SUMMARIES OF WILDLIFE RESEARCH FINDINGS 2004

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

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USING DISTANCE SAMPLING TO ESTIMATE DENSITIES OF WHITE-TAILED DEER IN WATONWAN COUNTY, MINNESOTA

Michelle A. LaRue, Marrett D. Grund, Clayton K. Nielsen[1], Robert G. Osborn, and Brock R. McMillan[2].

SUMMARY OF FINDINGS

We present an alternative approach to estimating white-tailed deer (*Odocoileus virginianus*) density in Minnesota's southern, farmland region. We collected data from spotlight surveys to estimate deer density in Watonwan County, Minnesota. We estimated time required to collect field data, mean cluster sizes of deer, and mean distances of deer from observers. We then calculated population densities using the program DISTANCE (Thomas et al. 2002). We found a relationship between survey time and number of routes conducted indicating observers became more efficient at collecting data with increased experience. We observed more deer at greater distances during the post-hunt trial period than the pre-hunt trial period. As expected, the population density estimate was lower after deer were harvested during the hunting season. We recommend selecting survey routes using a randomized design to improve accuracy of density estimates derived by distance sampling.

INTRODUCTION

Numeric and geographic expansion of white-tailed deer populations provide increased recreational opportunities for hunters, but also create more challenges and problems for wildlife managers. Effective management decisions are predicated, in large part, on the number of deer within a permit area relative to the population goal. Wildlife managers in Minnesota's farmland region recommended the agency improve techniques to estimate and monitor deer abundance (Haroldson 2003). This management recommendation is congruent with previous research recommendations (Grund and Woolf 2004).

A variety of techniques are available to estimate populations of deer, and each has associated advantages and disadvantages. Most wildlife agencies use some analytical technique to evaluate harvest data, and these data form the basis of population assessment and trend analysis (e.g., population reconstruction (Roseberry and Woolf 1991), harvest-age-structure (Harris 1984), life table (Caughley 1977), and catch-per-unit effort (Lancia et al. 1996)). Most techniques for analyzing harvest data require age-at-harvest data (Roseberry and Woolf 1991). Age-at-harvest data are not routinely collected from hunter-killed deer in Minnesota, so options for population assessment are limited to simulation modeling (Grund and Woolf 2004) and field techniques. Available field techniques include aerial surveys (Potvin et al. 2005), collecting mark-recapture data and utilizing Schnabel estimators (Lopez et al. 2004), or Lincoln-Peterson estimators (McCullough and Hirth 1988). Osborn et al. (2003) used an aerial survey technique to recalibrate Minnesota's farmland model. Although using aerial surveys to recalibrate a population model is recommended (Grund and Woolf 2004), the technique is expensive and requires aircraft and staff to be available under certain snow conditions. Snow conditions limit the application of aerial surveys, in most years, to northern Minnesota during winter. Our intent was to provide an alternative framework for wildlife managers to estimate deer population

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size in areas and situations where aerial surveys may not be feasible or appropriate.

Distance sampling is a technique that has been increasingly used during the past 2 decades (Thompson et al. 1998). Distance sampling is based on the concept that not all animals will be observed during surveys due to visibility bias caused by visual impediments and observer error (Buckland et al. 1993). A detection function is generated which estimates how detection of objects changes with increasing distance from the observer. The detection function is used to estimate the area from which objects are observed from transects. Density is then computed as the number of animals observed divided by the area sampled. Thus, density can be estimated as $D = nf(0)/2L$ where D=density of animals of the surveyed area, n=number of animals observed, $f(0)$=value of the detection function of perpendicular distances, and L=the length of the transect traveled during surveys (Buckland et al. 1993). There are 3 assumptions associated with distance sampling: 1) all animals on transect lines are observed, 2) detected individuals or clusters of individuals are observed at their original location, and 3) sighting distances are measured without error. Buckland et al. (1993) provides a comprehensive review of the technique and Rivera-Milan et al. (2005) provides an application of the technique.

OBJECTIVES

- To collect field data on deer in Watonwan County to derive pre-hunt and post-hunt population densities using the program DISTANCE (Thomas et al. 2002); and
- To examine factors that may affect density estimates using distance sampling.

METHODS

We conducted 24 spotlight surveys in Watonwan County, MN, from 21 October – 28 December 2004. Twelve surveys were conducted prior to the regular firearms deer season (21 October – 4 November) and 12 surveys were conducted after the season (15 November- 28 December). During surveys, observers searched for deer using hand-held spotlights while a pickup truck traveled 2 east-west transects, approximately 40 km in length (25 miles; Figure 1) at speeds <32 km/hour (<20 miles/hour). Our starting point varied each night to reduce the probability of observing the same deer in the same places at corresponding times of surveys. Surveys began at dusk and were completed after traveling the 2 east-west transects. Survey start and end times ranged from 1700 – 2300 hours, respectively. Observers estimated distance to centers of deer clusters with a laser range finder and determined angles to centers of clusters using a prismatic compass. We defined a cluster of deer as a group of deer that were observed in the same field. Observer location (universal transmercator coordinates) at the time of each deer sighting was determined using a global positioning system receiver after distances and angles were recorded. We also recorded whether animals were observed in cropland, forest, or tall grass habitat types.

We performed a least-squares regression analysis with survey completion times and Julian date data to examine the relationship between observer experience and time spent afield. We used SAS (SAS Institute 1999) to calculate all descriptive and inferential statistics. A 2-way analysis of variance model was fitted to test survey period effects, habitat effects and their interactions on distance and cluster size data. We used the computer program DISTANCE to estimate detection functions, population densities, and precision. Density estimates and associated standard errors were adjusted to the proportion of woody habitat present in Watonwan County. To make this adjustment, land cover data were obtained and proportions of woody cover within the county were calculated using geographic information system maps within ArcView (Environmental Systems Research Institute 1999). Original population densities estimated by DISTANCE were then adjusted so that the sampled area represented the composition of woody cover in Watonwan County.

RESULTS

Time spent afield per survey during pre- and post-hunting periods was 4.4 (SD = 0.7) and 3.5 hours (SD = 0.4), respectively. There was a negative, curvilinear relationship (P<0.01) between time spent afield per survey and Julian dates (Figure 2).

Mean deer cluster size during pre- and post-season periods was 2.1 (SE = 0.1) and 2.9 deer/cluster (SE = 0.2), respectively. The survey period x habitat interaction effects for cluster sizes was significant ($F_{2,467}$ = 7.4, P < 0.001). There was a simple habitat effect during the post-hunt survey period ($F_{2,467}$ = 12.9, P < 0.01). Post hoc Tukey comparisons indicated that more deer (P < 0.05) were observed in tall grass habitat during the post-hut period than in forests or cropland habitats (Figure 3). The simple effect for habitat during the pre-hunt period was not significant ($F_{2,467}$ = 0.6, P > 0.05).

The survey period x habitat interaction effect for distances was not significant ($F_{2,467}$ = 1.7, P > 0.18). Post hoc Tukey HSD comparisons indicated distances from observer to clusters differed (P < 0.05) between seasons (pre-hunt = 128 m, SE = 5 and post-hunt = 145 m, SE = 7). Deer were also observed at greater distances (P < 0.05) in cropland habitats (mean = 153 m, SE = 6) than in forested (mean = 123 m, SE=11) or tall grass habitats (mean = 108 m, SE = 5; Figure 4).

We observed 259 clusters of deer (537 individuals) during the pre-season period and 215 clusters (620 individuals) during the post-season period. The unadjusted pre- and post-hunt population density estimates for Watonwan County were 7 ± 2 and 5 ± 2 deer/km^2, respectively (17 ± 4 and 14 ± 4 deer/mile^2, respectively). The adjusted pre-hunt density estimate was 2.5 deer/km^2 (6.4 deer/mile^2) and the adjusted post-hunt estimate was 2.0 deer/km^2 (5.3 deer/mile^2). These estimated densities are comparable to simulated output from Minnesota's farmland model in Permit Areas 457 and 458, both of which encompass Watonwan County (Figure 5).

DISCUSSION

We were encouraged that estimated densities were comparable between survey periods and the estimates were logical with the post-hunt density being lower than the pre-hunt density. It is noteworthy that we would have concluded that the population had increased after the hunting season if we just conducted spotlight surveys (537 deer observed during pre-hunt and 620 deer observed post-hunt). Probably due to crop harvest and leaf drop, we were able to observe more deer (n) during the post-hunt period. However, we also sampled a larger area (a) during the post-hunt period due to our ability to see farther distances. Thus, the estimated post-hunt population density (D) was lower than the pre-hunt density even though we observed more deer during the post-hunt period ($D = n/a$).

Our adjusted distance sampling estimates generally agreed with modeled deer densities in Permit Areas 457 and 458. The 2 distance sampling estimates are not directly comparable to modeling estimates because Permit Areas 457 and 458 extend beyond the boundaries of Watonwan County. Thus, we cannot conclude that the estimates derived from distance sampling were more accurate than modeling estimates. However, we believe an improved sampling design for Permit Areas 457 and 458 could produce accurate distance sampling estimates for those units. We needed to adjust the original distance sampling density estimates because we repeatedly over-sampled woody cover using our sampling design. We believe collecting field data in randomly-selected units or stratifying the survey routes by cover type within a study area would remedy this problem, and this sampling scheme should be evaluated.

We believe further evaluation of this technique is warranted due to the need to: 1) recalibrate Minnesota's farmland deer model for improved deer management, 2) estimate population sizes of white-tailed deer for research purposes, and 3) potentially estimate population sizes during non-winter months to respond to unforeseeable management crises, such as an outbreak of chronic-wasting disease.

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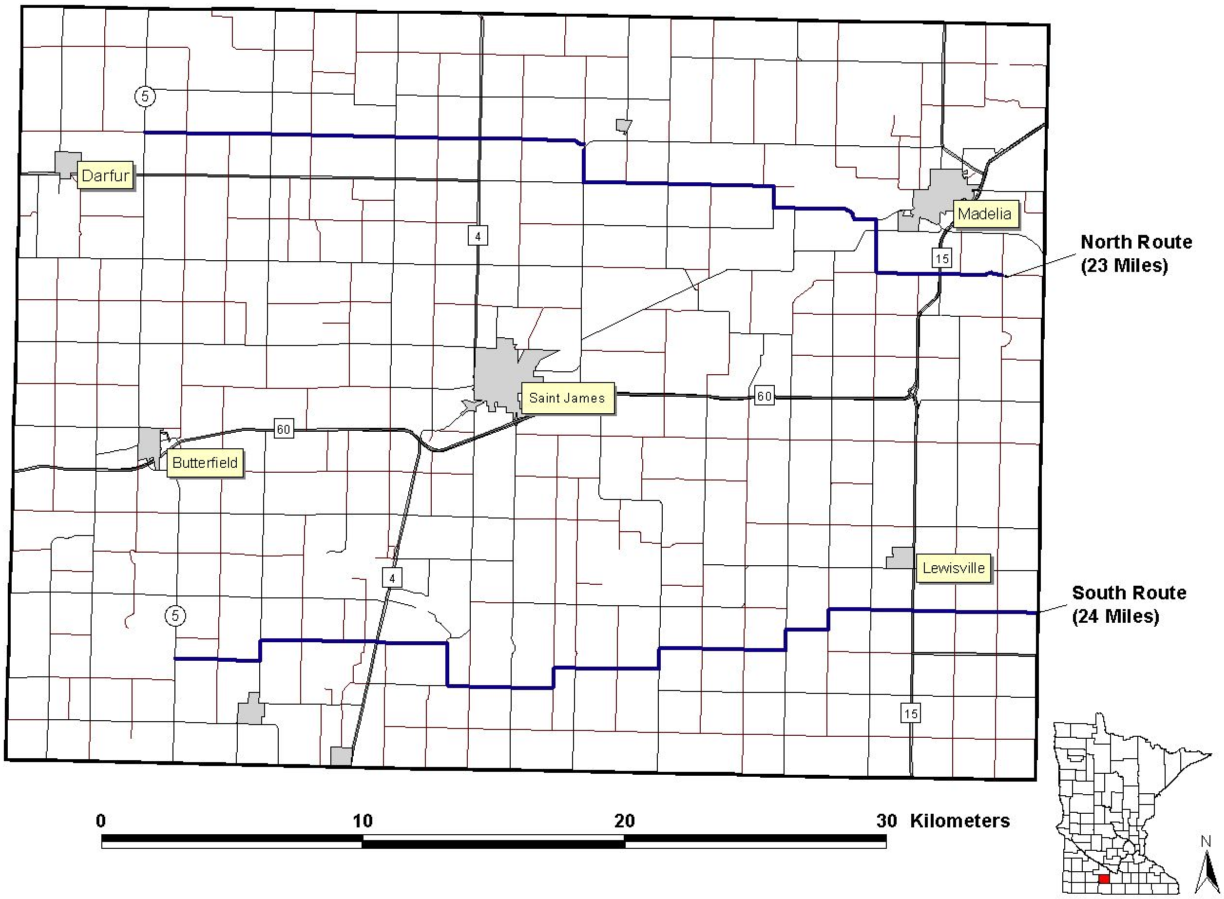

Figure 1. Transects driven in Watonwan County to collect field data for distance sampling analysis, Minnesota, 18 Oct – 28 Dec 2004.

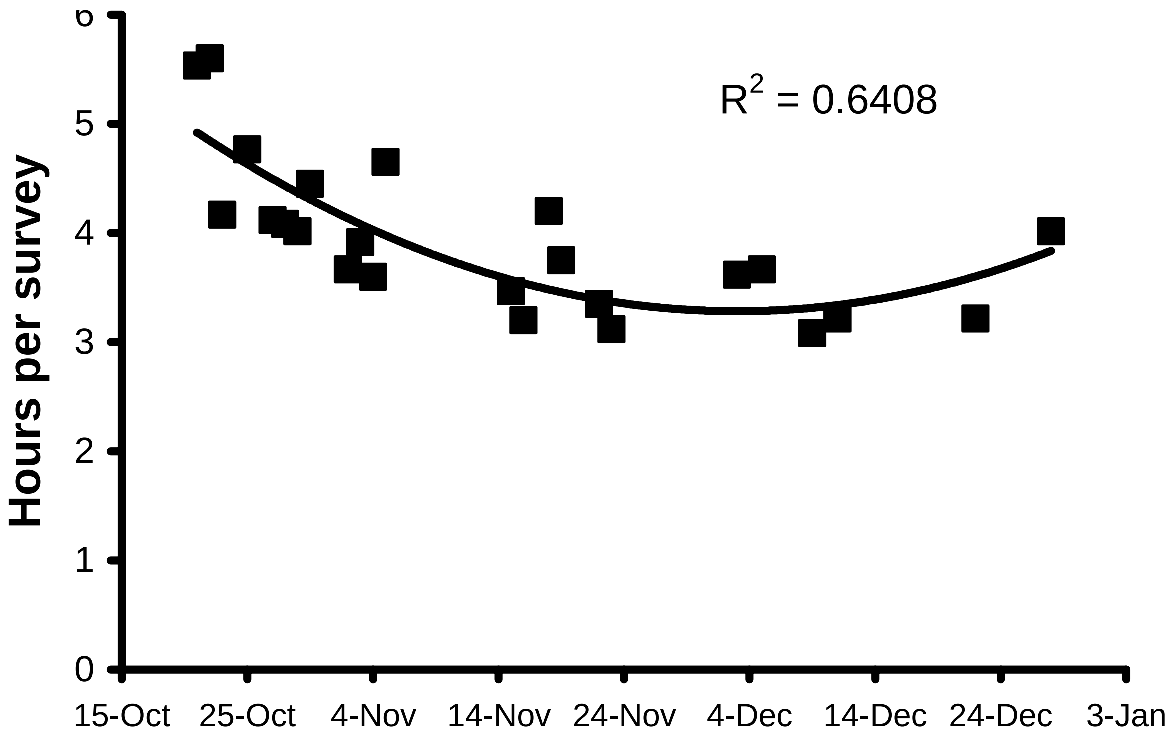

Figure 2. Time required to collect spotlight survey data on white-tailed deer versus Julian date Watonwan County, Minnesota, 2004.

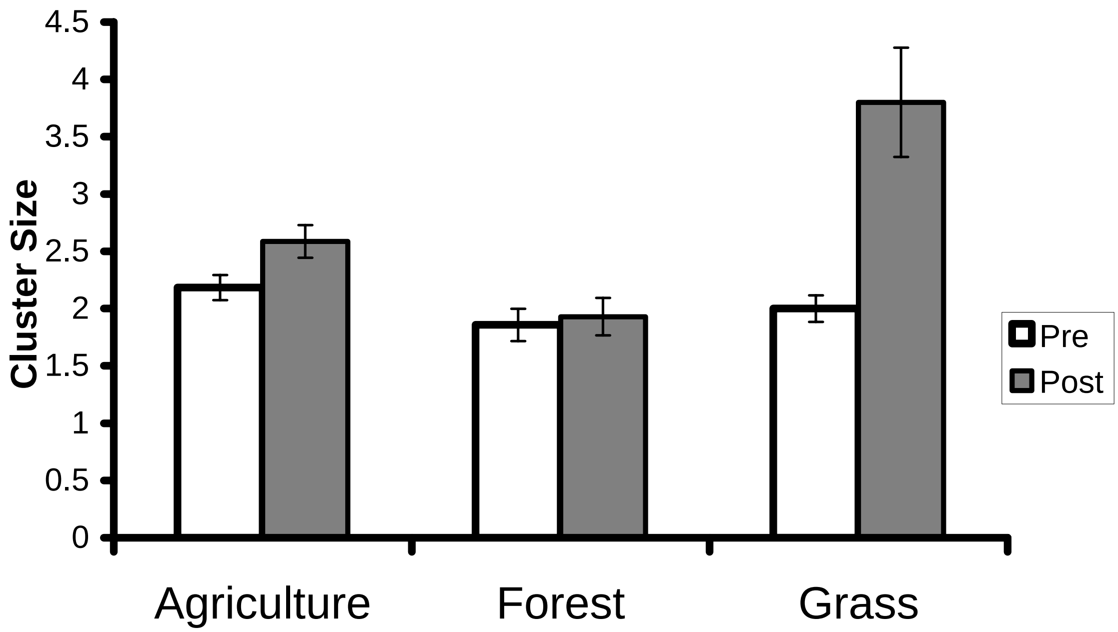

Figure 3. Mean cluster sizes (SE) of white-tailed deer by habitat type and survey period, Watonwan County, Minnesota, 18 October – 28 December 2004.

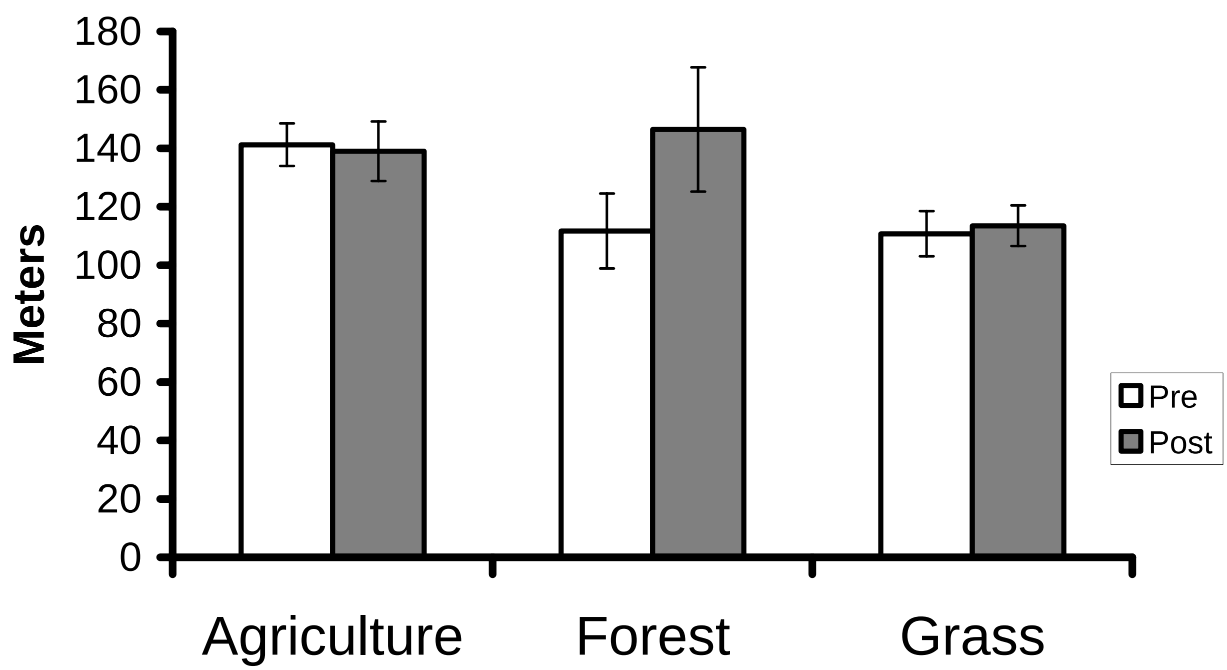

Figure 4. Mean distances (SE) between observers and deer by habitat type and survey period, Watronwan County, Minnesota, 18 October – 28 December 2004.

