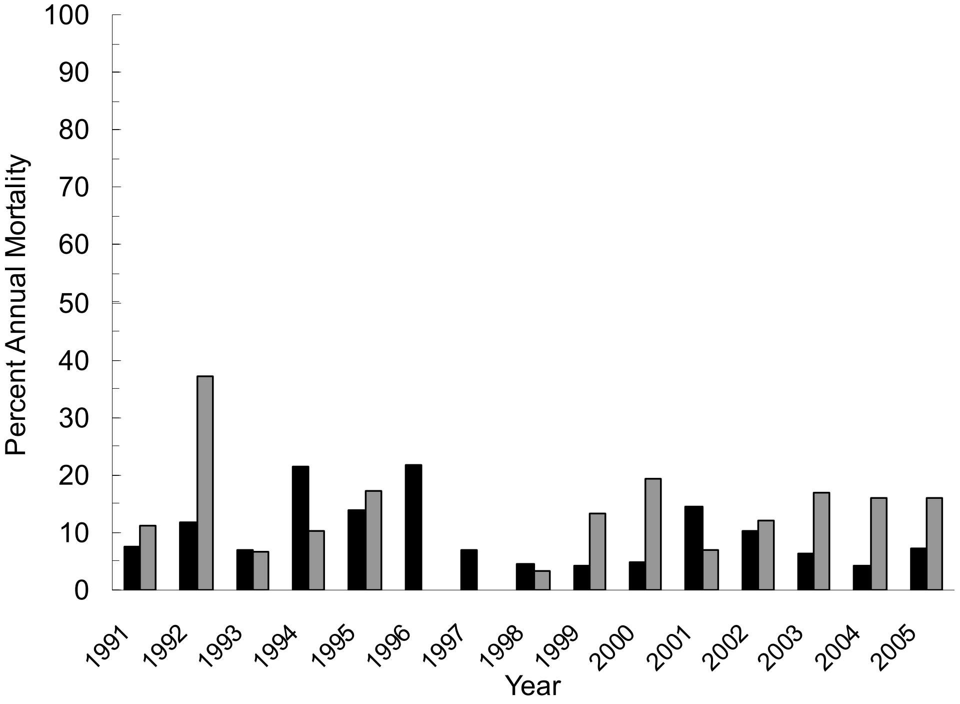


Figure 6. Annual (1 January–31 December) percent mortality of radiocollared, female white-tailed deer (top; sample sizes are above bars), and annual percent mortality attributable to wolf predation and hunter harvest (bottom, 4 sites pooled), north-central Minnesota 1991–2005. (Hunter harvest was calculated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

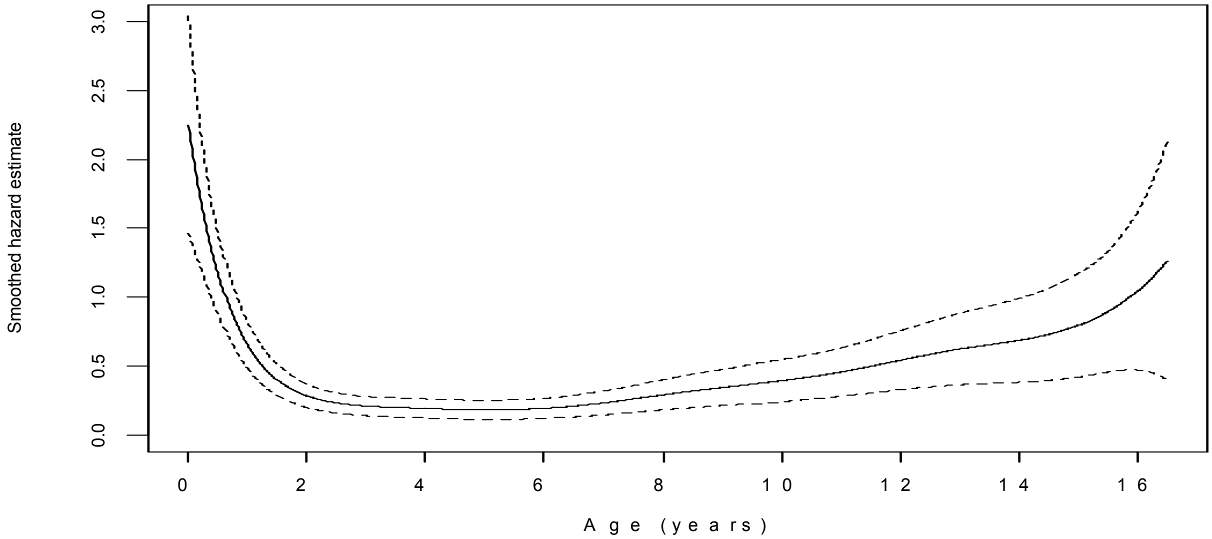


Figure 7. Penalized likelihood estimate of the 'all-causes' hazard (i.e., instantaneous probability of death) for radiocollared, female white-tailed deer (including neonates), north-central Minnesota. The data include 302 females ≥0.5 years old, monitored from 1 January to 31 December 2003, and 76 neonates (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).