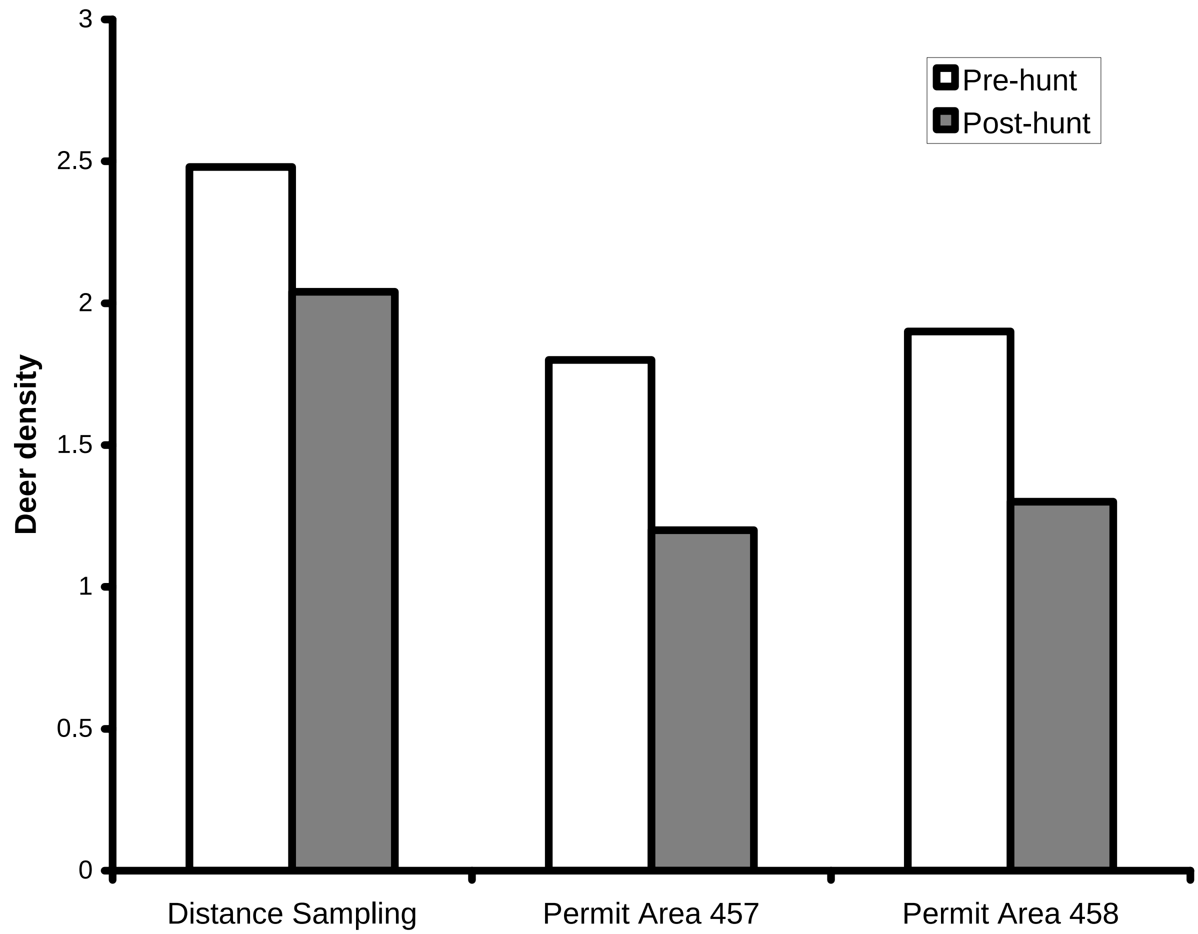

Figure 5. Comparison of deer densities (deer/km^2) estimated for Watonwan County through distance sampling and for Permit Areas 457 and 458 estimated through simulation modeling, Minnesota, 2004.

SIMULATING ANTLER POINT RESTRICTION REGULATIONS IN POPULATIONS OF WHITE TAILED DEER IN NORTHWEST MINNESOTA USING A GENERALIZED SUSTAINED YIELD MODEL.

Marrett D. Grund

SUMMARY OF FINDINGS

I developed a population model to simulate the effect antler-point restriction regulations (herein referred to as point rules) have on white-tailed deer (*Odocoileus virginianus*) populations and hunter harvests in northwest Minnesota. I used sustained yield theory as the foundation of the model so that the effect density-dependence has on deer populations could be evaluated. Six management strategies were modeled under low and high deer densities. Buck:doe ratios increased under point rules and were maximized when these regulations were coupled with high harvest rates of adult females. Buck harvests were variable, but generally decreased under point rules. Antlerless harvests responded more to herd size management strategies than to point rules. However, fewer antlerless deer were required to be harvested under a 4-point rule.

INTRODUCTION

Over the past 70 years, deer management has evolved from focusing on promoting deer population expansion through habitat protection, hunting regulations, and predator control, to serious concerns about how best to limit deer densities and the consequent impacts of deer on society (Conover 1997) and forest ecosystems (Garrott et al. 1993). Concurrent with managers' concerns regarding growth rates of deer populations, there is a growing interest by hunters, hunting organizations, landowners, and ultimately legislators to increase buck:doe sex ratios in deer populations. Most of these groups recommend some variation of selective harvest criteria on antlered deer such as point rules (Strickland et al. 2001).

A Senate bill passed during the 2004 legislative session mandating that the Department of Natural Resources (MNDNR) consider point rules in 5 northwestern Minnesota counties. Due to increasing interest of several organizations, individuals, and the legislation implied that the MNDNR would examine the issue. The MNDNR recognized the need to study social and biological impacts of alternative management strategies on Minnesota white-tailed deer populations. Consequently, I developed a simulation model to study effects point rules have on deer populations and hunter harvests in northwest Minnesota.

OBJECTIVE

- To develop a population model that will forecast effects various levels of point rules have on deer populations and harvest levels in northwest Minnesota.

METHODS

Population Model

I developed a computer model for simulating population dynamics of white-tailed deer under alternative harvest regimes. Similar to most big game models, my model explicitly tracked sex and age classes over time and was driven by 3 primary sets of variables representing harvest, recruitment, and natural mortality. I structured the model to generally reflect important biological periods during the life cycle of a white-tailed deer. Thus, the model operated in a similar fashion to many other deer population models (e.g., Walters and Gross 1972, Xie et al. 1999, Grund and Woolf 2004).

The model was partitioned into 3 distinct seasons of the year beginning with the pre-hunting season (June – early September), the hunting season (mid-September – December), and then the post-hunting season (January – May), (Figure 1.) The model simulated age, reproduction, and non-hunting survival for 4 sex and age classes (fawn male, fawn female, adult male, and adult female). Yearlings were separated from adults during the hunting season so that independent harvest rates could be applied to that cohort during that time period. For this paper, fawn refers to deer <1 years old and adult refers to deer >1 years old, except during the hunting season. During the hunting season, adult refers to deer >2 years old and yearling refers to deer approximately 1.5 years old.

Density-Dependent Vital Rates

The foundation of my model was based on a density-dependent relationship. Estimates for stage-specific, density-dependent parameters were derived from a generalized sustained yield model for white-tailed deer (Downing and Guynn 1985). Most parameters were stochastic so that variation in estimates could be modeled to simulate expected, temporal variation in population vital rates (Table 1).

Density-Independent Vital Rates

In Minnesota, winter severity has a direct and significant impact on deer populations by influencing winter mortality rates (DelGuidice 2003), and the winter weather parameter has substantial influence on model output (Grund and Woolf 2004). Verme (1977) also found that winter weather influenced fetal development during late gestation and, as a consequence, had a negative impact on natal survival. The impact winter conditions had on vital statistics were estimated and integrated into the model after density-dependent vital statistics were estimated.

To estimate winter mortality rates, I calculated winter severity indices (WSI) using minimum temperature and snow depth data from National Weather Service stations located throughout Minnesota from 1 January 1982 – 15 May 1999 (United States Department of Commerce, National Oceanic and Atmospheric Administration, 1999). The WSI value was calculated by summing the number of days temperature values were $<-17.8^{o}$ C and the number of days snow depths were >38 cm at each weather station for each month. These values were then spatially interpolated using the inverse distance weighting method based on the nearest 6 weather stations (Burrough and McDonnell 1998).

The model randomly generated WSI values based on a uniform distribution of probability categories representing mild, moderate, and severe winters for northwest Minnesota. Probability ranges for mild winters were based on the proportion of WSI values that were <100 from 1983-2000. The minimum WSI observed from 1983-2000 was the lowest value that could have been selected during simulations. Probability ranges for moderate winters were based on the proportion of WSI values that fell between 100-120 from 1983-2000. Probability ranges for severe winters were based on the proportion of WSI values that were >120 from 1983-2000. Maximum WSI values used in simulations was determined by the maximum WSI calculated from 1983-2000.

After the WSI value was randomly determined, winter mortality rates for adult female deer were estimated from a linear equation derived from a 14-yr winter mortality study conducted in north-central Minnesota (DelGuidice 2003). I assumed winter mortality for adult male deer was similar to adult female deer. I used linear equations based on a previous study to estimate winter mortality rates for fawns (Grund and Woolf 2004). Fawn summer survival was partitioned into a baseline survival rate derived from the sustained yield model and a neonatal survival rate based on the previous winter's WSI value (Verme 1977). The neonatal survival rate was then multiplied with the baseline survival rate to derive a composite fawn summer survival rate.

Harvest Management Strategies

Point rules are intended to protect some pre-determined proportion of antlered bucks during the hunting season. I examined antler and sex-age-kill data from deer at registration stations in Minnesota from 1993-2000 to determine protection levels (percentage of antlered deer protected by the point rule) of yearling and adult male deer under a 3-points-to-a-side (3-point) and 4-points-to-a-side (4-point) point rule in 4 deer management units (Tables 2 and 3). The model assumed harvest rates for antlered deer slightly increased as deer densities decreased (Roseberry and Woolf 1991). For this paper, I only modeled protection levels associated with yearling and adult bucks in northwest Minnesota.

There is not a direct link between decreasing harvest rates of adult males and increasing harvest rates of antlerless cohorts. Thus, managing antlerless harvests can be restricted or liberalized via antlerless license quotas in conjunction with point rules. I modeled adult female harvest rates that would achieve 2 different management goals over a 20-year period: 1) maintain stable population sizes (population size ±10% of population size in year 20), and 2) to reduce population size by 40-60%. Adult female harvest rates were randomly selected in 3 different range categories: low (10-20%), moderate (20-30%), and high (30-40%). Which range category was used to select an adult female harvest rate depended on where the population size was relative to its pre-determined goal. For example, the model would randomly select a value within the low harvest rate range if the population size was below the pre-determined population goal. The model would then simulate a population increase in response to a conservative management strategy of having a low harvest rate of adult females. Conversely, the model would select a value within the high harvest rate range if the population size was above the pre-determined population goal. Numerical fawn harvests were always about 50% of adult doe harvests. The composition of fawns in antlerless harvests fluctuates both across permit areas and years, but often comprises about one-half of the adult female harvest in Minnesota.

Initial Conditions and Simulations

I first determined initial sex and age compositions of deer populations by conducting 10-year simulations and calculating an average sex and age structure of the stable population for each modeling scenario. Initial sex and age compositions were adjusted based on the stable population then model runs were performed.

I allowed the model to simulate deer herd dynamics for 20 years to represent traditional rules (legal bucks had >3 inch antlers). Six management strategies were simulated beginning in year 21: 1) traditional rules with a management goal of maintaining a stable population size, 2) traditional rules with a management goal of reducing population size by 50%, 3) 3-point rules with a management goal of maintaining a stable population size, 4) 3-point rules with a management goal of reducing population size by 50%, 5) 4-point rules with a management goal of maintaining a stable population size, and 6) 4-point rules with a management goal of reducing population size by 50%. Each management strategy was simulated for 20 years during the second period (years 21-40).

Model simulations were performed with low and high deer densities to evaluate the effect density-dependence had on populations and hunter harvests. One simulation scenario had an initial population size starting at 30% of carrying capacity (carrying capacity=10,000 deer), and the second simulation had an initial population size starting at 90% of carrying capacity. Thus, 12 different simulation scenarios were modeled (6 strategies x 2 deer densities). I ran 500 simulations of each scenario so that different combinations of vital rates associated with stochastic model parameters were selected. Output generated by model runs were averaged and means were presented to depict population and harvest trends.

RESULTS

Population Output

Pre-hunt Deer Numbers—Regardless of management strategy and whether initial population size was low (Figure 1[a]) or high (Figure 1[b]), mean pre-hunt deer population sizes declined in similar fashions when the model simulated population reductions. Likewise, mean pre-hunt population sizes were similar when the model simulated stable population size strategies after year 20 (Figure 1).

Buck:Doe Ratios—Simulated post-hunt buck:doe ratios (fraction of adult males per adult female) differed substantially among management strategies for both low (Figure 2[a]) and high (Figure 2[b]) deer density simulations. Four-point rules had the most substantial effect on post-hunt buck:doe ratios (Figure 2). Further, post-hunt buck:doe ratios tended to be higher when deer populations were reduced rather than maintaining stable population sizes (Figure 2). A 3-point rule had minimal impact on simulated post-hunt buck:doe ratios when the population was high, and the objective was to maintain a stable population size (Figure 2[b]).

Harvest Output

Antlered Harvests—When deer populations were modeled at low densities, buck harvests were reduced in all simulations except for maintaining a stable deer population size under traditional rules (Figure 3[a]). Marked reductions in the buck harvest occurred during the initial year of point rules (year 21; Figure 3[a]). However, buck harvests increased in subsequent years as protected, yearling bucks matured to legal status as adults. Interestingly, mean buck harvests simulated under traditional rules with a management goal of population reduction were greater than mean buck harvests under a 3-point rules with population reduction. However, it was lower than mean buck harvests occurring when a 4-point rule was simulated and the management goal was population stability (Figure 3[a]).

No consistent trend in buck harvests was observed under any of the regulations associated with high deer densities (Figure 3[b]). When a high population density was reduced under traditional rules, buck harvests temporarily increased as pre-hunt densities declined (Figure 3[b]). Under 3-point rules, buck harvests temporarily increased for 1 or 2 years, but then buck harvests declined as pre-hunt population sizes were reduced. A marked reduction in buck harvest was apparent during the initial year of 4-point rules, but then buck harvests corresponded to changes in pre-hunt population sizes as well.

Antlerless Harvests—Antlerless harvests responded more to managing deer population sizes toward pre-determined goals than to point rules (Figure 4). However, it is noteworthy that the number of antlerless deer required for harvest was lower when 4-point rules were simulated compared to 3-point or traditional rules.

DISCUSSION

Pre-hunt population trends were similar among management strategies. This should be encouraging to wildlife managers. This suggests deer population dynamics occurring under alternative management strategies should mimic deer population trends observed in the past under traditional regulations. Thus, the public should not expect a change in population trends if point rules are adopted in the future.

As expected, buck:doe ratios increased when yearling bucks were protected from harvest. However, it is noteworthy that buck:doe ratios were higher when deer populations were reduced rather than maintained at original densities. This was a result of increasing the antlerless harvest rate to reduce population size. As a result of increasing the harvest rate on adult females, a smaller percentage of adult females existed in the post-hunt population. Consequently, a reduced number of adult females coupled with an increased number of adult bucks in the population under 4-point rules thereby maximized post-hunt buck:doe ratios. Managers concerned about skewed adult sex ratios and associated biological effects (Ditchkoff et al. 2001) should therefore liberalize antlerless harvest opportunities in concert with implementing point rules to maximize the potential of adjusting adult sex ratios.

With the exception to a marked reduction in buck harvest during the initial year of a 4-point rule, simulated buck harvests showed no consistent trends under any of the regulations. This should be somewhat alarming to managers wishing to provide the public some expectation of how much buck harvests might change under point rules. Perhaps even more concerning is the fact that this model only considered the effect age has on protecting bucks. Soil fertility and nutrition also effect antler development (Wood and Tanner 1985, Strickland and Demarais 2000). Variation in these factors would create even more variability in percentages of legal bucks vulnerable to harvest thereby making an accurate prediction of buck harvest almost impossible.

In the farmland region, managers should cautiously interpret buck harvests as an index to deer density even under traditional rules. Buck harvests increased as a result from reducing a high deer population density. This occurred because recruitment was stimulated as the population density reached about 50% of carrying capacity. This suggests that density-dependence effects may confound the straightforward interpretation of a relationship between buck harvest trends and deer population size (McCullough et al. 1990).

Antlerless harvests increased as a result of simulating higher antlerless harvest rates to achieve population reduction. In practice, increasing antlerless harvest rates has been challenging for a variety of reasons (Brown et al. 2000). An interesting finding from this modeling was that the number of antlerless deer required for harvest was slightly lower when point rules were implemented. This was likely due to density-dependent effects on population growth and the concept of sustained yield theory (Caughley 1977, McCullough 1979, Downing and Guynn 1985, Lancia et al. 1988). Under sustained yield theory, reproduction and mortality are negatively affected by increased population abundance. Thus, assuming principles of sustained yield theory operate in hunted deer populations, protecting bucks would negatively affect population reproductive and mortality rates. As a result, this may reduce the number of antlerless deer required for harvest to manage populations at a particular goal density. Whether this concept is true outside of a computer model warrants testing if managing antlerless harvests becomes a future concern to managers.

Possibly the most important, but least conspicuous finding from this modeling relates to comparing results from other wildlife agencies to those that could occur in Minnesota. Even within Minnesota, protection levels associated with point rules vary spatially, and the impacts different protection levels have on deer populations and harvests, particularly antlered harvests, can be substantial. Protection levels associated with different point rules likely differ in other areas of the United States where these regulations have been tested. Further, some wildlife agencies choose to manage deer populations near carrying capacity while others manage deer densities at or below a density that corresponds to maximum sustained yield. As discussed, where the deer population is relative to carrying capacity, and how population size is managed after implementation affects population and harvest trends. In addition, factors not considered in my model such as hunter density, land use patterns, hunter access, and deer accessibility vary from state-to-state, which also influences the outcome of alternative hunting regulations. Thus, it is not appropriate to expect similar population or harvest patterns from alternative management strategies employed in other states to occur in Minnesota. Expectations associated with alternative management strategies for

Minnesota should be based exclusively on data from and models developed for Minnesota.

FUTURE WORK

Additional modeling will include: 1) performing sensitivity analyses to systematically evaluate how changes in model parameters affect simulated output, 2) modeling different levels of "deer refugia" to determine how limited hunter access may affect alternative harvest regulations, 3) modeling additional alternative regulations such as earn-a-buck and buck lottery regulations, and 4) performing simulations in other deer management units where sex-age-kill and antler data have been collected.

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Table 1. Density-dependent parameter estimates used in the white-tailed deer population model to evaluate effects of altering harvest survival rates of antlered and antlerless deer.

Parameter	Predictive equation[a]	Range	References
Non-hunting survival			
Adult male	$(x^2*[3*10^{-9}]) + (x*[1*10^{-5}]) + 0.95$	Predicted value ± 0.05	Downing and Guynn (1985)
Adult female	$(x^2*[6*10^{-10}]) + (x*[6*10^{-6}]) + 0.96$	Predicted value ± 0.05	Downing and Guynn (1985)
Fawn	$(x^2*[6*10^{-9}]) + (x*[1*10^{-5}]) + 0.88$	Predicted value ± 0.10	Downing and Guynn (1985)
Recruitment rate			
Adult	$(x*[1*10^{-4}]) + 2.19$	Predicted value * y^b	Downing and Guynn (1985) McCullough 1979
Fawn	$(x*[1.6*10^{-4}]) + 1.12$	Predicted value * y^b	Downing and Guynn (1985) McCullough 1979
Sex ratio	$(x*[2*10^{-5}]) + 0.404$	Predicted value ± 0.04	Downing and Guynn (1985)

[a] x = population size
[b] y = random value between 0.8 — 1

Table 2. Sample sizes, mean maximum number of points to a side (SD), and protection levels of yearling males under a 3-point and 4-point rules in farmland deer management units. Antler data were collected periodically from 1993-2001. Means that have different letters are statistically different (P < 0.05) according to Tukey HSD comparisons.

Deer Management Unit	*n*	Average	Standard Deviation	3-Point Protection (%)	4-Point Protection (%)
Big Woods					
Central	331	2.77A	1.0	40	72
North	710	2.99B	1.0	30	67
Southeast	1,611	3.03B	1.0	28	65
Mille Lacs	196	2.67A	1.0	44	80
Northwest[a]	885	2.39D	1.0	56	84
Prairie					
North	216	2.97AB	1.0	29	68
River	1,287	2.97B	1.0	30	67
South[b]	500	3.15B	1.0	23	60

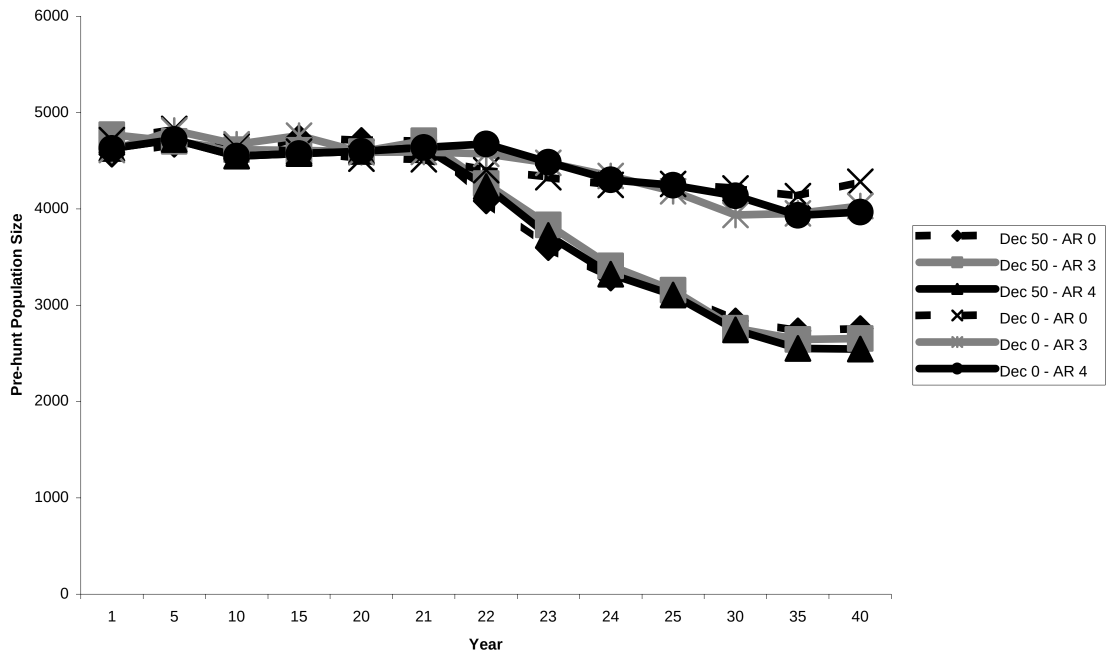
[a] Includes Red River and Agassiz Deer Management Units.
[b] Includes Southeast and Southwest Deer Management Units.