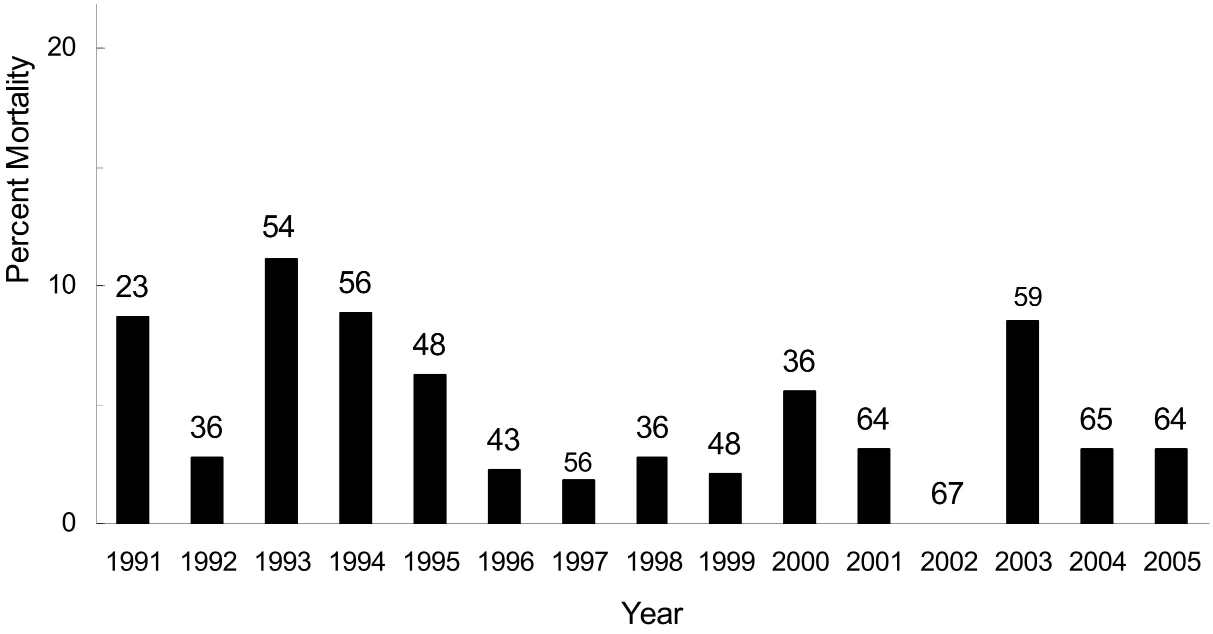


Figure 8. Percent non-winter (June-October) mortality of radiocollared, adult (≥1.0 yr old) female white-tailed deer (4 sites pooled), north-central Minnesota 1991–2005. (Sample sizes are above bars.)

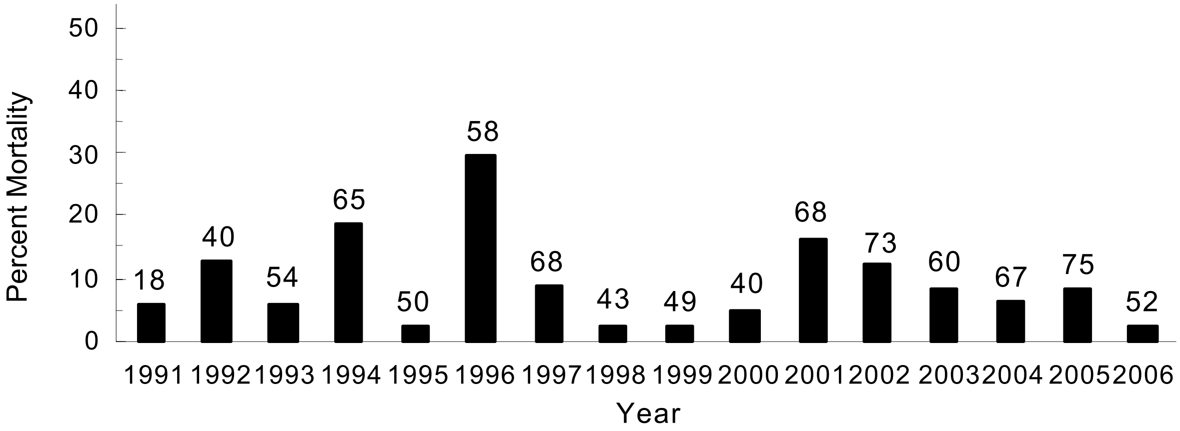


Figure 9. Percent winter mortality (December–May) of radiocollared, adult (≥1.0 year old) female white-tailed deer (4 sites pooled; sample sizes are above bars), north-central Minnesota, winters 1990–1991 to 2005–2006. (1990 = winter 1990–91, 1991 = winter 1991–92, etc…)

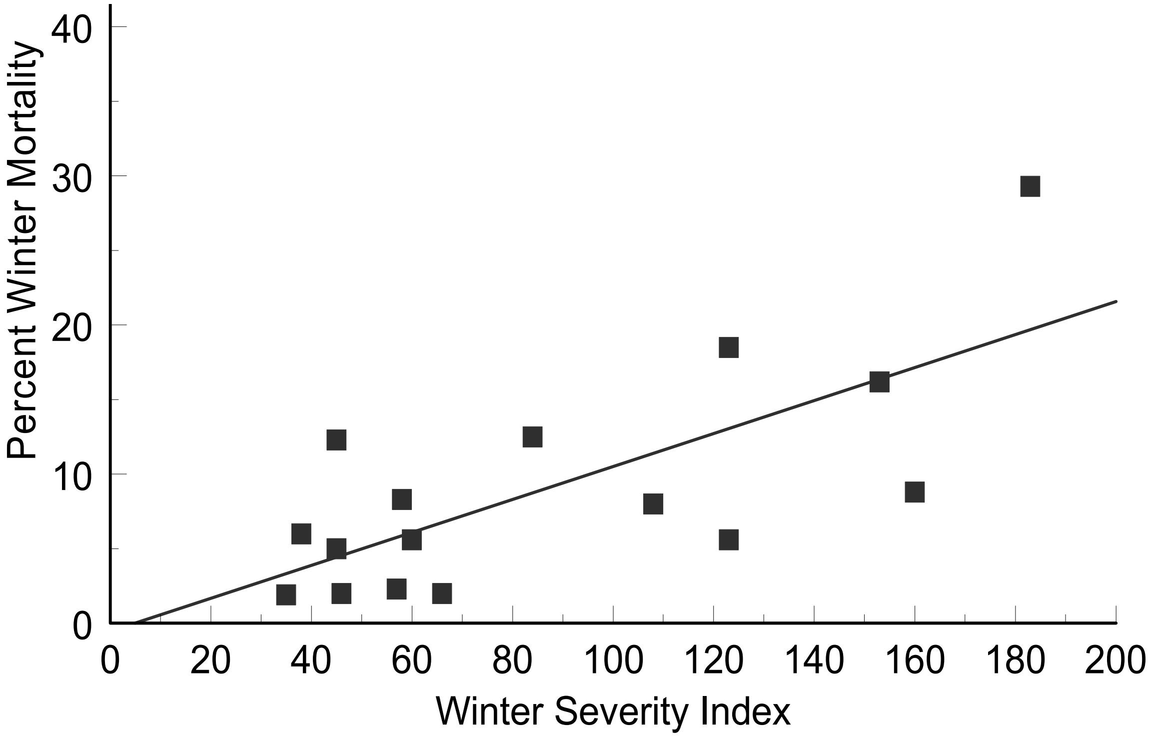


Figure 10. Relationship between Minnesota Department of Natural Resources' winter severity index (November–May) and percent winter (December–May) mortality (Y = -0.5511 + 0.1106*x*, r^2 = 0.52, *P* = 0.002) of radiocollared, adult (≥1.0 year old), female white-tailed deer (4 sites pooled), north-central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

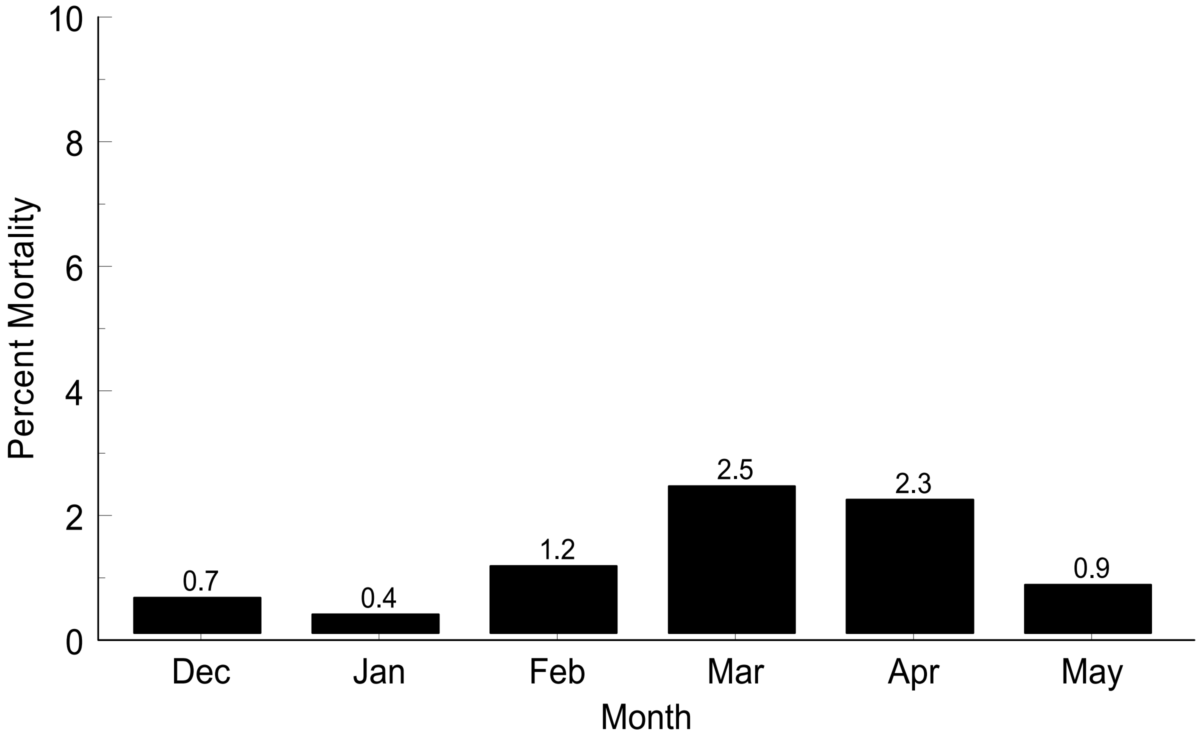


Figure 11. Monthly mortality of radiocollared female (fawns and adults) white-tailed deer caused by wolves (4 sites pooled), north-central Minnesota, winters 1990–91 to 2005–06.

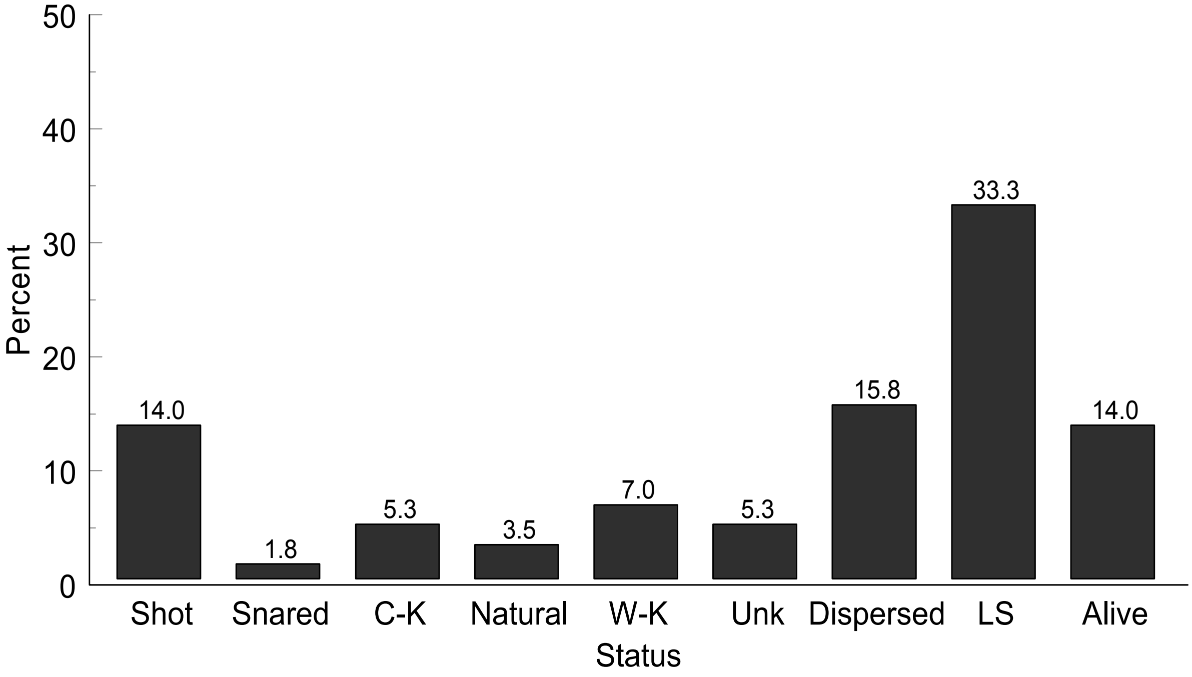


Figure 12. Status of radiocollared wolves, north-central Minnesota 1993–2006. (C-K = car-kill, Natural = natural causes, W-K = wolf-kill, Unk = unknown cause, LS = lost signal.)