Table 3. Sample sizes, mean maximum number of points to a side (SD), and protection levels of adult males under a 3-point and 4-point rules in farmland deer management units. Antler data were collected periodically from 1993-2001. Means that have different letters are statistically different (P < 0.05) according to Tukey HSD comparisons.

Deer Management Unit	*n*	Average	Standard Deviation	3-Point Protection (%)	4-Point Protection (%)
Big Woods					
Central	132	4.33AB	1.0	2	13
North	316	4.76CDE	1.4	1	6
Southeast	853	4.39CB	1.0	2	10
Mille Lacs	92	4.10A	1.2	7	24
Northwest[a]	560	4.09A	1.1	8	20
Prairie					
North	102	4.50AD	0.9	3	7
River	393	4.24AE	0.8	2	11
South[b]	217	4.32A	0.8	1	9

[a] Includes Red River and Agassiz Deer Management Units.
[b] Includes Southeast and Southwest Deer Management Units.

(a



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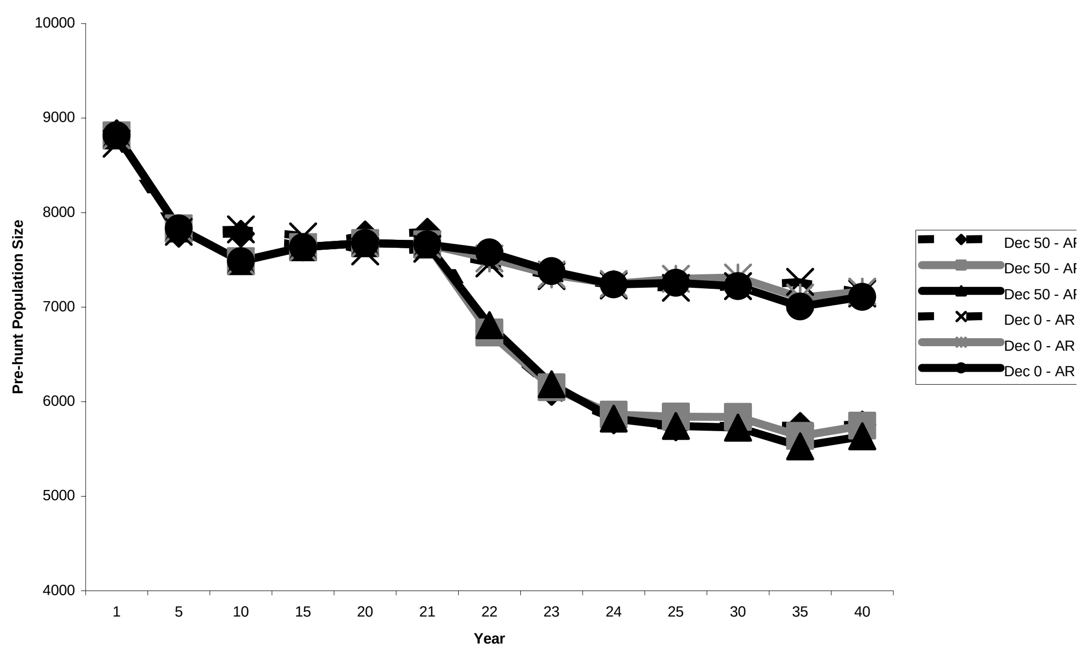


Figure 1. Temporal pre-hunt population trends of deer exposed to 6 different management strategies (Dec 50=50% population decline, Dec 0= Stable population size, AR0=Traditional point rules, AR 3=3-point rule, AR 4=4-point rule) in low (a) and high (b) deer densities, northwest Minnesota.

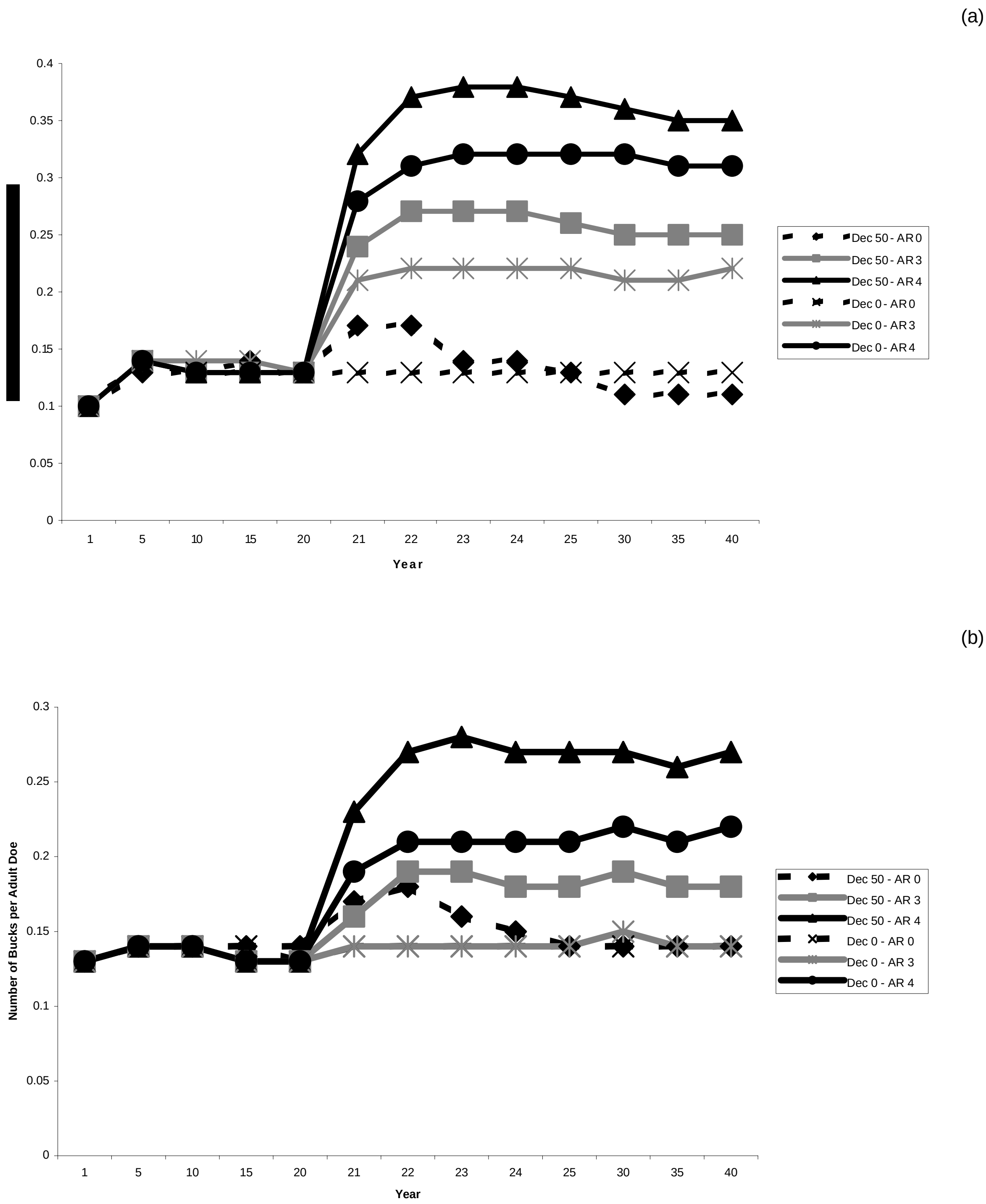
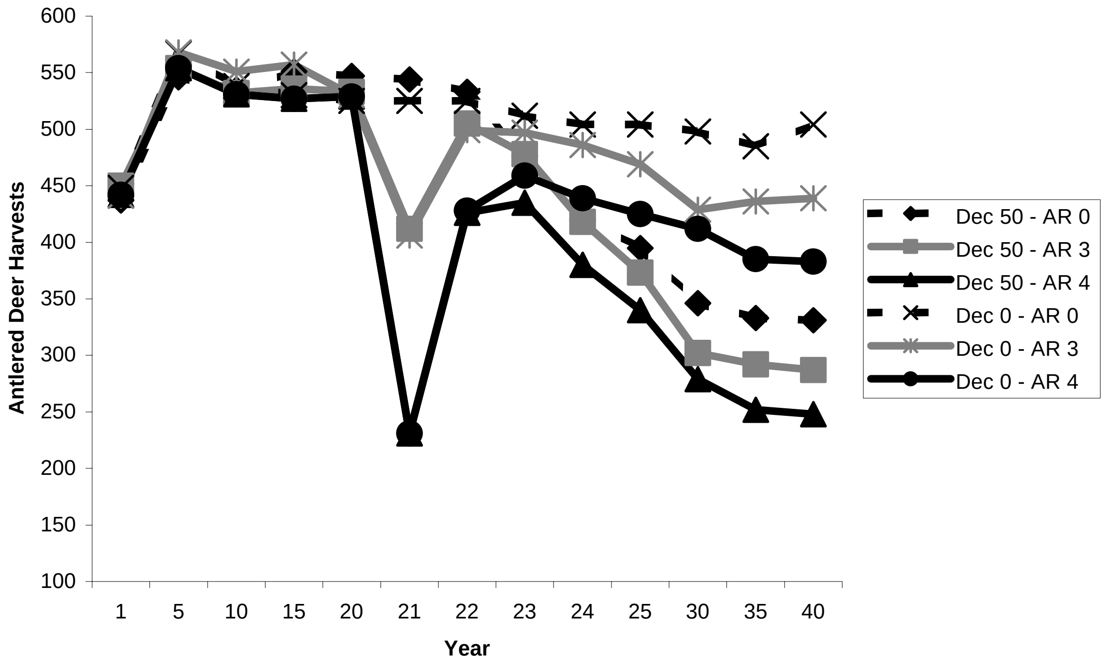


Figure 2. Temporal post-hunt buck:doe ratio trends of deer exposed to 6 different management strategies (Dec 50=50% population decline, Dec 0= Stable population size, AR0=Traditional point rules, AR 3=3-point rule, AR 4=4-point rule) in low (a) and high (b) deer densities, northwest Minnesota.

(a)



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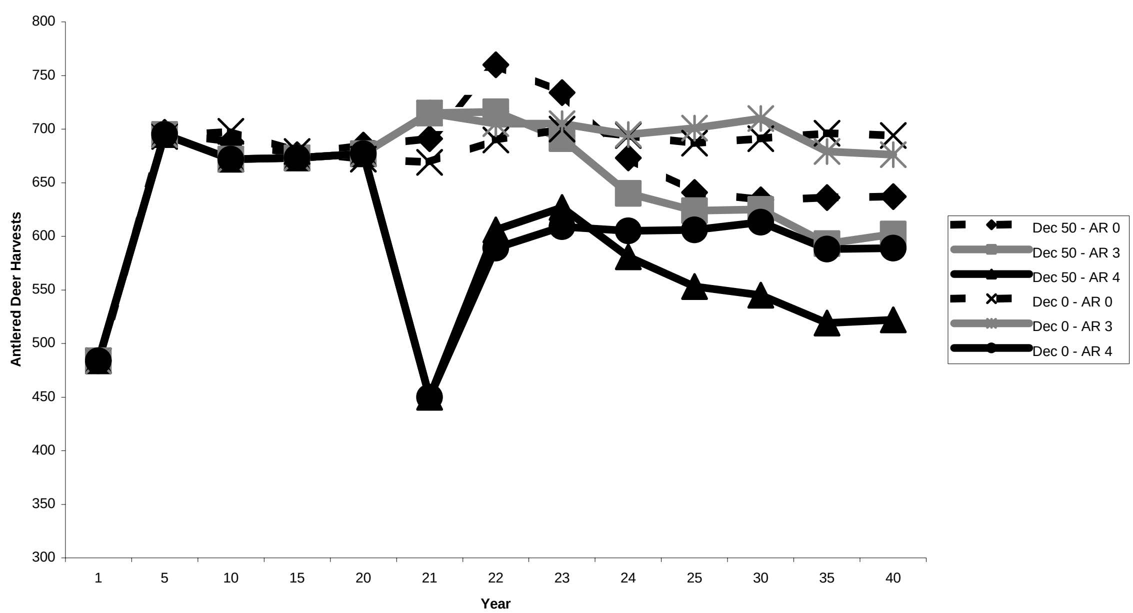
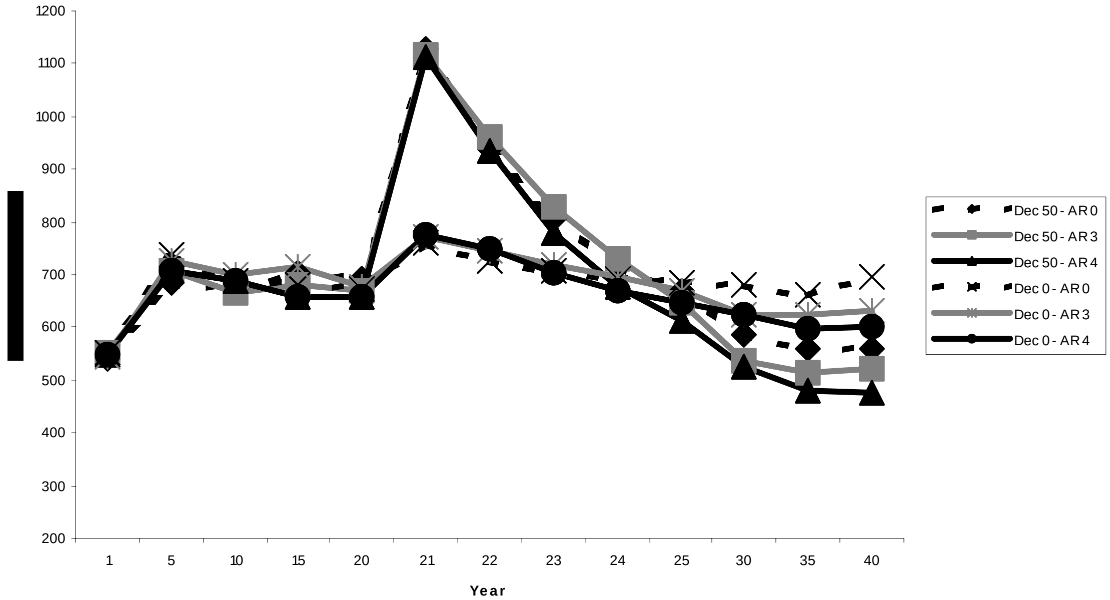


Figure 3. Temporal buck harvest trends of deer exposed to 6 different management strategies (Dec 50=50% population decline, Dec 0= Stable population size, AR0=Traditional point rules, AR 3=3-point rule, AR 4=4-point rule) in low (a) and high (b) deer densities, northwest Minnesota.

(a)



(b)

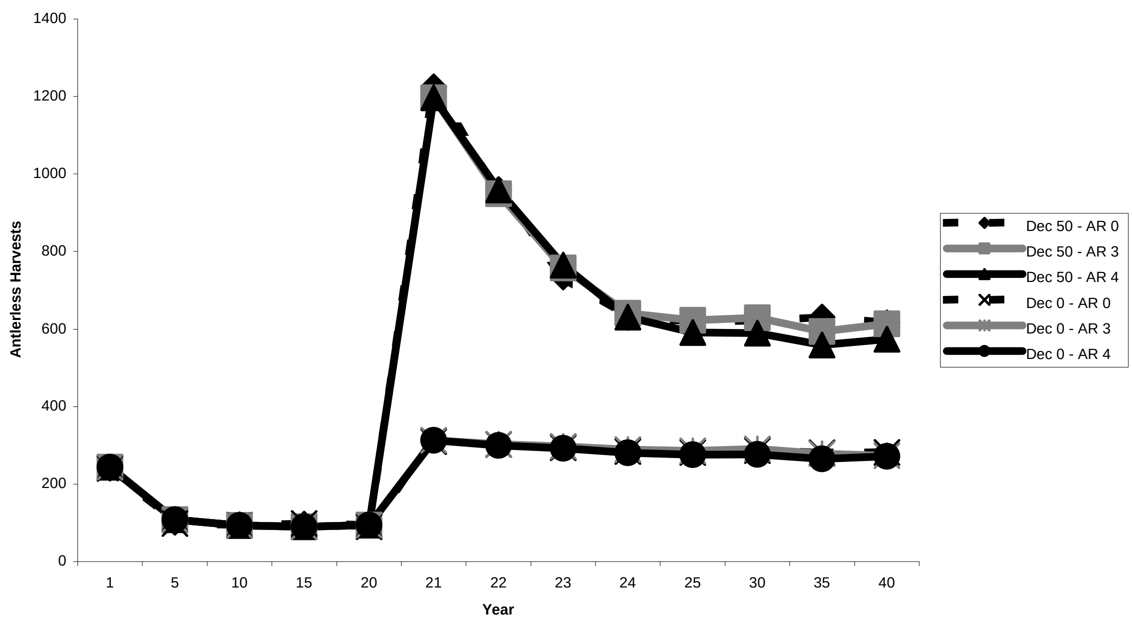


Figure 4. Temporal antlerless harvest trends of deer exposed to 6 different management strategies (Dec 50=50% population decline, Dec 0= Stable population size, AR0=Traditional point rules, AR 3=3-point rule, AR 4=4-point rule) in low (a) and high (b) deer densities, northwest Minnesota.

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

Kurt Haroldson, John Giudice, and Wendy Krueger

SUMMARY OF FINDINGS

The purpose of this study is to determine how much winter habitat is needed to sustain local populations of ring-necked pheasants (*Phasianus colchicus*) over a range of winter conditions. We estimated relative abundance of pheasant populations on 36 study areas using roadside surveys. In addition, we estimated amounts of winter cover, winter food, and reproductive cover on each study area by cover mapping to a geographic information system (GIS). During 2003-2004, pheasant indices varied in association with weather and habitat. A preliminary evaluation indicated that mean pheasant indices were positively related to habitat abundance in most, but not all, regions. Future work will include continued pheasant surveys for at least 3 additional years, improved estimates of habitat abundance, and more complex analysis of the association between pheasant indices and habitat parameters. A final product of this project will be a GIS habitat model that managers can use to target habitat development efforts where they may yield the greatest increase in pheasant numbers.

INTRODUCTION

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

Almost 1 million acres of cropland in Minnesota's pheasant range are currently retired under the Conservation Reserve Program (CRP). Wetland restorations, woody habitats, and food plots are eligible cover practices in the CRP, but most are inadequate in size, design, or location to meet pheasant habitat needs. Furthermore, small woody covers commonly established on CRP lands may reduce the quality of adjacent grass reproductive habitat without providing intended winter cover benefits. Pheasants use grasslands for nesting and brood rearing, and we previously documented a strong relationship between grassland abundance and pheasant numbers (Haroldson et al. 1998). However, information is lacking on how much winter habitat is needed to sustain pheasant populations during mild, moderate, and severe winters. The purpose of this study is to quantify the relationship between amount of winter habitat and pheasant abundance over a range of winter conditions.

OBJECTIVES

- Estimate pheasant abundance on study areas with different amounts of reproductive cover, winter cover, and winter food over a time period capturing a range of winter severities ($\geq$5 years);
- Describe annual changes in availability of winter cover as a function of winter severity; and.
- Quantify the association between mean pheasant abundance (over all

years) and amount of reproductive cover, winter cover, and winter food.

METHODS

We selected 36 study areas of contrasting land cover in Minnesota's core pheasant range to ensure a wide range of habitat configurations. Study areas averaged 9 miles^2 (5,760 acres) in size, and varied in the amount of winter cover, winter food, and reproductive cover. We defined winter cover as cattail (*Typha spp.)* wetlands ≥10 acres in area (excluding open water), dense shrub swamps ≥10 acres in area, or planted woody shelterbelts ≥3 acres in area, ≥200 feet wide, and providing dense cover at ground level (Gates and Hale 1974, Berner 2001). Winter food was defined as grain food plots left unharvested throughout the winter and located ≤1/4 mile from winter cover (Gates and Hale 1974). Reproductive cover included all undisturbed grass cover ≥20 feet wide. To facilitate pheasant surveys, 9 study areas were selected in each of 4 regions located near Marshall, Windom, Glenwood, and Faribault (Figure 1).