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.

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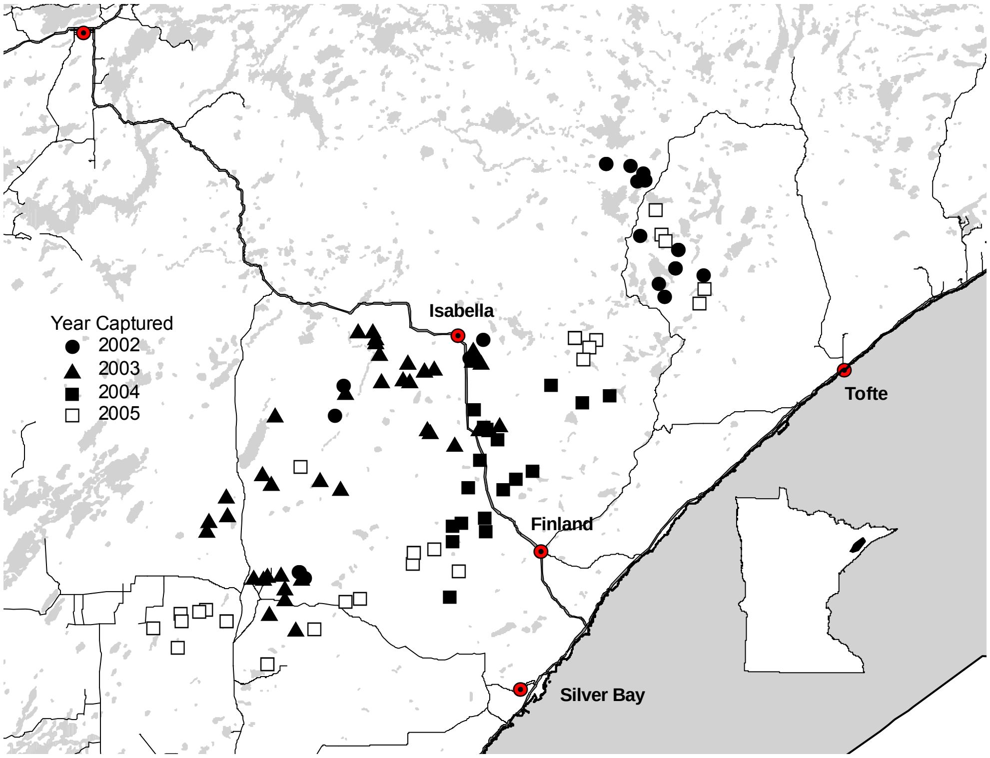
Table 1. Annual non-hunting and total mortality of collared moose. Number of collared moose in sample at beginning of calendar year is listed in parentheses.

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- × 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 $n_{\text{nonoverlap}}$ = 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.

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

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

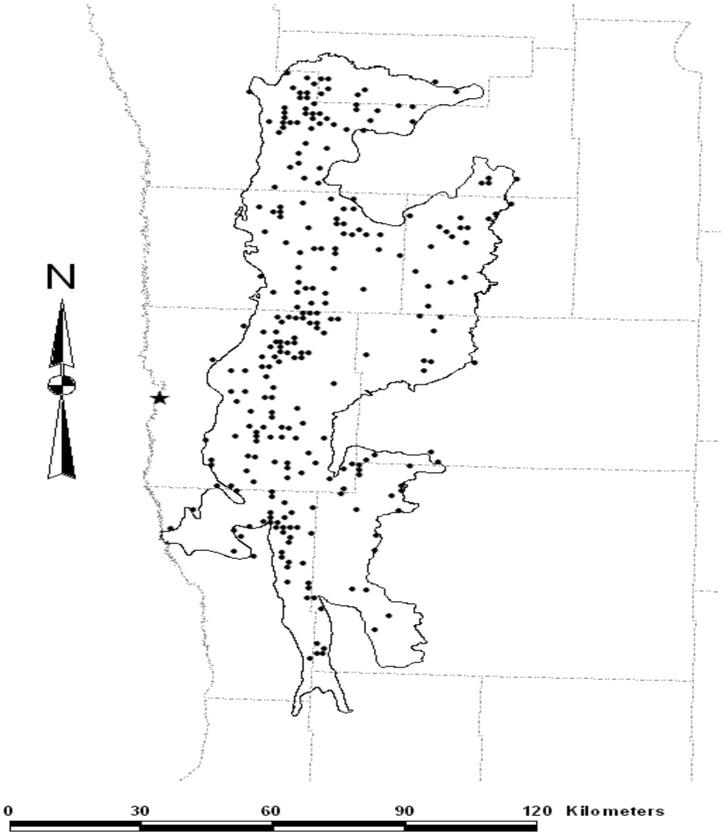

Figure 1. The Northwest prairie-chicken range based on land type associations of the Ecological Classification System (solid line) relative to county boundaries (dashed lines) in western Minnesota. Sample plots (dots) were not selected from areas >90 km southeast of Moorhead (star).