We estimated the amount of winter cover, winter food, and reproductive cover on each study area by cover mapping to a GIS from 2003 digital aerial photography. We used Farm Service Agency's GIS coverages of farm fields (Common Land Units) as base maps, and edited field boundaries to meet the habitat criteria of this project. Cover types were verified by ground-truthing all habitat patches visible from roads. Because cover mapping of cattail wetlands, shrub swamps, and undisturbed grasslands is still in progress, we made preliminary estimates of the amounts of these habitats from GIS coverages of the National Wetlands Inventory, Wildlife Management Areas, Waterfowl Production Areas, and CRP enrollments. We recognize that not all cattail wetlands, shrub swamps, and undisturbed grasslands are included in these GIS coverages. Furthermore, habitat omissions appear to be much more common on the Glenwood and Faribault study areas than on Marshall and Windom study areas.

We estimated relative abundance of pheasant populations on each study area using roadside surveys (Haroldson et al. 1998). Roadside surveys consisted of 10–12 mile routes primarily on gravel roads (≤ 4 miles of hard-surface road). Observers drove each route starting at sunrise at about 15 miles/hour and recorded the number, sex, and age of pheasants observed. Surveys were repeated 10 times on each study area during spring (April 20 – May 20) and summer (July 20 – August 20). Surveys were conducted on mornings meeting standardized weather criteria (cloud cover <60%, winds ≤10 miles/hour, temperature ≥32°F, dew present) 1–2 hours before sunrise; however, surveys were completed even if conditions deteriorated after the initial weather check. We attempted to survey all study areas within a region on the same days, and observers were systematically rotated among study areas to reduce the effect of observer bias on roadside counts.

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

For each study area and season, we calculated an index of relative pheasant abundance (pheasants counted/100 miles surveyed) from the sum of the 10 counts/sum of total miles driven. To evaluate the effect of habitat on pheasant abundance, we calculated a cover index for each study area:

CI = [(UG/Max)x4 + (WCwFP/Max)x4 + (WCwoFP/Max)x2 + (FP/Max)] / 11 where UG = undisturbed grass (% of study area)

WCwFP = winter cover near a food plot (number of patches)

WCwoFP = winter cover without a nearby food plot (number of patches)

FP = food plot (number of patches)

Max = maximum observed value among all 36 study areas

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

RESULTS

Spring 2004 Surveys

Observers completed all 360 surveys (10 repetitions on 36 study areas) during the spring 2004 season. Weather conditions during the surveys ranged from excellent (calm, clear sky, heavy dew) to poor (wind >10 mph, overcast sky, no dew, or rain). Over all regions, 78% of the surveys were started with at least light dew present, which was slightly less than last year (84%). Fifty-six percent of surveys were started under clear skies (<30% cloud cover), and 43% reported wind speeds <4 miles/hour. Only 4% of surveys were started on mornings with wind >10 miles/hour. Among regions, Glenwood experienced the least dew (43% of surveys started with no dew), and Windom experienced the most wind (61% of surveys started with wind speed ≥4 miles/hour).

Pheasants were observed on all 36 study areas during spring 2004, but abundance indices varied widely among areas from 11.6–359.7 pheasants observed per 100 miles (Table 1). Over all study areas, the mean pheasant index was 123.8 birds/100 miles, an increase from spring 2003 of 24% (95% CI: 9–39%). Total pheasants/100 miles varied among regions from 86.0 in the Faribault region to 179.7 in the Windom region (Table 2). Compared to 2003, total counts increased significantly only in the Marshall region (Table 2).

Hens were relatively abundant among study areas in spring 2004. The overall hen index averaged 69.4/100 miles, a 45% increase (95% CI: 13–77%) from 2003 (Table 2). Among regions, the hen index ranged from 38.8/100 miles in Faribault to 103.9/100 miles near Windom. Hen indices increased 55% (95% CI: 26–84%) from 2003 in the Marshall region, but were not significantly higher elsewhere. The observed hen:rooster ratio varied from 0.35 to 2.19 among study areas (Table 1). Fewer hens than roosters were observed on 1 study area in the Marshall and Glenwood region, 2 areas in Windom, and 6 areas in Faribault.

Summer 2004 Surveys

Observers completed 357 of the 360 surveys during the summer 2004 season. Weather conditions during the summer surveys ranged from excellent (calm, clear sky, heavy dew) to poor (light or no dew, overcast sky, or rain). Over all regions, 87% of the surveys were started with medium-heavy dew present, which was slightly better than last year (81%). Sixty-eight percent were started under clear skies (<30% cloud cover), and 76% reported wind <4 miles/hour. In comparison, 97% of the statewide August Roadside Surveys were started under medium-heavy dew conditions, 85% under clear skies, and 76% with winds <4 miles/hour. The less desirable weather conditions reported in this study probably reflects the study procedure of deciding whether to survey based on weather conditions 1–2 hours before sunrise at a location distant from the survey route.

Adult pheasants and broods were observed on all 36 study areas during 2004, but abundance indices varied widely from 4.1–335.0 pheasants observed per 100 miles (Table 3). Over all study areas, the mean pheasant index was 101.8 birds/100 miles, a 36% (95% CI: 21–51%) decrease from 2003. Total pheasant counts/100 miles varied among regions from 54.4 in the Faribault region to 180.1 in Windom (Table 4). Compared to 2003, total counts decreased significantly only in the Glenwood and Faribault regions (Table 4).

The overall hen index (hens/100 miles) decreased 29% (95% CI: 17–41%) from last year, and varied among regions from 12.3 in the Glenwood region to 36.3 near Windom (Table 4). Hen indices decreased 49% (95% CI: 33–65%) in the Glenwood region and 34% (95% CI: 18–50%) in the Faribault region, but were not significantly lower than 2003 in the Marshall and Windom regions (Table 4). In contrast, overall and regional cock indices were similar to last year (Table 4). The observed hen:rooster ratio varied from 0.3 to 2.9 among study areas (Table 3), and averaged 1.5 overall. Fewer hens than roosters were observed on 1 study area in Marshall, 2 in Glenwood, and 6 in the Faribault region.

The 2004 overall brood index (broods/100 miles) decreased 41% (95% CI: 29–53%) from 2003, with regional indices ranging from 6.8 in Faribault to 24.2 in Windom (Table 4). Regional brood indices decreased significantly only in the Faribault (95% CI: 48–78%) and Glenwood (95% CI: 52–68%) regions (Table 4). Mean brood size averaged 4.7 chicks/brood overall, and was relatively consistent among all regions (4.8 in Marshall, 5.0 in Windom, and 5.0 in Faribault) except Glenwood (4.1). Mean brood size in 2004 was similar to that in 2003, except in Glenwood, which experienced a decline of 17% (95% CI: 6–28%). On average, 23.1 broods were observed for every 100 hens counted during spring surveys, a 47% (95% CI: 33–61%) decline from last year. This brood recruitment index (broods/100 spring hens) varied among regions from 14.7 in Glenwood to 29.8 in Marshall. Brood recruitment indices declined significantly in all regions except Windom (Table 4).

Habitat Associations

The mean pheasant index (total pheasants/100 miles averaged over summer 2003–2004) was positively related to the cover index in all regions except Glenwood (Figure 2). Cover index explained 72% of the variation in pheasant indices in the Marshall region, 32% in Windom, 13% in Faribault, and 0% in Glenwood.

DISCUSSION

A high spring hen population in 2004, indicated by the 45% increase in the hen index from 2003, was expected given the mild winter of 2003-04. However, unusually cool weather during the reproductive period apparently prevented the abundant spring hens from recruiting large numbers of young into the summer population. The proportion of spring hens in 2004 that successfully recruited a brood into the summer population was only about one-half that of 2003. Furthermore, average brood size in the Glenwood region declined significantly. Thus, the summer 2004 pheasant index was 36% below the 2003 index. A large decrease in the summer hen index while the summer cock index remained stable suggested that some hens were still nesting or with young broods (which are typically undercounted in roadside surveys) during our survey period. Thus, the true population decrease may have been less than indicated by our population indices.

At this early stage in our evaluation, we cannot explain the weak association between summer pheasant indices and habitat abundance on the Glenwood and Faribault study areas (Figure 2). However, habitat estimates will be improved as we finish cover mapping the study areas. In addition, future analyses of pheasant-habitat associations will use multiple regression models that treat reproductive cover, winter cover, and winter food as independent predictor variables.

For the next reporting period, we will continue to survey pheasant populations during spring and summer. In addition, we hope to finish cover mapping all 36 study areas. During the next moderate-severe winter, we will assess winter habitat availability in relation to snow depth and drifting. Finally, we will begin to assess the potential for immigration to and emigration from the study areas by mapping large habitat blocks within a 2-mile buffer around the study area boundaries.

ACKNOWLEDGMENTS

We thank the survey teams for their efforts in completing the roadside surveys during 2004. T. L. Rogers, J. M. Snyder, T. J. Koppelman, and S. L. Goetz assisted with cover mapping of study areas. R. O. Kimmel and P. J. Wingate reviewed an earlier draft of this report.

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Table 1. Pheasant indices (birds/100 miles surveyed) and sex ratios (female:male) after 10 repeated surveys (n) on 36 study areas in Minnesota, spring 2004.

Region	Study Area	n	Total	Cocks	Hens	F:M Ratio
Marshall	1	10	133.3	55.9	77.4	1.4
	2	10	130.0	73.3	56.7	0.8
	3	10	135.0	49.5	85.4	1.7
	4	10	291.0	91.0	200.0	2.2
	5	10	102.5	45.4	57.1	1.3
	6	10	78.3	38.7	39.6	1.0
	7	10	81.7	32.1	49.5	1.5
	8	10	40.6	18.8	21.8	1.2
	9	10	54.4	21.9	32.5	1.5
Glenwood	10	10	57.1	24.5	32.7	1.3
	11	10	57.6	21.2	36.4	1.7
	12	10	101.9	58.6	43.3	0.7
	13	10	87.0	38.7	48.3	1.2
	14	10	107.9	46.5	61.4	1.3
	15	10	236.1	94.9	141.2	1.5
	16	10	187.6	70.5	117.1	1.7
	17	10	11.6	5.8	5.8	1.0
	18	10	170.4	63.9	106.5	1.7
Windom	19	10	312.6	105.3	207.4	2.0
	20	10	359.7	145.4	214.3	1.5
	21	10	205.3	81.1	124.2	1.5
	22	10	140.3	66.9	73.3	1.1
	23	10	223.8	106.9	116.8	1.1
	24	10	116.0	61.5	54.5	0.9
	25	10	134.0	51.4	82.5	1.6
	26	10	68.4	40.8	27.6	0.7
	27	10	57.4	22.6	34.8	1.5
Faribault	28	10	139.6	85.8	53.8	0.6
	29	10	55.8	41.1	14.7	0.4
	30	10	50.8	29.0	21.8	0.5
	31	10	112.0	60.9	51.1	0.8
	32	10	163.0	73.4	89.6	1.2
	33	10	110.3	49.1	61.2	1.2
	34	10	62.3	38.2	24.1	0.6
	35	10	41.7	27.5	14.2	0.5
	36	10	38.3	19.2	19.2	1.0

Table 2. Regional trends (% change) in pheasant population indices (birds counted/100 miles surveyed) on 36 study areas in Minnesota, spring 2003–2004.

Region	Group	Study areas	2003	2004	% change	95% CI
Marshall	Total pheasants	9	87.2	166.3	31	±16
	Cocks	9	43.1	47.4	11	±14
	Hens	9	44.1	68.9	55	±29
Glenwood	Total pheasants	9	100.9	113.0	12	±27
	Cocks	9	48.7	47.2	3	±25
	Hens	9	52.2	65.9	21	±31
Windom	Total pheasants	9	162.3	179.7	21	±39
	Cocks	9	69.4	75.8	13	±26
	Hens	9	92.9	103.9	31	±53
Faribault	Total pheasants	9	70.3	86.0	32	±32
	Cocks	9	37.1	47.1	30	±16
	Hens	9	33.2	38.8	72	±11
All	Total pheasants	36	105.2	123.8	24	±15
	Cocks	36	49.6	54.4	14	±10
	Hens	36	55.6	69.4	45	±32

Table 3. Pheasant indices (birds/100 miles surveyed) and sex ratios (female:male) after 10 repeated surveys (n) on 36 study areas in Minnesota, summer 2004.

Region	Study Area	n	Total	Cocks	Hens	F:M Ratio	Chicks	Broods	Chicks/ Brood	Broods/100 Summer Hens	Broods/100 Spring Hens
Marshall	1	10	198.2	27.9	34.2	1.2	136.0	24.3	5.6	71.1	31.4
	2	10	126.7	19.6	24.6	1.3	82.5	17.5	4.7	71.2	30.9
	3	9	91.7	4.3	11.9	2.8	75.5	12.9	5.8	109.1	15.2
	4	9	215.6	20.0	41.1	2.1	154.4	31.1	5.0	75.7	15.6
	5	10	95.8	15.0	23.3	1.6	57.5	13.3	4.3	57.1	23.4
	6	10	84.0	9.9	12.7	1.3	61.3	15.1	4.1	118.5	38.1
	7	10	107.3	10.9	18.2	1.7	78.2	20.0	3.9	110.0	40.4
	8	9	94.6	7.7	16.5	2.1	70.4	14.3	4.9	86.7	65.7
	9	10	20.2	6.1	1.8	0.3	12.3	2.6	4.7	150.0	8.1
Glenwood	10	10	65.7	3.5	9.6	2.7	52.5	11.1	4.7	115.8	34.0
	11	10	31.4	4.2	5.9	1.4	21.2	5.1	4.2	85.7	14.0
	12	10	36.2	7.1	11.0	1.5	18.1	4.8	3.8	43.5	11.0
	13	10	35.7	10.9	8.3	0.8	16.5	3.5	4.8	42.1	7.2
	14	10	74.6	5.7	16.2	2.8	52.6	14.0	3.8	86.5	22.9
	15	10	144.4	24.5	29.2	1.2	90.7	16.7	5.4	57.1	11.8
	16	10	67.6	10.5	17.1	1.6	40.0	9.5	4.2	55.6	8.1
	17	10	4.1	1.7	0.8	0.5	1.7	0.8	2.0	100.0	14.3
	18	10	61.1	6.9	12.5	1.8	41.7	9.3	4.5	74.1	8.7
Windom	19	10	206.3	21.1	61.1	2.9	124.2	29.5	4.2	48.3	14.2
	20	10	335.0	25.7	61.7	2.4	247.7	52.4	4.7	85.0	24.5
	21	10	120.0	22.1	29.5	1.3	68.4	12.6	5.4	42.9	10.2
	22	10	143.5	26.2	28.0	1.1	89.4	19.9	4.5	71.0	27.1
	23	10	222.8	37.6	38.6	1.0	146.5	29.7	4.9	76.9	25.4
	24	10	110.0	21.0	21.0	1.0	68.0	14.0	4.9	66.7	25.7
	25	10	255.7	29.2	38.7	1.3	187.7	29.2	6.4	75.6	35.4
	26	10	116.7	13.6	25.9	1.9	77.2	15.8	4.9	61.0	57.1
	27	10	111.3	16.1	22.2	1.4	73.0	14.8	4.9	66.7	42.5
Faribault	28	10	85.8	18.4	27.8	1.5	39.6	11.3	3.5	40.7	21.1
	29	10	33.0	7.8	5.8	0.8	19.4	3.9	5.0	66.7	26.5
	30	10	45.2	13.7	10.5	0.8	21.0	4.8	4.3	46.2	22.2
	31	10	62.7	23.5	15.7	0.7	23.5	7.8	3.0	50.0	15.4
	32	10	66.9	20.8	14.0	0.7	32.2	7.6	4.2	54.5	8.5
	33	10	89.6	7.8	22.6	2.9	59.1	13.9	4.3	61.5	22.7
	34	10	70.2	11.8	14.5	1.2	43.9	8.8	5.0	60.6	36.4
	35	10	21.3	7.1	3.5	0.5	10.6	0.9	12.0	25.0	6.3
	36	10	15.0	5.8	3.3	0.6	5.8	1.7	3.5	50.0	8.7

Table 4. Regional trends (% change) in pheasant population indices (birds counted/100 miles surveyed) on 36 study areas in Minnesota, summer 2003–2004.

Region	Group	n	2003	2004	% change	95% CI
Marshall	Total pheasants	9	142.6	114.9	–13	±29
	Cocks		12.7	13.5	23	±37
	Hens		25.6	20.5	–15	±28
	Broods		22.3	16.8	–21	±23
	Chicks/brood		4.6	4.8	8	±18
	Broods/100 spring hens		59.9	29.8	–49	±13
Glenwood	Total pheasants	9	139.9	57.9	–59	±10
	Cocks		9.2	8.3	–6	±40
	Hens		23.5	12.3	–49	±16
	Broods		20.2	8.3	–60	±8
	Chicks/brood		5.0	4.1	17	±11
	Broods/100 spring hens		44.7	14.7	–64	±8
Windom	Total pheasants	9	283.5	180.1	–18	±42
	Cocks		25.9	23.6	–3	±27
	Hens		50.9	36.3	–17	±27
	Broods		36.2	24.2	–21	±32
	Chicks/brood		5.4	5.0	–5	±16
	Broods/100 spring hens		47.1	29.1	–13	±43
Faribault	Total pheasants	9	164.6	54.4	–55	±16
	Cocks		9.5	13.0	56	±70
	Hens		23.6	13.1	–34	±16
	Broods		23.6	6.8	–63	±15
	Chicks per brood		5.5	5.0	–6	±38
	Broods per 100 hens		85.4	18.6	–63	±23
All	Total pheasants	36	182.6	101.8	–36	±15
	Cocks		14.3	14.6	17	±24
	Hens		30.9	20.5	–29	±12
	Broods		25.6	14.0	–41	±12
	Chicks/brood		5.1	4.7	–5	±12
	Broods/100 spring hens		59.3	23.1	–47	±14

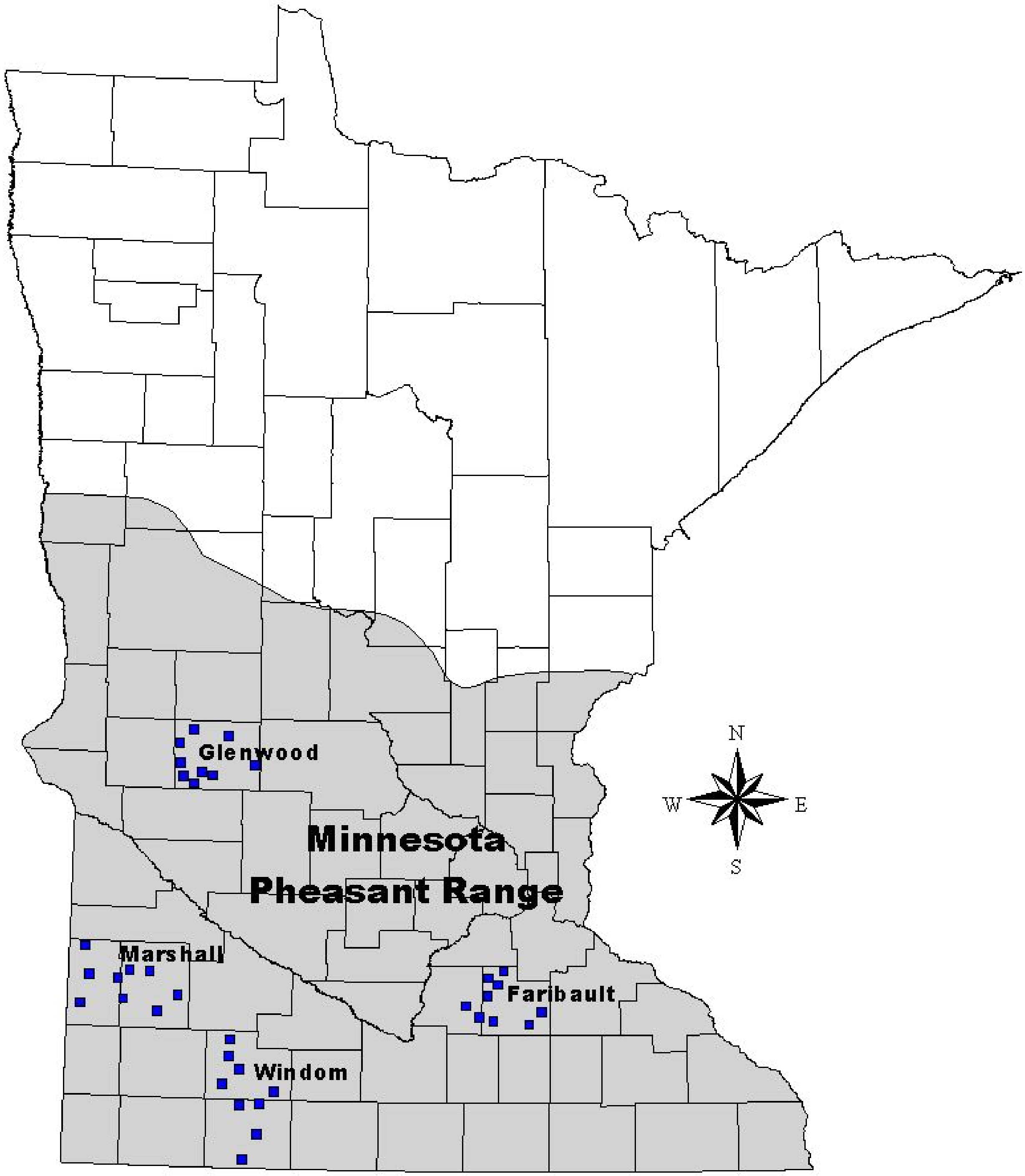

Figure 1. Locations of winter-habitat study areas within Minnesota's pheasant range, 2003-2004.