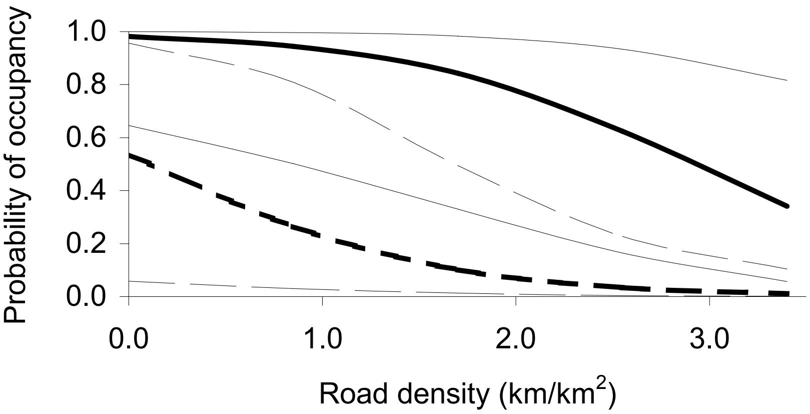

Figure 3. Predicted probabilities (heavy lines) and 95% confidence intervals (light lines) of a sample plot in Minnesota being occupied by a prairie-chicken lek during spring of 2005 over the observed range of road densities in the overlap domain (i.e., plots known to have contained a lek during 2004;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

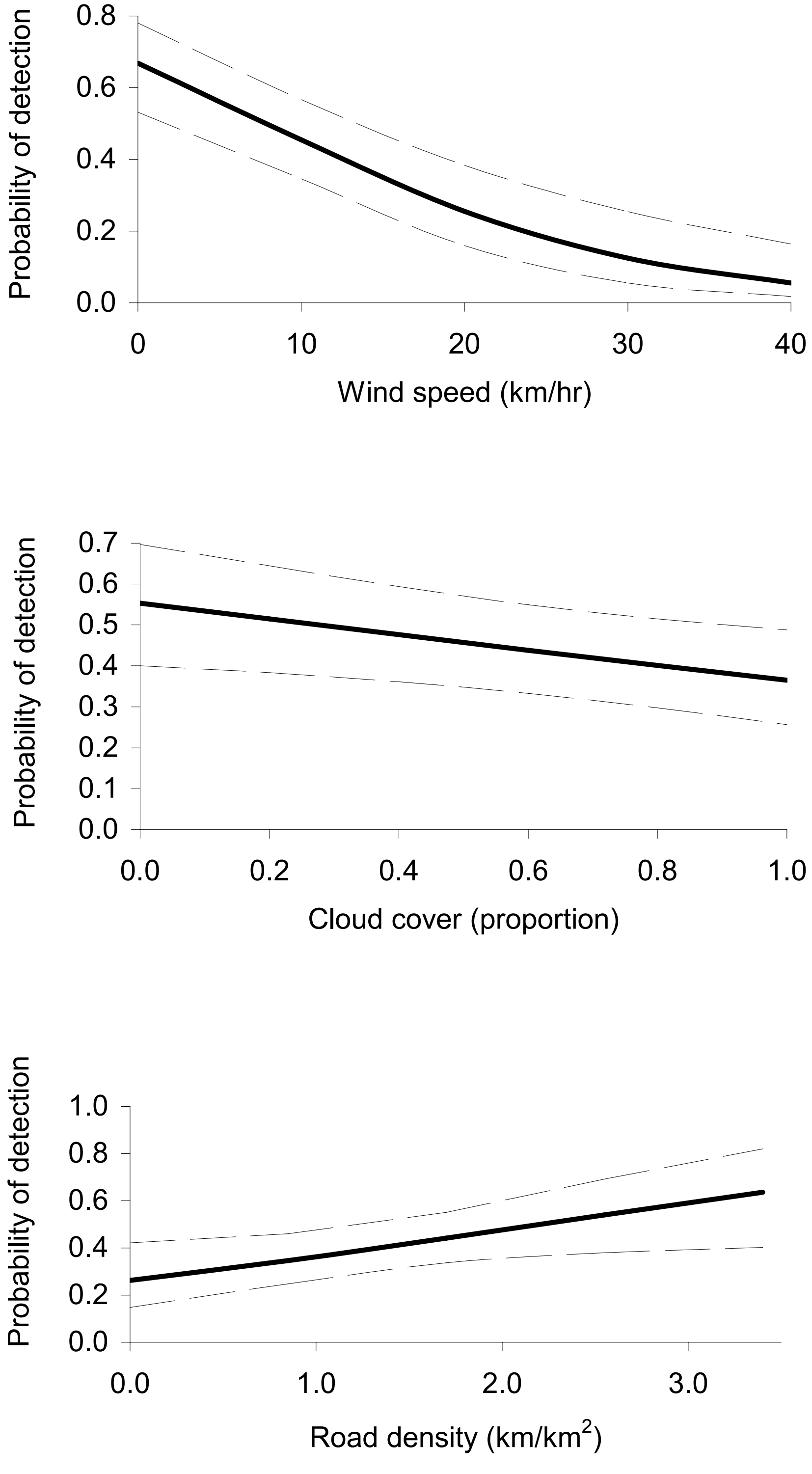

Figure 2. Predicted probabilities (and 95% confidence intervals) of detecting a prairie-chicken lek in sample plots in Minnesota during spring of 2005 over the range of observed values of selected model parameters.

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 2.5th and 97.5th 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 × 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; $\bar{r} = 0.67$, 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 ($\bar{r} = 0.59$, 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 ($\bar{r} = 0.24$, $n = 4$). Correlations were least between annual mean counts in the Paleozoic Plateau section and the other sections ($\bar{r} = 0.03$, $n = 6$).