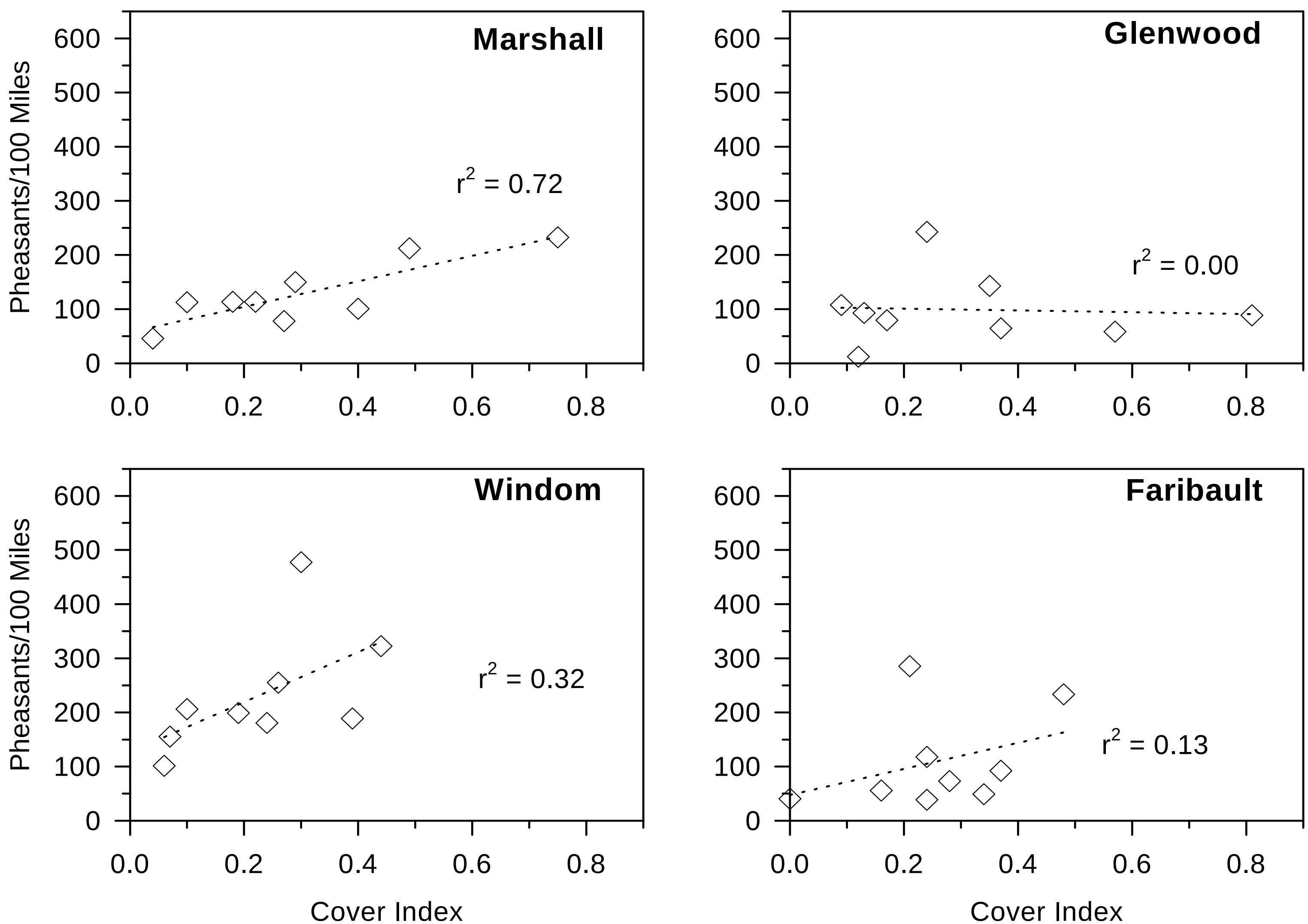

Figure 2. Relationship between relative pheasant abundance (pheasants counted/100 miles of survey) and amount of habitat (cover index) on 9 study areas in 4 regions in Minnesota during summer 2003-04.

MINNESOTA SPRING TURKEY HUNT LANDOWNER AND HUNTER SURVEY PILOT

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SUMMARY OF FINDINGS

Increased spring wild turkey (*Meleagris gallopavo)* hunter densities have resulted in concerns regarding hunt quality, hunter safety, and landowner tolerance of turkey hunters. The purpose of this study was to create 2 survey instruments to assess if hunter density affects hunter satisfaction and landowner attitudes. We sought to develop methodology that could be used to conduct an expanded study during the 2005 and 2006 spring turkey hunting seasons. Surveys were tested on 1 permit area (PA) in southeastern Minnesota (PA 343) during the spring 2004 turkey hunting season. The 3 most important issues the study evaluated were hunter access and safety, interference, and hunt quality. Hunter concerns about safety were low. Overall landowner attitudes were positive and most hunters found it very easy to gain access to private land. Interference by hunters or other individuals was infrequent. Based on hunter satisfaction and landowner attitudes the study found that a quality hunt was maintained at a hunter density of 1.6 hunters/mi2 of huntable habitat (forested area with a 50 m buffer; <2.8 hunters/mi2 of forested habitat).

INTRODUCTION

It is important to carefully allocate permit numbers to ensure hunter safety, limit hunter access problems, ensure landowner and hunter satisfaction, maintain hunt quality, and best manage the wild turkey population. Kimmel (2001) noted that season management strategies in Minnesota initially restricted numbers of hunting permits to protect developing wild turkey populations. Currently, permit numbers are restricted to ensure hunt quality. However, Dingman (2003) found that current hunter interference levels did not significantly affect hunter satisfaction. Still, managers in southeastern Minnesota have expressed concern that increasing hunter densities would impact hunt quality, hunter safety, and especially landowner tolerance of turkey hunters (G. Nelson, Minnesota Department of Natural Resources, personal communication).

For the spring 2004 turkey hunting season in Minnesota, PA 343 had the highest hunter density at <1.6 hunters/mi2 of huntable habitat (forested area with a 50 m buffer; <2.8 hunters/mi2 of forested habitat). Conrad et al. (1995) found that increasing hunter densities in southeastern Wisconsin to 3.0 hunters/mi2 of forested habitat had little impact on either hunters or landowners. Subsequently, Wisconsin Department of Natural Resources has increased permit levels that result in hunter densities of >6 hunters/mi2 of forested habitat in some areas (K. Warnke, Wisconsin Department of Natural Resources, personal communication). Hunter interest groups, in particular the Minnesota Chapter of the National Wild Turkey Federation, are aware of the higher hunter densities in Wisconsin and are requesting Minnesota Department of Natural Resources to increase spring wild turkey hunting permit numbers.

OBJECTIVES

- Create and test survey instrument to evaluate the effect of hunter density on hunter satisfaction
- Create and test survey instrument to evaluate the effect of hunter density on landowner attitudes about hunters
- Set landowner selection criteria

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- Determine appropriate sample sizes for surveying hunters and landowners

METHODS

Permit Area Selection

One permit area was used to pilot the methods and survey instruments. Wild turkey hunting PA 343 was selected for its high hunter density and ease in landowner selection. Permit area 343 had the highest spring hunter density in Minnesota during spring 2004. This PA only contained 1 county, Olmstead, which facilitated landowner selection.

Hunter and Landowner Selection

A sample of hunters was randomly selected from PA 343 using the Electronic License System (ELS) database of spring turkey hunting permit recipients. The ELS database contained permit recipients from all 8 spring turkey hunt time periods.

A sample of landowners was drawn from a database developed from county parcel data. Criteria for surveyed landowners included: ownership of at least 40 acres of land that intersects turkey habitat, parcels located outside of city limits, and exclusion of non-agricultural businesses and organizations. Each parcel was evaluated using these criteria with ArcView (Environmental Systems Research Institute, Redlands, CA). An Olmstead County parcel shapefile was obtained from the county tax role data, which included the taxpayer address, parcel size, and parcel location. The huntable turkey habitat shapefile (Kimmel 2001) was used to determine location of wild turkey habitat for PA 343. A city limit shapefile that identifies subdivisions and limits was also obtained. This shapefile had to be reprojected into the UTM zone 15 coordinate system (Manual 1) from Lambert Conformal Conic.

The Olmstead County parcel shapefile was queried to eliminate all parcels of land that were less than 40 acres in size or that fell within the city limits of Olmsted County. The shape file for PA 343 huntable habitat was used to identify parcels of land that had the potential to contain huntable turkey habitat (forest cover with a 50 m buffer). Parcels that intersected the huntable habitat shape file in ArcView were selected.

The resulting taxpayer database file was imported into Microsoft Excel. The file was brought into SAS and queried to combine the acres of parcels that were owned by the same landowner. Parcel acre data were summed by address to eliminate different names associated with identical addresses. Any landowner that had an out-of-state mailing address or was a government entity was eliminated from the database.

Survey Methodology

A hunter survey instrument was created to evaluate hunter satisfaction at varying hunter densities. The hunter survey instrument consisted of questions regarding hunter success, access, satisfaction, number of days hunted, time period, and interference from other hunters (Appendix A). For the spring 2004 wild turkey hunter survey, 450 surveys were mailed to a sample of turkey hunt permit holders in PA 343. The selected hunters received a survey and return envelope on the first day of the last time period of the spring turkey hunting season, (21 May 2004). A second mailing of surveys was sent to non-respondents three weeks after the initial survey mailing, (11 Jun 2004).

A landowner survey instrument was created to evaluate landowner attitudes about hunters at various hunter density levels. The landowner survey instrument contained questions regarding landowner attitudes about allowing access for spring turkey hunting (Appendix B). For the spring 2004 landowner survey, 500 surveys were mailed the last day of the turkey hunting season to landowners in PA 343 randomly picked from all landowners meeting selection criteria. Selected landowners were sent a survey and a return envelope on 21 May 2004. A second mailing was sent to non-respondents 2 weeks after the initial mailing, (10 Jun 2004).

RESULTS

We received an overall response rate of 79% for the hunter survey. The average number of turkeys seen by hunters was 17.8. The average number of turkeys shot at was 0.77. Hunters were more successful at bagging turkeys in the morning (82%) than in the afternoon. A total of 53% of hunters were successful at bagging a turkey.

The majority of hunters hunted on private land (89%) and of these hunters an average of 0.43 landowners turned down their request for access. Access to hunting was reported as very easy for the majority (52%; 178) of hunters (Fig. 1). Overall 99% of hunters responded no when asked if other hunters put them in danger at any time while hunting.

Overall, 96% (340) of hunters saw 0-2 hunters that were not part of their own hunting group (Fig. 2). The rate of interference from other hunters was 8% (28; Fig. 3), and from non-hunters was 11% (40; Fig. 4). Eighty-seven percent (284) of turkey hunters rated hunt quality above average (Fig. 5).

We received an overall response rate of 66% for the landowner survey. The top 2 reasons for landownership were farming and enjoying wildlife that lives on the property. Ninety-seven percent of landowners reported they did not lease out their land for spring turkey hunting. Overall, 87% of landowners reported seeing turkeys on their land in the past year.

Ninety-five percent of landowners did not personally hunt their land during spring 2004. Overall, <50% of landowners were asked for permission to hunt their land by each of the following groups: family (136), acquaintances (112), and strangers (84; Fig. 6). The majority (≥50%) of landowners did not allow any hunters on their land. Of landowners who allowed 1 or more individuals to hunt on their property, they were more likely to allow friends or family 42% (137) compared to acquaintances 29% (97) or strangers 15% (50) to hunt their land during the spring season (Fig. 7).

The majority 63% (207) of landowners reported that the number of hunters asking permission to hunt stayed the same over the past 5 years (Fig. 8). Landowners most often (54%; 179) neither agreed nor disagreed that there were too many hunters wanting to hunt their land (Fig. 9). Eighty-eight percent of landowners did not have problems with hunters trespassing on their land during the spring hunting season. Overall, 60% of landowners did not post signs on their land to control hunter access.

DISCUSSION

The survey instruments provide a way to evaluate issues important to hunter satisfaction and landowner attitudes. These issues include: hunter access and safety, interference, and hunt quality. In the future, tracking hunter and landowner responses to instrument questions in relation to varying hunter density levels will help maintain acceptable permit levels.

Hunter access was not indicated as a problem for PA 343 during the Minnesota spring 2004 turkey hunting season. Most hunters used private land for turkey hunting and the majority found access to be very easy. Hunter requests to use land for hunting from landowners were rarely denied. Hunters saw few individuals while hunting. Hunting interference rates were low, which likely led to greater hunter safety and satisfaction. Hunt quality ratings were high.

Landowner attitudes about spring wild turkey hunters were positive. Trespassing issues were low and posting land was not used to control hunting. Landowner perception of hunter density did not indicate they felt too many hunters were asking for hunting access.

This study indicated that hunters were not concerned with access issues, interference rates, and safety. Landowner attitudes about hunters were found to be at a level that allowed hunters access to land and did not indicate that landowners felt pressured by hunters requesting access. PA 343 had the highest hunter density in

Minnesota in spring 2004 and hunter satisfaction and landowner attitudes were at levels that indicated a quality hunt.

The methodology and results from this study will be used for an expanded study during the spring turkey hunting seasons of 2005 and 2006. We will compare hunter satisfaction and landowner attitude responses at varying and higher hunter density levels. This study will help to allocate permits at levels that will ensure a quality spring wild turkey hunt.

ACKNOWLEDGEMENTS

We would like to thank J. Guidice for statistical and study design consulting and T. Klinkner for developing a landowner mailing list database.

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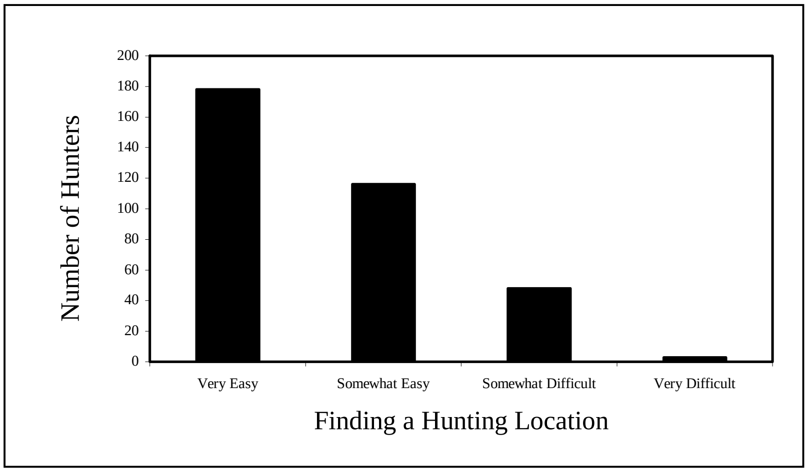

Figure 1. Difficulty ratings of finding a hunting location by Minnesota spring wild turkey, April-May 2004.

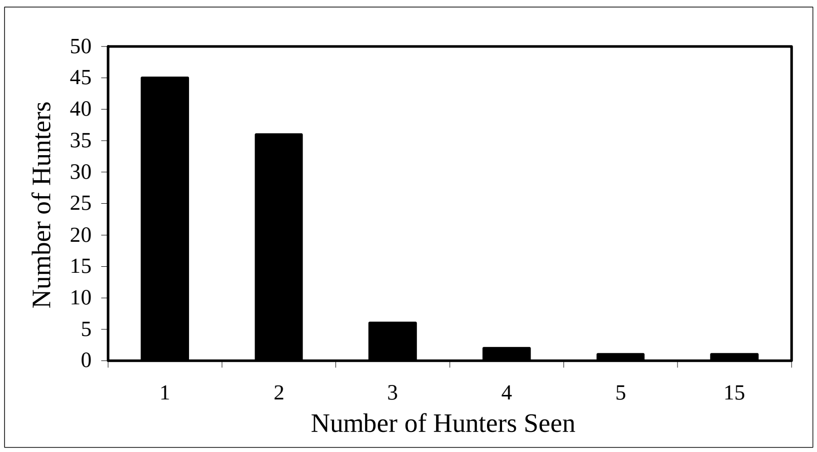

Figure 2. Number of hunters, not part of a hunter's own party, seen by hunters while hunting during the Minnesota spring wild turkey season, April-May 2004.

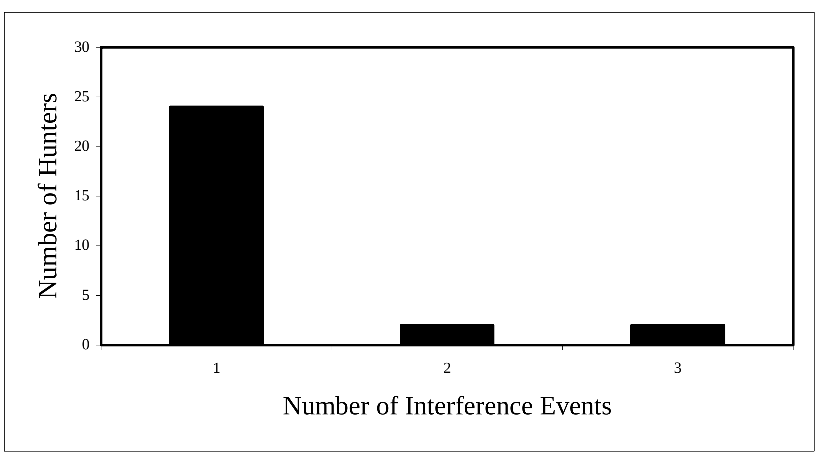

Figure 3. Number of times hunters were interfered with by other hunters while hunting during the Minnesota spring wild turkey season, April-May 2004.

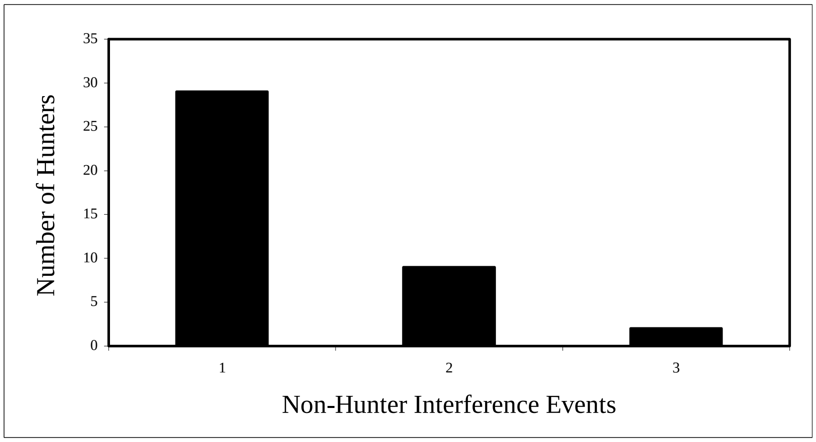

Figure 4. Number of hunters interfered with by non-hunters while hunting during the Minnesota spring wild turkey season, April-May 2004.

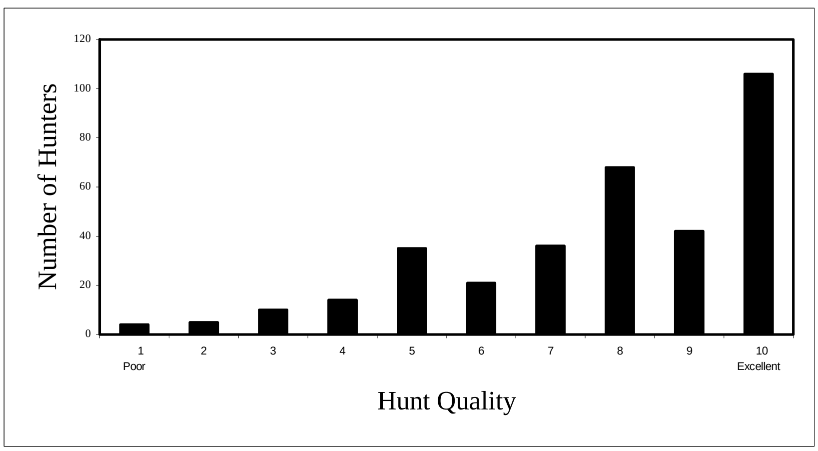

Figure 5. Hunt quality for the Minnesota spring wild turkey hunting season, April-May 2004.

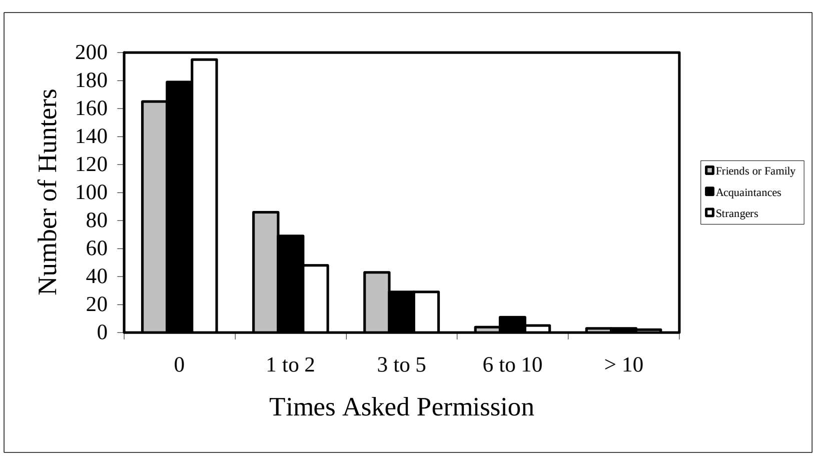

Figure 6. Number of times landowners were asked for permission to hunt their land by hunters for the Minnesota spring wild turkey season, April-May 2004.

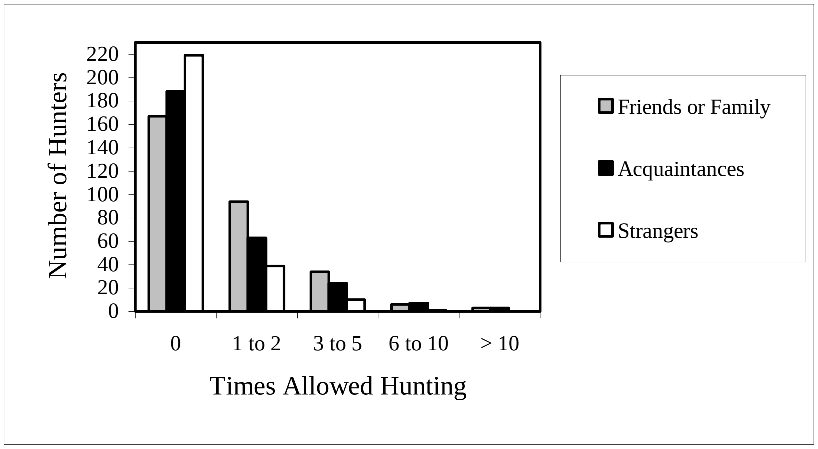

Figure 7. Number of times landowners granted hunting permission on their land during the Minnesota spring wild turkey season, April-May 2004.

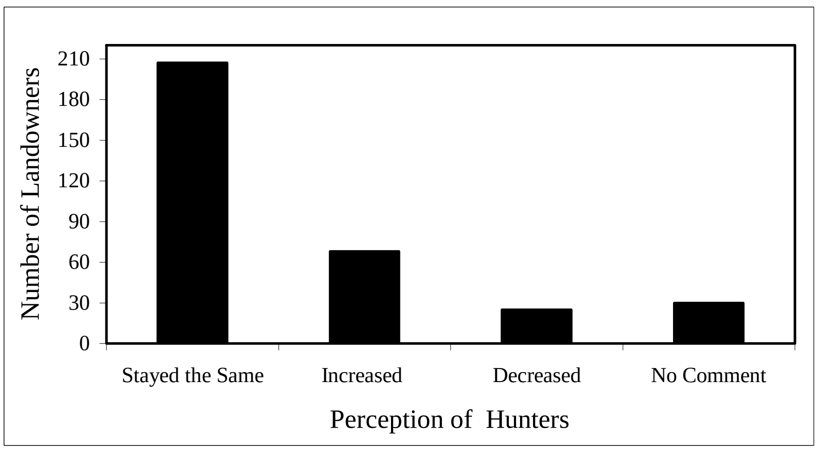

Figure 8. Landowner perception of the number of hunters requesting permission to hunt their land over the past 5 years, April-May 2004.

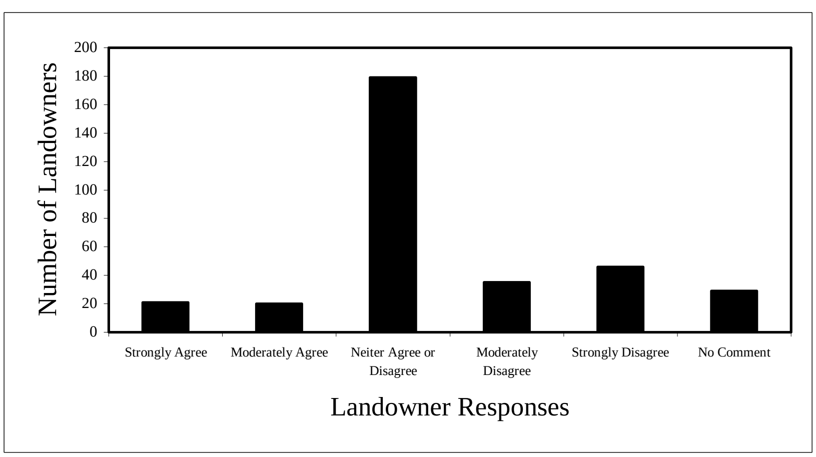

Figure 9. Landowner responses when asked if too many hunters wanted to hunt their land during the Minnesota spring wild turkey season, April-May 2004.

Appendix A. Hunter instrument for the 2004 Minnesota spring wild turkey hunting season survey.

Minnesota Spring Turkey Hunter Survey

*Please respond to all questions based on the **SPRING 2004 TURKEY SEASON**.

1. Did you hunt turkeys in Minnesota during the spring 2004 season? Yes____ No*____
   *If no, you do not need to continue but please return survey.

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

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

4. Which season did you hunt?

   April 14-18___ April 19- 23___ April 24-28___ April 29-May 3___ May 4-8___

   May 9-13___ May 14-20___ May 21-27____

5. How many days did you hunt turkeys during spring 2004? __________

6. How many turkeys did you see while turkey hunting in 2004? _______

7. How many turkeys did you shoot at? ______

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

9. How difficult was it for you to find a place to hunt during the spring 2004 wild turkey hunting season? (check one answer)
   Very easy____ Somewhat easy____ Somewhat difficult____ Very difficult____

10. Did you hunt on public land or private land during the spring 2004 season?
    Public_____ Private*_____ Both_____
    *If you hunted on private land, how many landowners turned down your request for permission? _____

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

12. On average, how many hunters, other than members of your own party, did you see each day while you were actually in the field hunting during spring 2004? ______

13. How many times did hunters, other than members of your own party, interfere with your hunting during spring 2004? ______

14. How many times did people **other than hunters** interfere with your hunting during spring 2004? __

15. Rate the quality of your turkey hunting experience during spring 2004 on a scale of 1-10 (check one number):

Poor Quality Average Quality Excellent Quality

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

Additional comments can be written on the back.

Appendix B. Landowner instrument for the 2004 Minnesota spring wild turkey hunting season survey.

Minnesota Spring Turkey Hunt Landowner Survey

*Please respond to all questions based on your land in **Olmsted County** for the **SPRING 2004** Turkey Hunting Season.

1) In what township is the majority of your land / farm located within Olmsted County?

_______________________________Township

2) How many total acres of land do you own in Olmsted County?

Acres Cropland___________ Acres Woodland __________ Other Acres___________

3) How long have you owned your land?

☐ 0-5 years ☐ 6-10 years ☐ > 10 years

4) Is your primary residence on this land?

☐ Yes ☐ No

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

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

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

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

7) Have you seen wild turkeys on your land in the past year?

☐ Yes ☐ No

8) Did you personally hunt wild turkeys on your land during spring 2004?

☐ Yes ☐ No

9) During the spring of 2004, how many turkey hunters **asked permission to hunt** on your land that were family or friends, acquaintances, or strangers? (Please check one box for each category.)

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

Acquaintances ☐ 0 ☐ 1-2 ☐ 3-5 ☐ 6-1 ☐ >10

Strangers ☐ 0 ☐ 1-2 ☐ 3-5 ☐ 6-10 ☐ >10

10) During the spring of 2004, how many turkey hunters did you **allow to hunt** on your land that were family or friends, acquaintances, or strangers? (Please check one box for each category.)

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

Acquaintances ☐ 0 ☐ 1-2 ☐ 3-5 ☐ 6-1 ☐ >10

Strangers ☐ 0 ☐ 1-2 ☐ 3-5 ☐ 6-10 ☐ >10

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

☐ Increased
☐ Decreased
☐ Stayed the same

12) How do you feel about the following statement:

There are too many spring turkey hunters that want to hunt on my land?

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

13) Did you have a problem with hunters trespassing on your property during the 2004 spring turkey hunt?

☐ Yes ☐ No

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

☐ Yes ☐ No

15) Provide any additional comments.

ANNUAL SURVIVAL AND PRODUCTIVITY OF WILD TURKEY HENS TRANSPLANTED NORTH OF THEIR ANCESTRAL RANGE IN CENTRAL MINNESOTA

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SUMMARY OF FINDINGS

Wildlife managers have succeeded in establishing wild turkey (*Meleagris gallopavo*) populations north of their suspected ancestral range. Supplemental food is being used to increase winter survival, but limited data exists regarding its influence on turkey condition. We tested 2 hypotheses: (1) supplemental food increases winter survival of transplanted wild turkey hens; and (2) although supplemental food increases winter survivorship, annual survivorship is similar due to increased predator abundance on supplemental food areas. During 2004, we conducted research on 6 113-km^2 study areas in rural, east-central Minnesota. Eastern wild turkey (*M. g. silvestris*) hens were captured in southeastern Minnesota and transplanted into study areas within 24 hrs of capture from January-March 2004. Hens were located via telemetry 3-5 times/week to determine fate (live/dead). Winter survival of transplanted wild turkey hens was higher on supplemental food study areas than on control study areas. Hen survival on the study areas was lowest during nesting and brood rearing. Over one half of mortalities occurred during these periods. Difference in survival rates of hens between supplemental food and control study areas was no longer apparent by December 2004.

INTRODUCTION

Wildlife managers have succeeded in establishing wild turkey populations north of the ancestral range reported by Leopold (1931). This expansion has lead to increased opportunity in hunting and wildlife viewing. How far north this range can be extended remains unanswered, and little information is available on the survival and productivity of transplanted turkeys to guide management. Porter et al. (1983) suggested severe winters can lower both over-winter survival, and the reproductive success of the surviving turkeys.

Supplemental food is being used to increase winter survival, but limited data exists regarding its influence on turkey condition. Porter et al. (1980) found corn is an important food resource that can increase survival and condition of wild turkeys during severe winter conditions (long periods of deep snow) in southeastern Minnesota. Kane (2003) also found higher winter survival of transplanted wild turkey hens in study areas with supplemental food plots in east-central Minnesota. Establishing supplemental food plots is expensive, and more information on winter survival is needed to justify these costs.

Although Kane (2003) found over-winter survival differed between treatment areas, annual survival of turkeys between supplemental food sites and control sites was similar. Hens on the supplemental food study areas had higher mortality during the nesting and brood rearing periods than hens on the control study areas. Kane (2003) did not evaluate nesting ecology of transplanted turkeys, and was unable to explain the cause for this difference, but higher predator abundance on the supplemental food study areas could explain this pattern. For example, Vander Haegen et al. (1988) and Palmer et al. (1993) found mortality of wild turkey hens is highest during the nesting period with predation being the dominant cause.

We tested 2 hypotheses: (1) supplemental food increases winter survival of transplanted wild turkey hens; and (2) although supplemental food

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increases winter survivorship, annual survivorship is similar due to increased predator abundance on supplemental food areas.

OBJECTIVES

- Monitor overwinter and annual survival;
- Determine productivity; and
- Determinepredator abundance.

STUDY AREA

We conducted our research on 6 113-km^2 study areas in rural, east-central Minnesota within the Mille Lacs Upland at the southern edge of the Laurentian Mixed Forest Province (Figure 1). The Mille Lacs Upland is characterized by level to rolling topography, and is a transitional zone between the Anoka Sand Plains (oak barrens, brushlands, and prairies) and the Tamarack Lowlands (lowland conifers, upland aspen-birch, upland conifers, and sedge meadows) (Marschner 1975, Hanson 2000). Evidence of past and present disturbances such as agricultural and forest clearings exists. With the exception of some state forest and state or county wildlife lands, the majority of land ownership is private. Historically, this region has a 30 day mean snow cover of $\geq$30.5 cm (Minnesota Department of Natural Resources 2004).

We added 2 study areas to Kane's (2003) study design. Morrison study area has supplemental food, and was located at approximately the same latitude as the 2 control study areas (Bradbury and Snake River; Figure 1). Sherburne study area served as a control, and was located at approximately the same latitude as the 2 supplemental food study areas (Foreston and Bock; Figure 1). The 3 supplemental food study areas (Foreston, Bock, and Morrison) were located in agricultural areas consisting primarily of corn, soybeans and hay, with dairy farms scattered throughout the landscape. Supplemental food plots consisted of standing corn left over-winter and some turkey feeders. Residual corn was available in some fields, as well as manure (potential food source) from livestock and dairy farms. Supplemental food study areas had a higher density of roads and buildings than control study areas, but large woodland and wetland patches also existed.

The 3 control study areas were located in areas where limited agriculture existed. However, some supplemental food was available in the form of bird feeders and the occasional resident leaving corn for wildlife. We assumed supplemental food, such as birdseed, was relatively constant among the 6 study areas. Large forest and wetland patches, and few roads and buildings characterized control study areas.

METHODS

Minnesota Department of Natural Resource (MNDNR) biologists used rocket nets to capture eastern wild turkey hens in southeastern Minnesota from January-March 2004. All hens were weighed to the nearest 0.23 kg, aged (juveniles or adults), leg-banded, fitted with a radio-transmitter, and transplanted to the study areas within 24 hrs of capture. Transmitters (95 - 104 g, 40 cm whip antenna) had a 3-year battery life and a mortality sensitive switch (Advanced Telemetry Systems-ATS, Isanti, MN, USA). Transmitters were positioned backpack style (Nenno and Healy 1979). We used model R2000 receivers (ATS) with either a handheld 3-element yagi antenna, or an omni-directional whip antenna attached to the roof of a pickup truck to monitor turkey movements and survival.

Winter Season

We designated the winter season as 1 January 2004 through 31 March 2004. Hens were located via telemetry 3-5 times/week to determine fate (live/dead). We investigated mortality sites the day of discovery, whenever possible, after a mortality signal was received. We determined causes of death by investigating the mortality site, and looking for species-specific predator sign such as tracks and hair or feathers. Mortalities were classified as mammalian or avian predation, emaciation, human (road kill), or unknown (See Miller et al. 1998).

White-tailed deer (*Odocoileus virginianus*) and other wildlife consumed all corn on all supplemental food plots prior to turkey releases. Therefore, we strategically placed "turkey feeders" on the supplemental food study areas to ensure food availability for the hens. We observed hens on the supplemental food study areas, and examined the crop contents of dead turkeys to determine the importance of supplemental food to over-wintering hens.

Summer Season

We designated the summer season as 1 April 2004 through 31 August 2004. We determined nest success of transplanted hens by radio-tracking twice/week from 1 April through 31 July. We considered hens that remain stationary for 7 days to be nesting (Vander Haegen et al. 1988), and we marked nests by flagging vegetation 30-50m around nest sites (Roberts et al. 1995, Badyaev and Faust 1996, Badyaev et al. 1996). We located and examined nests after hens and/or broods left the area to determine clutch size (number of unhatched eggs and egg caps), initial brood size (number of hatched eggs), and hatch success (proportion of hatched eggs/clutch; Vander Haegen et al. 1988). If a nest was depredated or an incubating hen was killed, we investigated the nest site and attempted to identify the predator.

We determined the relative abundance of mammalian predators by treatment type during the nesting season, because the majority of mortality on the supplemental food study areas occurred during summer (Kane 2003). We conducted scent-station surveys for mammalian predators. Surveys consisted of 10 linear stations placed $\geq$480 m apart along unpaved roads (Sargeant et al. 1998; M. Sovada, U.S. Geological Survey, personal communication). Scent discs were placed in the center of sand/soil areas approximately 1m in diameter. Stations were checked 48 hrs later and mammalian tracks were identified conservatively (a track was only added to the data if a positive identification was made). Identification of individuals of the same species could not be determined, and each species had a maximum of one track per station used in the analysis.

Data Summary

Because this is a preliminary report, survival analyses have not been conducted. Some of the hens survived the study conducted by Kane (2003) and were also used in the survival summary. Following Kane (2003), we censored newly released hens surviving <7 days post-release from the survival summary, because these deaths could have been associated with trapping-related stress or complications with the transmitter or harness (Vangilder 1996, Miller et al. 1998). Hens that disappeared from the study area, because of large movements or transmitter failure, were also censored from the survival summary during the period they disappeared.

We assumed that survival of each turkey was independent. Mortality dates were estimated using the midpoint between the last day we detected the bird alive and the first day we detected mortality.

RESULTS

The MNDNR trapped 62 hens for this study in 2004. Four hens were released on 9 January, 12 on 23 January, 10 on 27 January, 7 on 14 February, 7 on 21 February, and 22 on 13 March. We also monitored the movements and survival of 21 hens from Kane's (2003) study.

During 2004, we censored 8 of 62 (12.9%) hens from the overall survival summary, which reduced the total winter sample sizes to 36 hens on supplemental food study areas and 39 hens on control study areas. Three additional hens were lost from the study because of large movements or transmitter failure during the summer season, which reduced the supplemental food sample size to 33 hens for the summer season.

Winter survival was higher for hens on supplemental food study areas than on control study areas (Figure 2). In contrast, survival during summer was lower on supplemental food areas. Annual survival by treatment was similar.

Winter survival was 63% (20/32) for adults and 86% (6/7) for juveniles on control areas. Summer survival was 80% (16/20) for adults and 50% (3/6) for juveniles on control areas. Annual survival was 47% (15/32) for adults and 43% (3/7) for juveniles on control areas. Only adults were released on supplemental food areas during 2004.

Winter and annual survival was higher for previously released (residuals) than newly released hens (Figure 3). Summer survival was similar for newly released hens and residual hens.

The MNDNR trapped 57 hens for this study in 2005. Twenty-five hens were released on 12 January, 12 on 19 January, 9 on 20 January, and 11 on 21 January. We also monitored 27 hens remaining from previous releases.

During 2005, we censored 3 of 57 (5.3%) hens from the survival summary, which reduced the total winter sample sizes to 41 hens on supplemental food study areas and 39 hens on control study areas. Winter survival in 2005 was higher for hens on supplemental food study areas (95%; 39/41) than on control areas (74%; 29/39).

Causal Mortalities

Thirty-eight hens died during 2004 (30 newly released and 8 residual). We could not determine cause of death for 20 hens. Mammalian predation (coyote [*Canis latrans*], red fox [*Vulpes fulva*], and bobcat [*Lynx rufus*]) was the most common cause of mortality, followed by avian predation (great horned owl [*Bubo virginianus*] and bald eagle [*Haliaeetus leucocephalus*]), vehicle strikes, and starvation (Figure 4). Eight hens were censored from the survival summary and their causes of mortality included: avian predation (n = 3), trapping related stress (n = 2), unknown (n = 2), and mammalian predation (n = 1).

Scent survey results from June-September 2004 indicated higher predator abundance on control study areas. Predators visited 57% of the scent stations on control study areas. In contrast, mammalian predators visited only 12% of the scent stations on supplemental food study areas.

Crop contents of hens on supplemental food study areas included corn, soybeans and acorns during the winter months, and acorns, berries, grasses and invertebrates during the summer months. Crop contents of the hens on the control study area included acorns, berries and grasses throughout the year, and invertebrates when available.

Hens were observed nesting in tall grass, timber, and marshes in both supplemental food and control study areas. Brood sizes of both radioed and unmarked hens ranged from 1 – 13 poults/brood. Broods were observed on both supplemental food and control study areas.

DISCUSSION

Winter survival of wild turkey hens was higher on supplemental food study areas than on control study areas. Hens on supplemental food study areas were observed using the supplemental food during the majority of the winter. Winter survival was 100% in 2004 and 95% in 2005 for hens on supplemental food study areas. Kane (2003) found lower winter survival (81% in 2002 and 76% in 2003) for hens on supplemental food study areas. Kane (2003) also found winter survival of hens on control study areas to be 38.9% in 2002 and 45.5% in 2003. We found higher winter survival (67% in 2004 and 74% in 2005) for hens on control study areas.

Effects of supplemental food on winter survival should be interpreted with caution. The value of supplemental food may have been overestimated because 15 hens were released on 13 March 2004 into supplemental food study areas.

Survival may not be as high as indicated as these hens were only at risk for 2.5 weeks of the winter season on the study area. However, survival analyses taking time of survivorship into consideration may improve our understanding. The winter of 2003-2004 was relatively mild, only February had snow cover >20 cm (Minnesota Department of Natural Resources 2004). However, cold temperatures existed in January and February with 6 days $\leq$-29 C°. In a mild winter, Porter et al. (1980) found similar survival between hens with supplemental food and hens without supplemental food.

Consistent with other studies, hen survival on our study areas was lowest during nesting and brood rearing (Palmer et al. 1993, Wright et al. 1996, Kane 2003). Over one half of our mortalities occurred during these periods. Summer survival in 2004 was higher on control study areas than on supplemental food study areas. Predation, as in other studies, was the most common cause of mortality (Porter et al. 1980, Miller et al. 1998, Kane 2003). Preliminary predator survey results indicate a higher abundance of predators on control study areas, but data were very limited and more research is needed. Other factors may be responsible for decreased summer survival of hens on supplemental food study areas.

The difference in survival rates of hens between supplemental food and control study areas for winter and summer of 2004 was no longer apparent by December 2004. As noted above, summer survival was lower on supplemental food study areas. The reason for this remains unknown, but the increased number of individuals on supplemental food study areas could have caused a higher risk of mortality.

Our pooled 2004 winter survival of 83% is noticeably higher than winter survival rates in Minnesota found by Kane (60.4%; 2003) and Porter (59.7%; 1978). Pooled annual survival (47%) was similar to annual survival for established populations of turkeys in Wisconsin (53%; Wright et al. 1996), Mississippi (51%; Miller et al. 1998), and Missouri (44%; Kurzejeski 1987). Wild turkeys are resilient and can survive north of their ancestral range as long as reproduction and productivity compensate for losses.

Residual hens had higher winter and annual survival than newly released hens. Experience with local environments increases survival for residual versus transplanted hens (Miller 1990). Knowledge of local habitats provides an advantage for residual hens in finding food, roost sites, and potentially avoiding predators. However, newly released and residual hens had similar summer survival indicating transplanted hens became acclimated by summer.

Juvenile hens on control study areas had higher winter survival than adults. Kane (2003) also noticed this trend in mild winters. However, in a winter with deep snow, Porter et al. (1980) found adult survival to be higher than juveniles in study areas without corn. Adults had higher summer and annual survival than juveniles on control study areas.