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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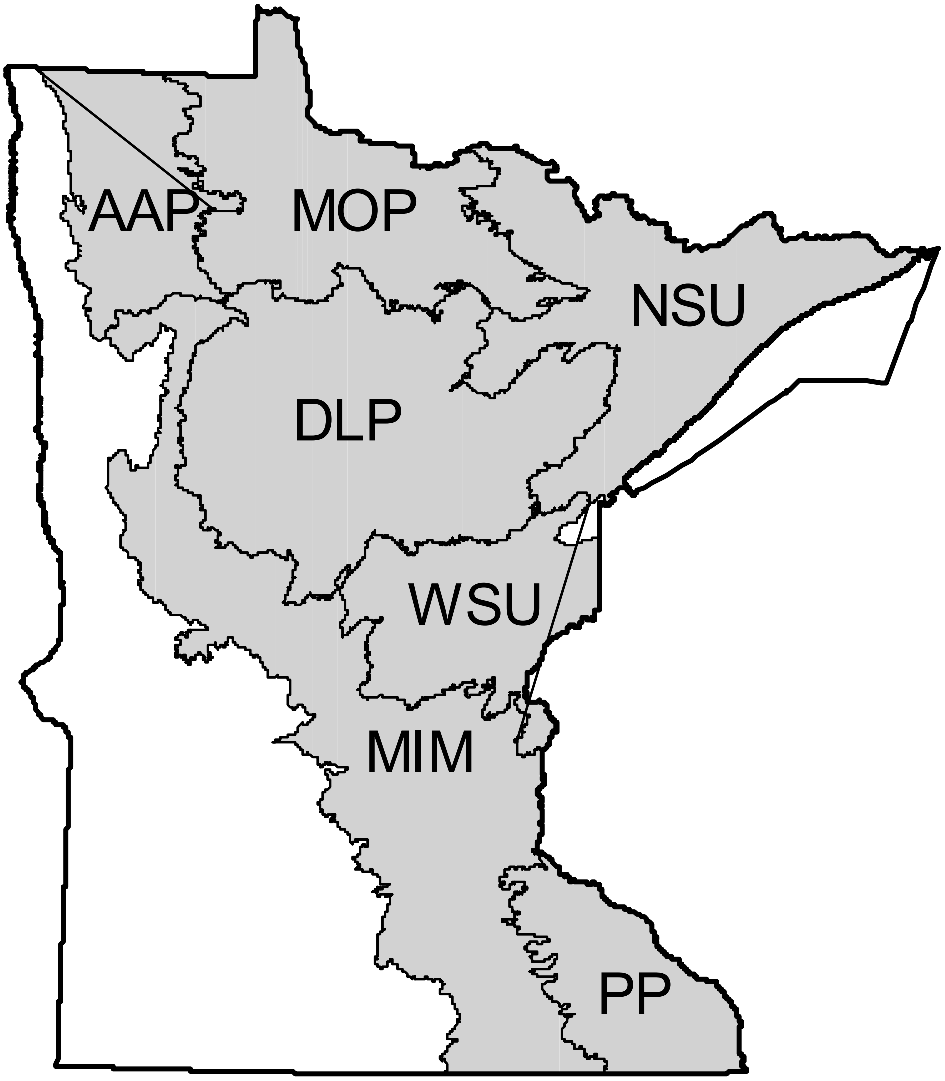


Figure 1. Forested sections of the Ecological Classification System in Minnesota. The MOP, NSU, DLP, and WSU sections constitute the Laurentian Mixed Forest province. AAP = Lake Agassiz & Aspen Parklands, MOP = Northern Minnesota & Ontario Peatlands, NSU = Northern Superior Uplands, DLP = Northern Minnesota 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.

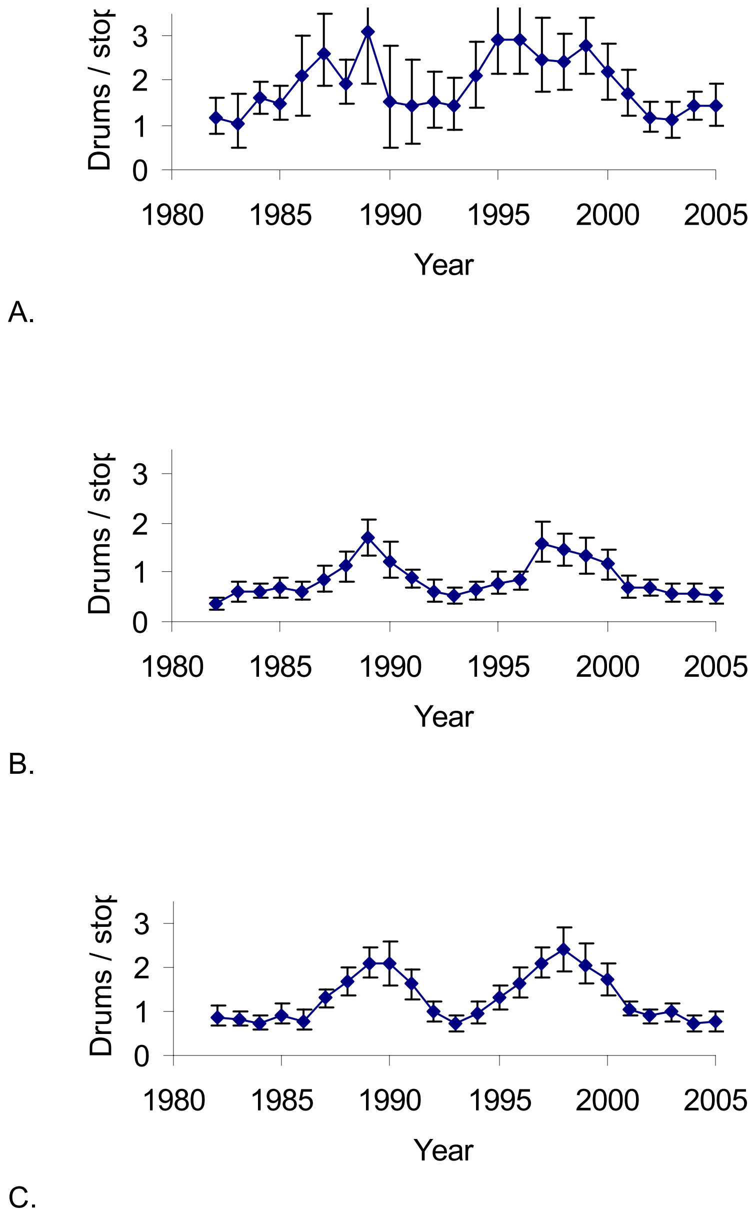
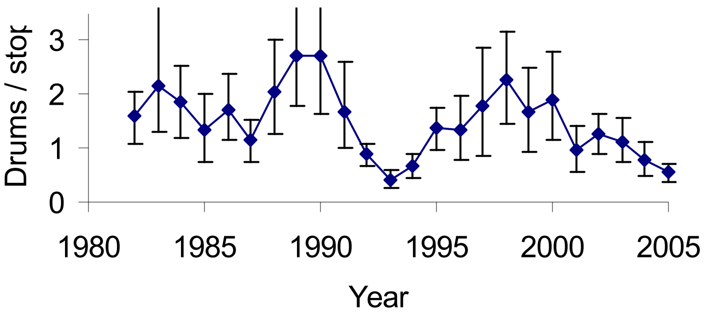