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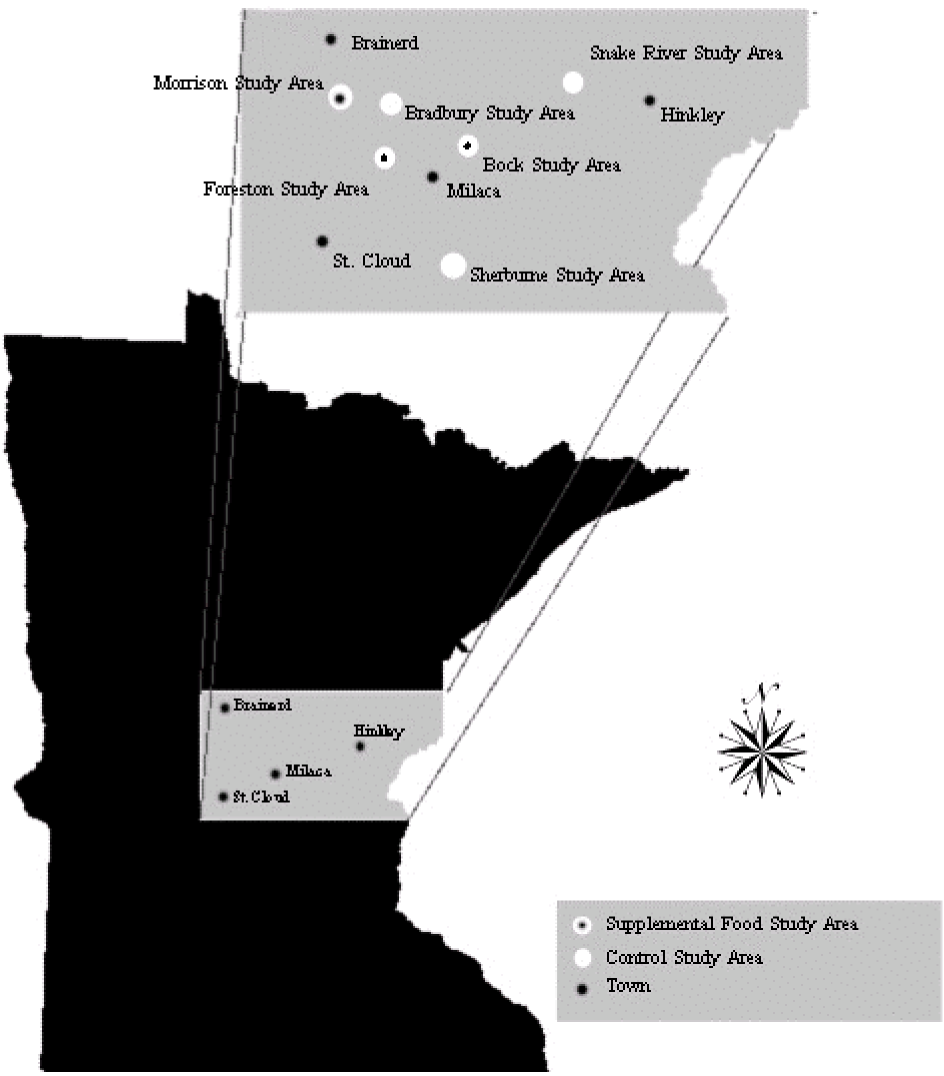

Figure 1. Locations of 6 wild turkey hen survival study areas (3 supplemental food and 3 control) north of presumed wild turkey ancestral range in east-central Minnesota, 2004.

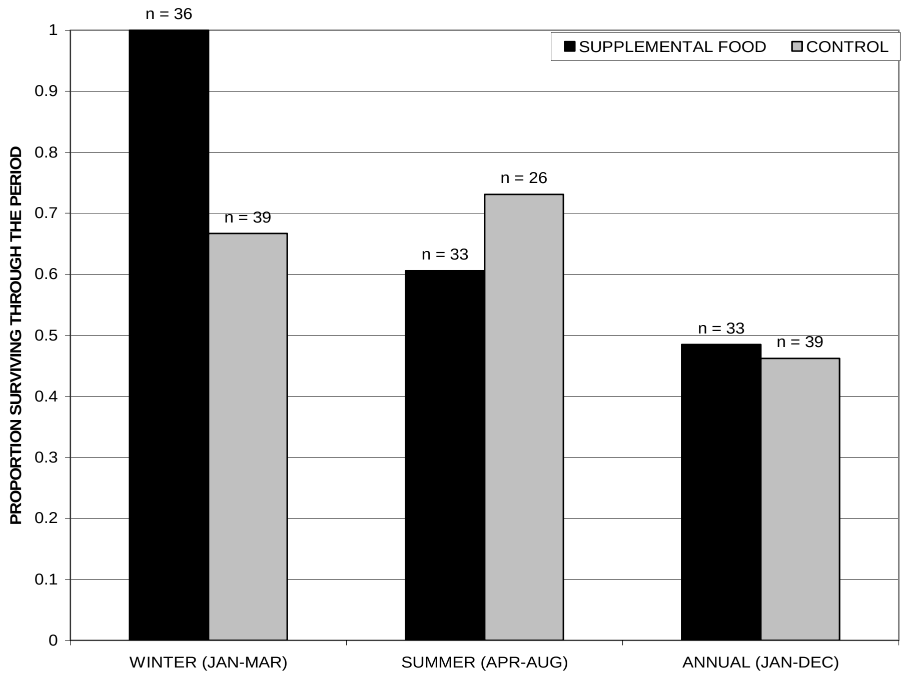

Figure 2. Seasonal and annual survival of eastern wild turkey hens in supplemental food and control study areas in east-central Minnesota, 2004.

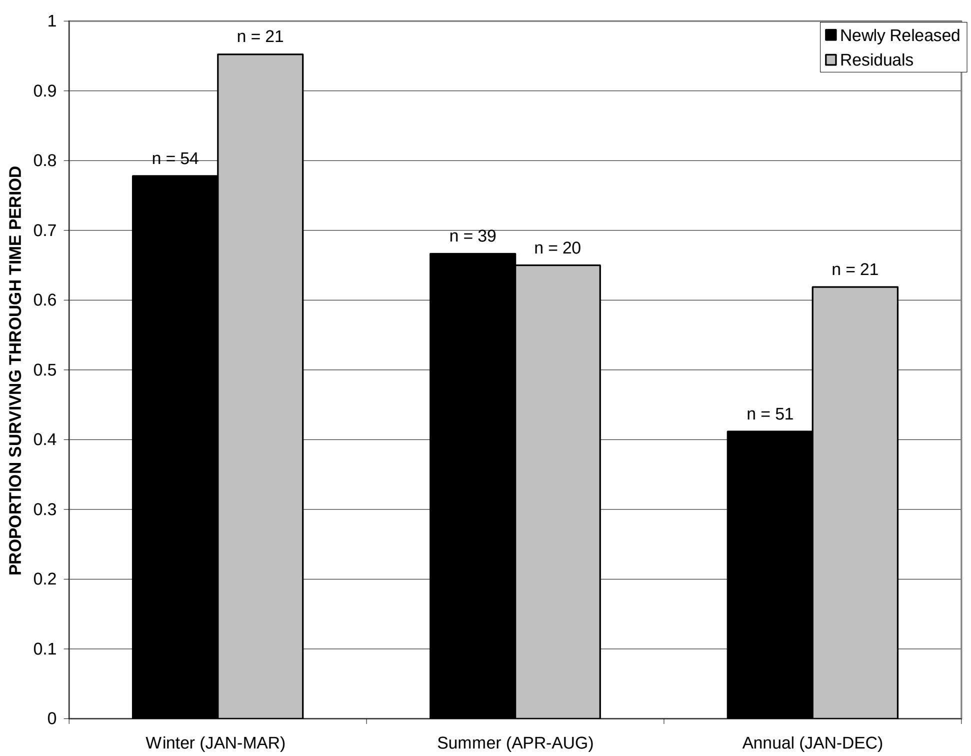

Figure 3. Seasonal and annual survival of newly released and residual eastern wild turkey hens in east-central Minnesota, 2004.

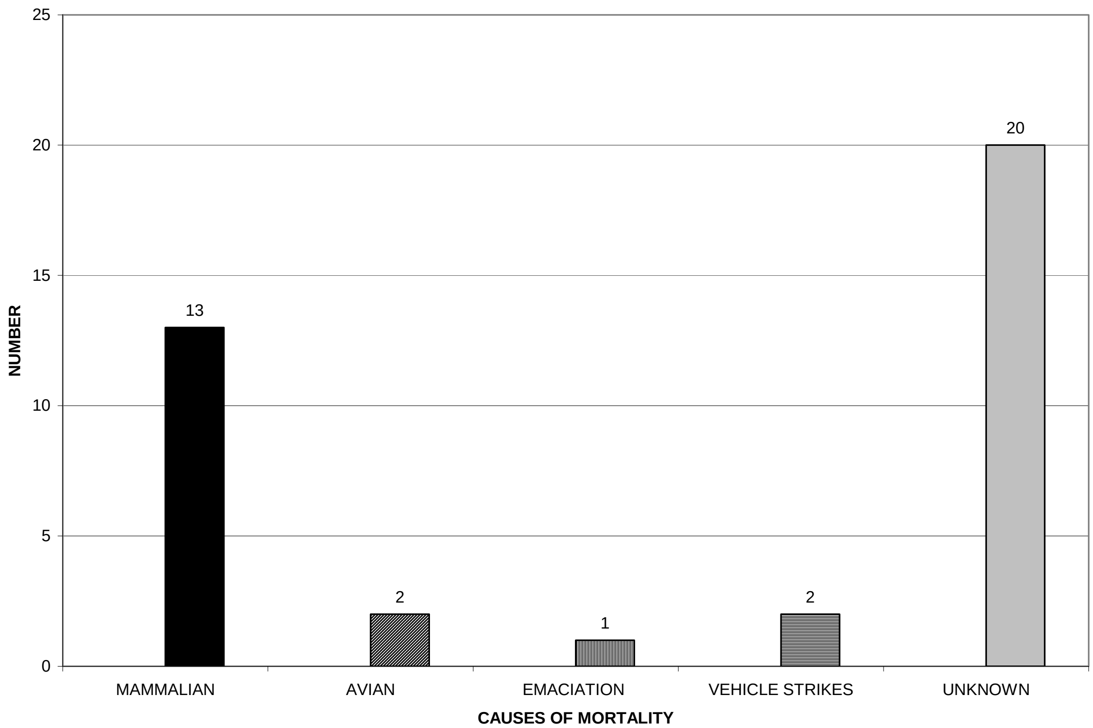

Figure 4. Causes of eastern wild turkey hen mortalities in east-central Minnesota, 2004. The emaciated hen weighed 5 kg at the time of the release and only 2.3 kg a month later.

WILD TURKEY DISTRIBUTION AND URBAN HUMAN/TURKEY INTERACTIONS ALONG THE RED RIVER VALLEY IN NORTHWESTERN MINNESOTA

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SUMMARY OF FINDINGS

This study was initiated in 2003 with an initial objective of using mail surveys to estimate the minimum wild turkey (*Meleagris gallopavo)* population in the Red River Valley in the Fargo, North Dakota/Moorhead, Minnesota (F/M) area. The Red River Valley offers suitable turkey habitat in a relatively narrow corridor surrounded in the F/M area by a dense human population. In 2004, we also monitored urban human/turkey interactions. In 2005, we added a survey to assess public opinion on wild turkey management options in the event abatement measures were necessary due to problems with urban turkeys.

A total of 537 and 368 turkeys were reported in 2003 and 2004, respectively. We consider this a minimum estimate of turkey populations in the area, because reports believed to be duplicates were eliminated. In 2004, 12.5 % of the survey respondents (n=40) reported negative human/turkey interactions. Landowners expressed concerns about turkeys at bird feeders, on decks, and in yards close to houses. To date, a higher number of complaints has been received for 2005, including reports of turkeys blocking a bridge, roads,, and driveways, as well as entering yards and eating from bird feeders and gardens. One report mentioned aggressive turkey behavior towards a young child.

Public opinion surveys of management options for abatement are currently being compiled. While initial opinions are mixed, approximately one-half of the survey respondents agreed or strongly agreed with a turkey hunting season option to reduce potential problems.

INTRODUCTION

The Red River Valley, along the northwestern border between Minnesota and North Dakota, is near the northern range where turkey transplants have occurred in Minnesota. Records indicate that several releases of wild turkeys were conducted in the 1980s and 1990s surrounding F/M. Wild turkeys released along the Sheyenne River near Lisbon, ND are assumed to have spread to the F/M area (L. Tripp, North Dakota Game and Fish Department [retired], personal communication). Minnesota Department of Natural Resources (MNDNR) released turkeys in southeastern Clay County, Minnesota , but this release is approximately 40 miles from F/M and separated from F/M by open farmland (G. Nelson, MNDNR, personal communication). In addition, residents indicate that pen-raised turkeys were likely released in the F/M area. Turkey populations along the Red River Valley in F/M and surrounding areas have been increasing and expanding in recent years, as indicated by turkey population surveys conducted by MNDNR in 1999 and 2002.

The F/M area along the Red River Valley is an ideal place to evaluate human/turkey interactions in an urban setting. Turkeys use the narrow wooded riparian corridors along the Red and the Sheyenne rivers. Both rivers intersect a number of cities and towns, including the highly populated F/M area. Human/turkey interactions are increasingly becoming a problem in other urban areas where turkeys have been established for a longer time. Wild turkeys released on the fringes of the Twin Cities in Minnesota

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have expanded into urban areas resulting in increased complaints about problems with turkeys at bird feeders, in yards, and as threats to children (Moriarty and Lueth 2003).

The initial objective of this study in 2003 was to obtain a minimum estimate of the turkey population in the F/M area. However, in 2004 our focus shifted to monitoring human/turkey interactions, determining if urban turkey problems are developing, and determining possible abatement measures.

In this report, we summarize our methods and results from 2003, 2004, and early 2005. In 2005, we are concentrating on monitoring human/turkey interactions, and assessing public opinion on possible problem turkey abatement measures.

OBJECTIVES

- Estimate minimum wild turkey populations along the Red River Valley in the F/M area;
- Monitor urban human/turkey interactions and conflicts; and
- Conduct a public-opinion survey regarding abatement measures to reduce human/turkey problems.

METHODS

Study Area

The study area includes the Red River Valley in the F/M area. The area extends north along this riparian corridor to Georgetown, Minnesota (approximately 20.9 km north of Highway 10 in Moorhead, Minnesota) and south to Wahpeton, North Dakota/Breckenridge, Minnesota (approximately 72.4 km south of Moorhead). The Fargo, West Fargo, and Moorhead area has a combined human population of approximately 140,000. The Wahpeton/Breckenridge area has approximately 13,000 people, while Georgetown has a considerably lower human population (approximately 125). Most of the study area is included in Cass County, North Dakota and Clay County, Minnesota, but extends south into Richland County, North Dakota and Wilkin County, Minnesota.

Survey Methods

In spring 2003, we hand-distributed a 1-page survey, requesting information on numbers and locations of turkey observations, to landowners along the Red River Valley. We also requested turkey observation information in local news media: *The Barnesville Recorder* and *The (Fargo-Moorhead) Forum.* Local residents who saw turkeys were encouraged to contact us by phone, email, or by completing a survey. In spring 2004, surveys were mailed to respondents from 2003 along with newspaper requests. During winter 2005, surveys were mailed to all prior respondents in addition to randomly selected landowners obtained from the Cass and Clay County tax roles.

Surveys in 2003 contained questions about numbers of turkeys observed, and respondents were asked to indicate the location of the observation on a map of the local area. For survey maps and data summarization in 2003, the study area was divided into 3 sections covering a 24.1 km radius north and south of F/M: Red River North (the river corridor north of F/M), Red River South (south of F/M), and Fargo/Moorhead (the area within the cities). In 2004, we added a fourth section: Sheyenne River (the area southwest of F/M covering the Sheyenne River Valley near Horace, ND). Surveys distributed in 2004 included questions about human/turkey interactions. Two different surveys were mailed in 2005, one requesting turkey observation information and a second survey with opinion questions about landowner attitudes regarding wild turkey management options for potential problem turkey abatement measures. Questions for this survey requested opinions about such options as modifying habitat (exclosures for bird feeders, gardens, etc.), using visual/audio stimuli to deter turkeys, relocating problem turkeys, removing bird feeders/turkey attractants from yards, and opening a turkey hunting season. Sightings with similar numbers of turkeys in the same locations were considered duplicates and were eliminated from the analysis.

RESULTS AND DISCUSSION

In spring 2003, we distributed 100 surveys with 64 returned surveys and 11 e-mail responses. In spring 2004, we mailed 150 surveys with 40 returned surveys and 12 e-mail responses. Based on survey responses, the minimum wild turkey population was estimated at 537 for 2003 and 368 for 2004 (Tables 1 and 2). In winter 2005, we mailed 500 surveys and, at this writing, we have received 42 responses. Preliminary population estimates for 2005 appear to be similar to 2003. Even though we made an attempt to eliminate duplicate sightings, population estimates may be inflated due to repeat sightings of the same turkey flocks. However, we also assume that we are not receiving reports of all the turkeys in the area. Thus, we consider the estimates to be reasonable as minimum populations estimates for the F/M area.

In 2004, the reported negative interactions between wild turkeys and humans were quite low (12.5%, n = 40; Table 3). Complaints included turkeys at bird feeders, on decks, and close to landowner homes.

The reported human/turkey interactions from the 2005 surveys, while not complete, indicate a potential increase in negative interactions with urban turkeys. At this writing, we have received 10 complaints from 42 returned surveys (23.8%). For 2005, we have received reports of turkeys as a "traffic hazard." Four reports from Georgetown, Minnesota (north of F/M) noted turkeys blocking traffic on a main bridge. Other respondents reported turkeys blocking a driveway or a road. One response from a resident near Harwood, ND, reported turkeys on a lawn displaying aggressive behavior towards a 2-year old child.

Results are currently being compiled for the 2005 public opinion survey regarding problem turkey abatement measures. Although data are incomplete, approximately one-half of the returned surveys agreed or strongly agreed that a wild turkey hunting season would be an acceptable option to reduce potential urban turkey problems. Hunting restrictions within cities may limit possibilities of using this option.

During 2005, we plan to gather more data on human/turkey interactions in urban areas. We plan to identify what type of interactions occurred, where interactions occurred, and investigate whether types and frequency of turkey problems are related to turkey population density. We would like to conduct aerial surveys to refine population estimates in our study area.

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Table 1. Minimum wild turkey population estimates from landowner surveys distributed in the Red River Valley in the Fargo, ND/Moorhead, MN area in spring of 2003. Data are based on 64 surveys returned out of 100 distributed surveys plus 11 e-mails.

Section	Turkeys Observed	Known Males	Known Females
Red River North	287	27	47
Red River South	105	5	10
Fargo/Moorhead	145	19	37
Total	537	59	94

Table 2. Minimum wild turkey population estimates from landowner surveys distributed in the Red River Valley in the Fargo, ND/Moorhead, MN area in spring of 2004. Data are based on 40 surveys returned out of 150 distributed surveys plus 12 e-mails.

Section	Turkeys Observed	Known Males	Known Females
Red River North	47	30	8
Red River South	50	0	0
Fargo/Moorhead	211	13	5
Sheyenne River	60	14	6
Total	368	57	19

Table 3. Negative human-turkey interactions recorded from a landowner survey distributed in the Red River Valley - Fargo, ND/Moorhead, MN area in spring of 2004. Data are based on 40 returns from 150 mailed surveys and responses from people who responded to newspaper articles and did not receive a survey in the mail.

Section	Recorded Negative Interaction
Red River North	2
Red River South	0
Fargo/Moorhead	0
Sheyenne River	3
Total	5

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ECOLOGY AND POPULATION DYNAMICS OF BLACK BEARS IN MINNESOTA

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

SUMMARY OF FINDINGS

During April 2004 – March 2005, 42 radiocollared black bears 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, 781 individual bears were handled of which 474 were radiocollared at these 3 sites, beginning in 1981 in the CNF. In recent years, GPS radiocollars provided detailed information on movements of bears, yielding insights into establishment of home ranges, seasonal forays, dispersal, and wanderings after translocation. Mortality data were obtained through collars turned in by hunters or collars tracked to carcasses. Hunting is by far the largest source of bear mortality. In the past few years, however, hunters were asked not to shoot collared bears, so our study no longer provides representative mortality data. The study is now focused primarily on reproduction. Reproductive output varies among the 3 study sites in response to food conditions, but no upward or downward trends through time are evident. Most bears in Minnesota begin producing cubs at 3–7 years old (at 3 only in the southern area, 6–7 only in the central and northern areas), and produce an average of 2.6 cubs per litter (modal litter size = 3) every 2 years (mean = 2.06 years). Cub mortality, which is nearly twice as high for males as for females, varies by area from 17–33%. These data have been used to model and track the statewide population.

INTRODUCTION

A paucity of knowledge about bear ecology and effects of harvest on bear populations spurred the initiation of a long-term telemetry based bear research project by the Minnesota Department of Natural Resources (MN DNR) in the early 1980s. For the first 10 years, the study was limited to the Chippewa National Forest (CNF), near the center of the Minnesota bear range. After becoming aware of significant geographic differences within the state in sizes, growth rates, and productivity of bears, apparently related to varying food supplies, we started other satellite bear projects in different study sites. Each of these began as graduate student projects, supported in part by the Minnesota 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 three 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 three statewide population estimates over a span of 12 years (Garshelis and Visser 1997). 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 and Noyce 2005). All of these represent areas of continued research and monitoring.