Figure 2. Ruffed grouse drum count index values in the 7 forested ECS sections in Minnesota (Panel A: MOP, panel B: NSU, panel C: DLP, panel D: WSU, panel E: AAP, panel F: MIM, panel G: PP; abbreviations explained in the text and the caption for Figure 1). Vertical error bars represent 95% confidence 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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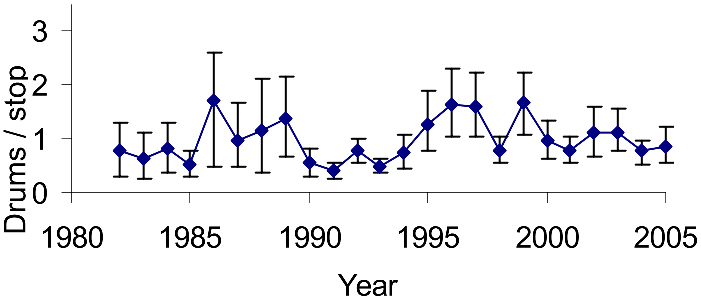
D.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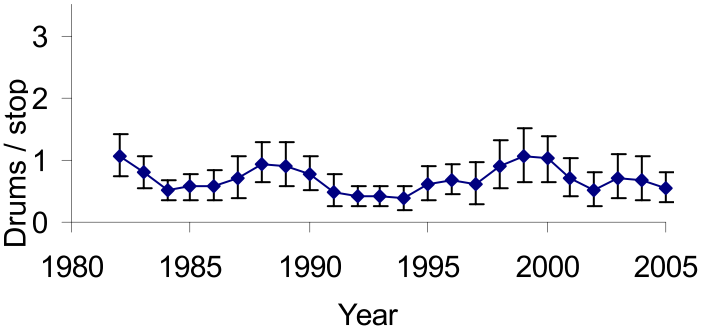
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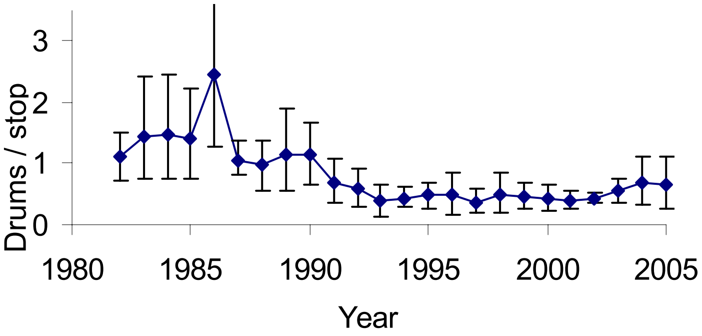
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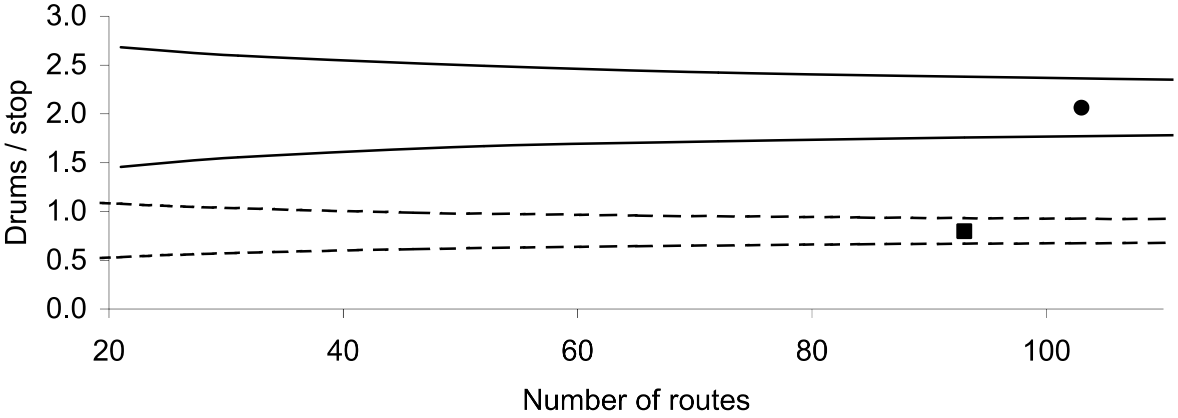

Figure 3. Ninety-five percent confidence intervals for bootstrapped samples of different numbers of routes in the Laurentian Mixed Forest province of Minnesota using ruffed grouse survey data from 1998 (solid lines) and 2004 (dashed lines), when mean drum counts were 2.06 (*n* = 103 routes, circle) and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

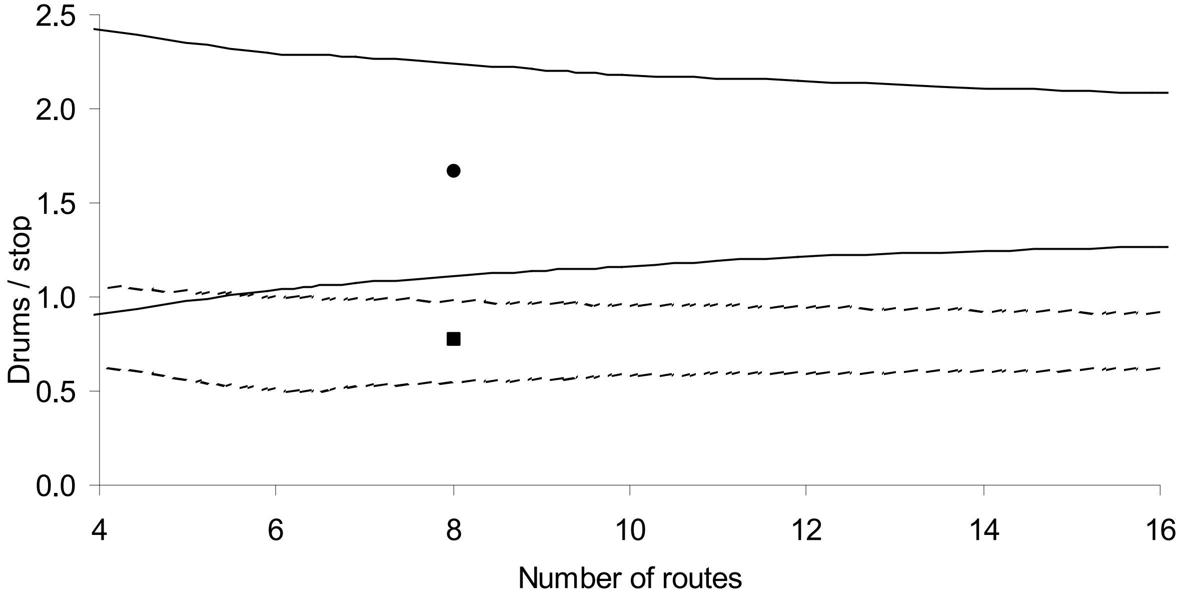

Figure 4. Ninety-five percent confidence intervals for bootstrapped samples of different numbers of routes in the Lake Agassiz and Aspen Parklands section of Minnesota using ruffed grouse survey data from 1999 (solid lines) and 2004 (dashed lines), when mean drum counts were 1.68 (circle) and 0.78 (square,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

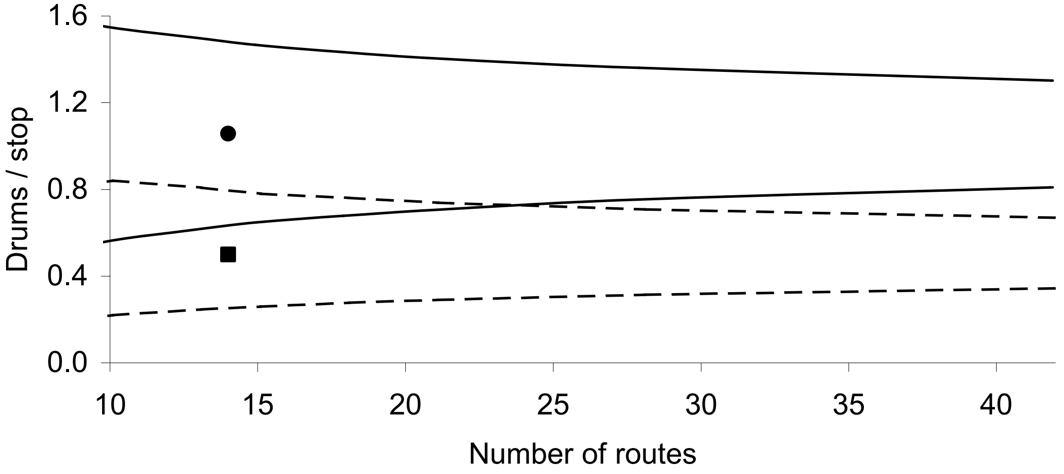

Figure 5. Ninety-five percent confidence intervals for bootstrapped samples of different numbers of routes in the Minnesota and Northeast Iowa Morainal section of Minnesota using ruffed grouse survey data from 1990 (solid lines) and 1993 (dashed lines), when mean drum counts were 1.06 (circle) and 0.50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

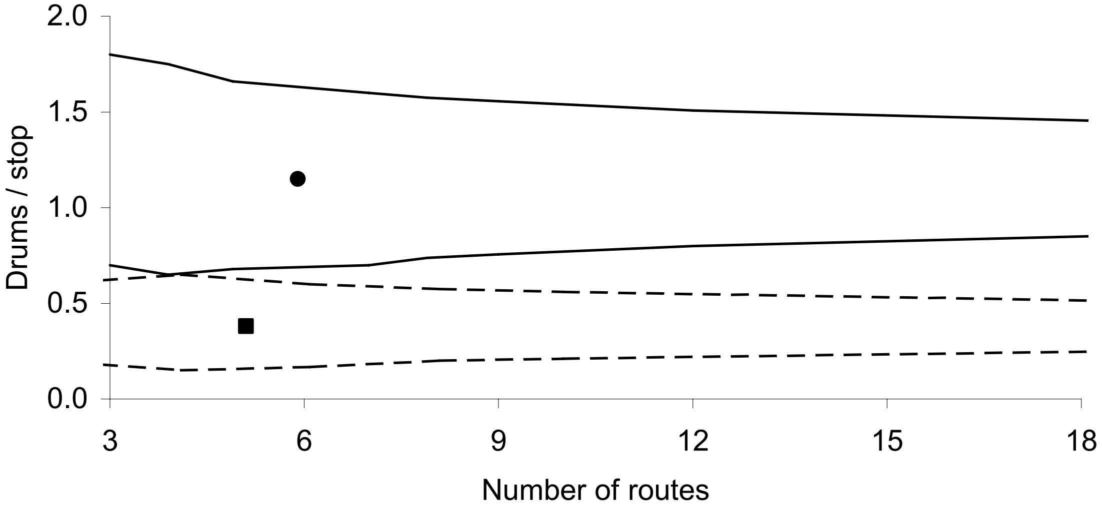

Figure 6. Ninety-five percent confidence intervals for bootstrapped samples of different numbers of routes in the Paleozoic Plateau section of Minnesota using ruffed grouse survey data from 1990 (solid lines) and 1993 (dashed lines), when mean drum counts were 1.15 (*n* = 6 routes, circle) and 0.38 (*n*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

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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/.

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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.

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