OBJECTIVES

- Monitor temporal and spatial variation in cub production and survival;
- Obtain additional, improved, measurements of body condition, heart function, and wound healing abilities; and
- Examine habitat use and detailed movements (dispersal, establishment of home ranges, fall excursions, etc.) with GPS telemetry.

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 a year at their den site. Bears in dens were immobilized with an intramuscular injection of Telazol, administered with a jab stick or Dan-Inject dart gun. Bears were then removed from the den for processing, which included changing or refitting the collar, or attaching a first collar on yearlings, measuring, weighing, and obtaining blood and hair samples. We also measured biolelectrical impedance (to calculate percent body fat) and vital rates of all immobilized bears. Additionally, with the cooperation of investigators from the University of Minnesota (Dr. Paul Iaizzo) and Medtronic (Dr. Tim Laske), heart condition was measured with a 12-lead EKG and ultrasound on a select sample of bears (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 eartagged. 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 2004, a total of 634 individual bears were handled in and around CNF, 76 at Camp Ripley, and 71 at VNP. Of these, we collared and monitored 386 bears in CNF, 49 at Camp Ripley, and 39 in VNP. As of April 2004, the start of the current year's work, we were monitoring 22 collared bears in the CNF, 5 at Camp Ripley, and 8 in VNP. By April 2005, after deaths, failed radiocollars, and the addition of some new bears obtained through trapping, released orphaned cubs, and den visits, 37 bears were collared on the 3 study sites.

Movements

We have been using collars containing both VHF radios and GPS units during the past few years to obtain more reliable data on movements and habitat use than obtainable with standard VHF collars. Twelve bears (some in all 3 study areas) were equipped with GPS collars, but 5 collars failed, 1 was dropped by the bear, and 3 of the bears were shot; thus, we obtained a full year of GPS data on only 3 bears, and a partial year of data on 4 other bears.

Four GPS-collared bears at Camp Ripley provided particularly interesting data. One 3-year-old female, the daughter of a bear who wore a GPS collar in 2002, used nearly the same area as her mother (Figure 1). Like her mother, she avoided the target impact area at the north end of camp where National Guard troops shoot live ammunition. It is virtually devoid of trees. The 3-year-old, however, did use the more southern impact area during the fall, as this area contains highly productive oak trees. The mother, who in 2002 had an older model GPS collar that collected less data was never located in the southern impact area, but about a month of fall data were missing from her record (only 370 locations were recorded for the mother vs 1072 for the daughter). The young female also spent some time outside the Camp, which was not evident in the mother's record.

Three GPS-collared males all spent time outside the Camp (Figure 2). An 8-year-old made a southward movement of 28 miles (45km) during the fall (8 August–17 September), and returned to the Camp to den. A 5-year-old that was trapped as a nuisance outside the Camp in early May was translocated 50 miles north. He immediately began moving westward, and in 10 days traveled 67 miles (109 km); he covered the last 46 miles (73 km) in 80 hours, moving mainly at night, ending at an unfenced beeyard where he was shot and killed. A 1-year-old, that was collared near the hunter's bait site where his mother was killed the previous year (and weighing 106 pounds as a cub), initiated a dispersal at the end of May. He moved 45 miles (72 km) northeastward, then retraced his route and settled 26 miles (42 km) from his natal range. From 21 August until the opening of bear season on 1 September he remained within an area of only ⅛ mi^2 (possibly smaller – within the error of the GPS unit). As he was shot the first day of the season, we suspect that his small area of use was centered on a hunter's bait.

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, 2 of 18 (11%) collared females from the CNF and 2 of 11 (18%) collared bears at Camp Ripley were shot by hunters (bear hunting is not allowed on Camp Ripley, but bears are vulnerable to hunters when they leave this area, as noted above). Four additional collars were lost track of during the hunting season, either as a result of premature battery failure or being destroyed and not reported by hunters.

In addition to these hunter-related mortalities, 4 bears were shot as nuisances; 2 of these were reported as required by statute, and 2 were found only because they were collared. One bear was killed in late April at a bird feeder, 1 was killed during opening fishing season for unknown reasons, 1 was killed at a beeyard that was not protected by electric fence, and 1 was killed because it purportedly attacked a pet dog on a porch. Nuisance-related deaths are the second-highest cause of bear mortality in the CNF (Table 1). Vehicle collisions are the second-leading source of mortality at Camp Ripley. Smaller patches of habitat and higher road densities, resulting in increased traffic-related deaths, probably limit the southward expansion of bears. Very few bears, other than cubs, die of natural mortality.

Reproduction

Two 5-year-old CNF females produced their first litters in 2005, and one 3 and one 4-year-old produced their first litters at Camp Ripley. At Camp Ripley, where hard mast (especially oak) is more abundant, bears have a somewhat earlier age of first reproduction than in CNF.

Litter size tends to be less responsive to food conditions than age of first reproduction. However, first litters by young females are often smaller and have higher cub mortality than subsequent litters (Noyce and Garshelis 1994). The younger age of first birthing by females at Camp Ripley thus explains their somewhat lower average litter size and higher cub mortality, compared to CNF (Tables 2 and 3). VNP, having lower natural food availability than either Camp Ripley or CNF, had the oldest age of first reproduction as well as smaller litters and higher 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 81 (2%) collared bears in this area produced cubs at 3. Age-specific cub production increased until about 7 years old (Figure 3), at which point nearly all bears had produced their first cubs. From age 7 to 25 years, 47.5% 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 104 observed intervals between successful litters, all but 6 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, varied by 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.

Cub production among radiocollared females in CNF did not show an upward or downward trend during our 25 years of monitoring, but exhibited a strong 2-year cycle since 1995 (Figure 4). Other black bear studies indicated that such cycles are instigated when productivity is synchronized by a poor food year that causes reproductive failures, especially among potentially primiparous females (McLaughlin et al. 1994, Miller 1994). However, the cycling in our study began just prior to a food failure during the summer and fall of 1995. Thus, the poor cub production in 1996 was a consequence of both good productivity the year before (as bears cannot produce surviving litters in 2 consecutive years) as well as poor food production in 1995. The continued cycling, though, is somewhat an artifact of our sample. Once a sample of reproductive bears becomes synchronized on a 2-year cycle, this synchrony continues through time, being diluted only by deaths of some of these bears, the inclusion of newly-collared bears, or newly maturing collared bears producing cubs in the off years. Since the late-1990s, we collared fewer new bears each year than in the early years of the study, so apparent reproductive cycles within our sample tend to persist. Nevertheless, a matching cycle of productivity is also evident in the age structure of harvested bears from a wide area in northern Minnesota (Garshelis and Noyce 2005).

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 (24% M vs 11% F in CNF; 33% M vs 17% F in Camp Ripley; 40% M vs 25% F in VNP). However, sex ratios at birth were skewed towards males in all areas (51–53%; Tables 1–3). These results have been used as inputs in a statewide population model.

ACKNOWLEDGMENTS

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

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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, 1981–2005. 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	207	8	8
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	1
Died from unknown causes	3	1	0
Total deaths	268	20	11

[a] Lost track of during the hunting season.

Table 2. Black bear cubs examined in dens of radiocollared mothers in or near the Chippewa National Forest during March, 1982– 2005.

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%	—
Overall	**168**	**437**	**2.6**	**51%**	**17%**

[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 3. Black bear cubs examined in dens of radiocollared mothers in Camp Ripley Military Reserve during March, 1992–2005.

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%	—
Overall	**18**	**42**	**2.3**	**52%**	**25%**

[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 4. Black bear cubs examined in dens of radiocollared mothers in Voyageurs National Park during March, 1999–2005.

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%	—
Overall	**15**	**30**	**2.0**	**57%**	**33%**

[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.

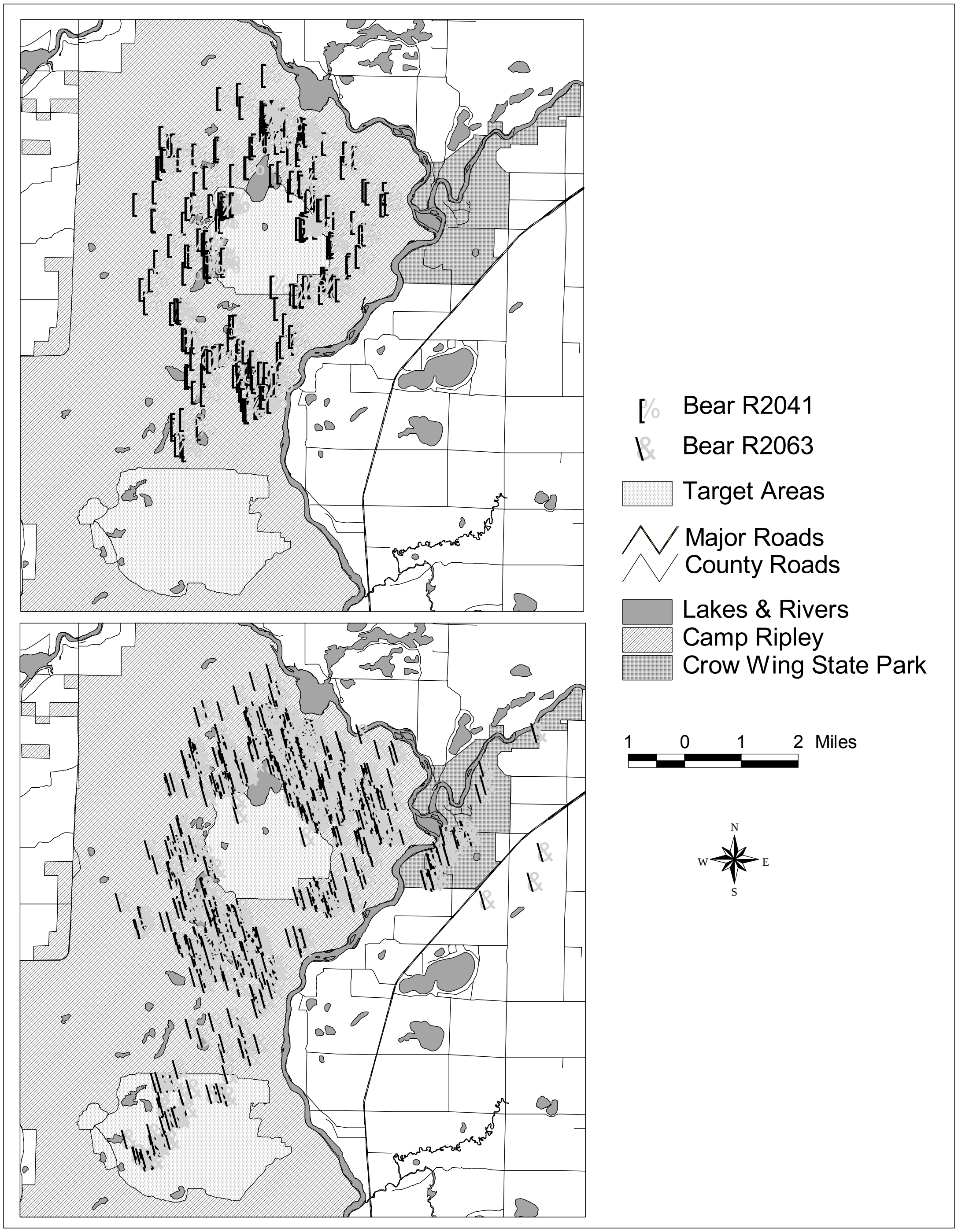

Figure 1. Movements of a GPS-collared female (top) in Camp Ripley during 2002 compared to her 3-year-old daughter (bottom) in 2004. Both bears avoided the northern impact target area (with few trees), but the daughter was attracted to productive oaks in the southern target area during the fall. More data were obtained in 2004 because of a newer model GPS collar.

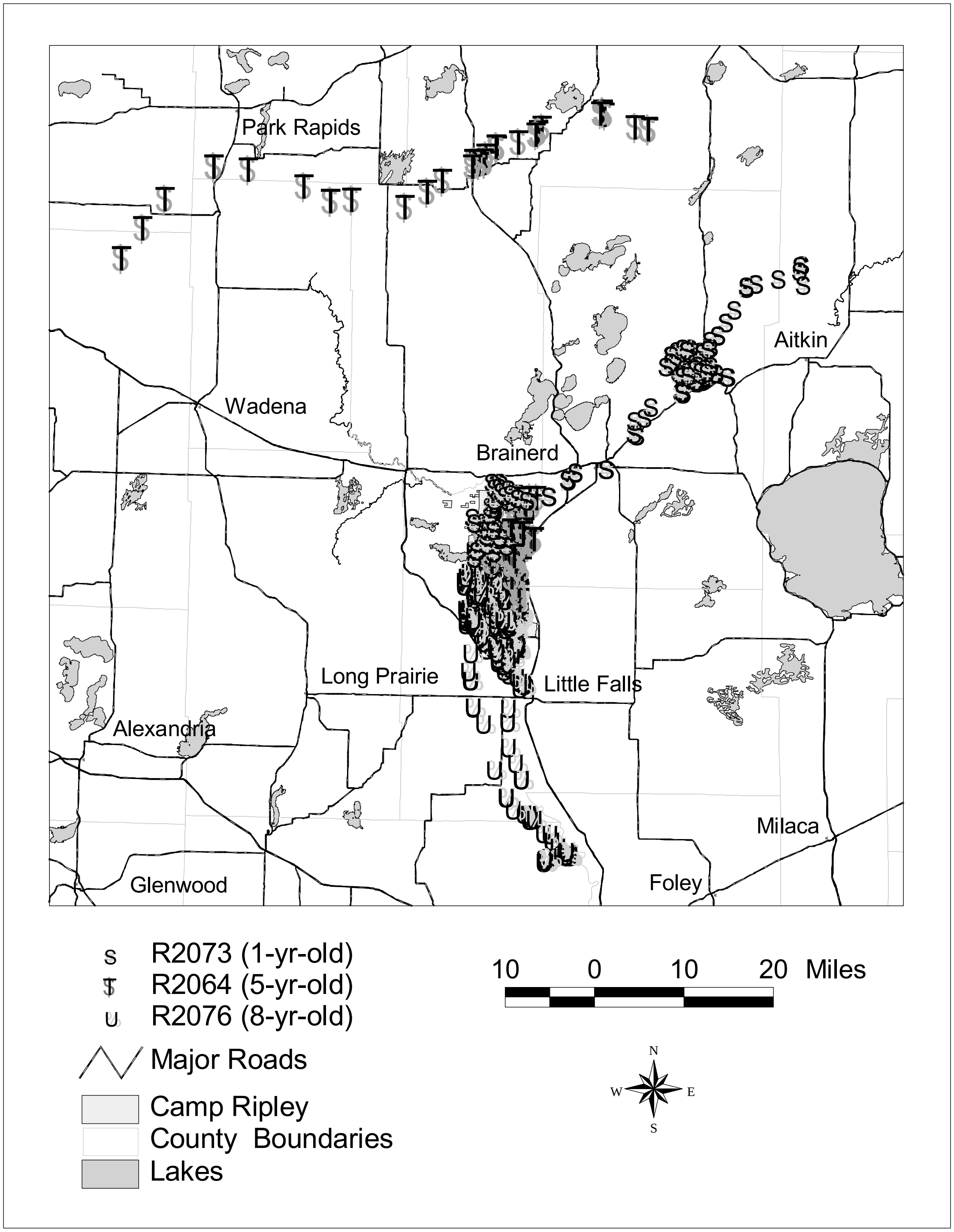

Figure 2. Movements of 3 GPS-collared males in Camp Ripley (southern Minnesota bear range) during 2004. The 8-year-old made a fall foray south of Camp, the 5-year-old was translocated north and then moved westward and was killed in a beeyard, and the 1-year-old dispersed northeastward and was shot by a hunter.

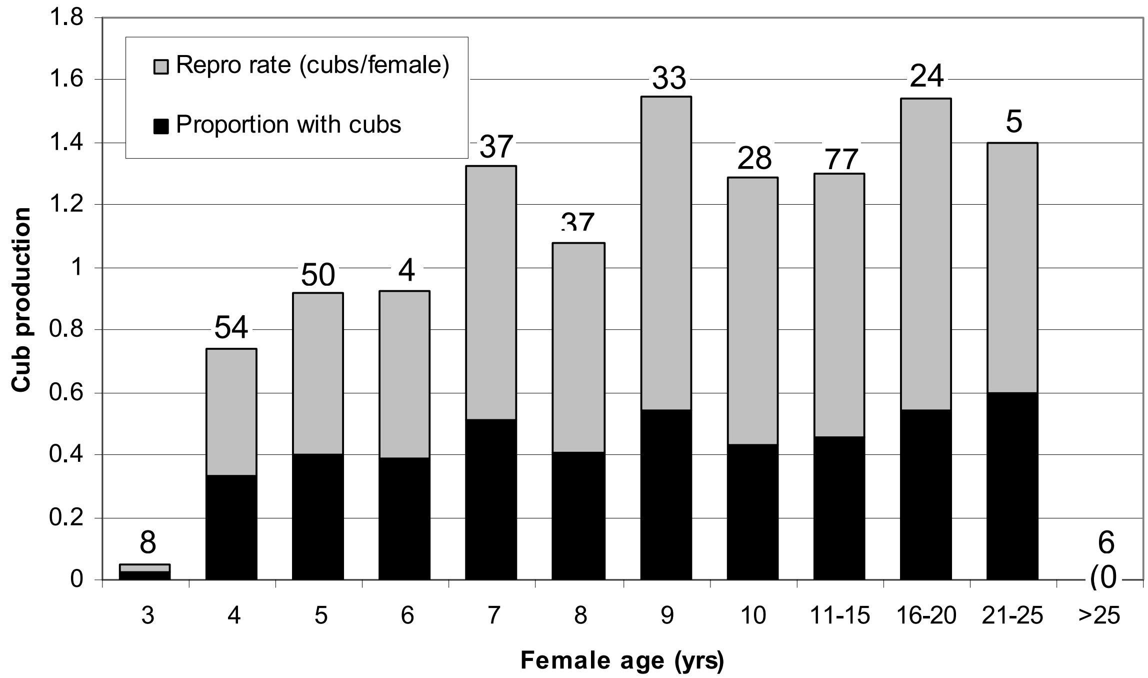

Figure 3. Age-specific cub production of bears in the Chippewa National Forest (central Minnesota) measured as the proportion of females with cubs during March den visits, 1982–2005, and cubs (M+F) per female. Sample sizes shown above bars represent bear-years (bears x years). However, only one individual bear was monitored past age 20 (for 11 years).

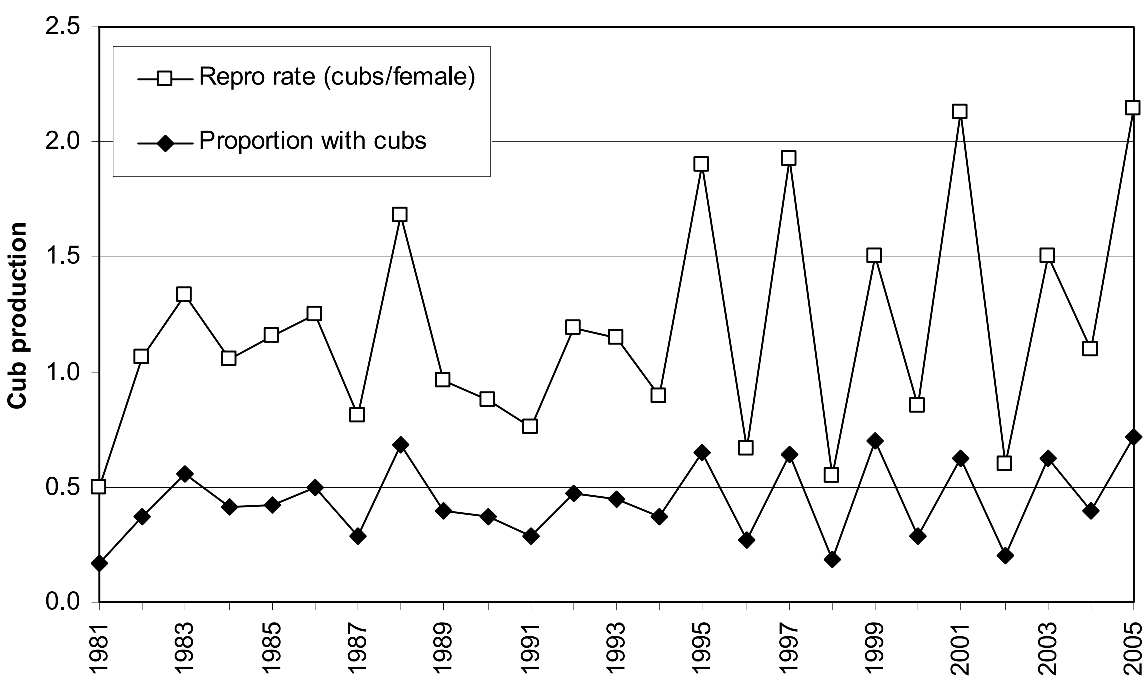

Figure 4. Year-specific cub production of bears in the Chippewa National Forest measured as the proportion of females with cubs during March den visits and cubs (M+F) per 4+ year-old female. Sample sizes vary from 5–25 females monitored per year (mean = 16).

GRIZZLY BEAR DEMOGRAPHICS IN AND AROUND BANFF NATIONAL PARK AND KANANASKIS COUNTRY, ALBERTA

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Abstract: The area in and around Banff National Park (BNP) in southwestern Alberta, Canada, is one of the most heavily used and developed areas where grizzly bears (*Ursus arctos*) still exist. During 1994–2002 we radiomarked and monitored 37 female and 34 male bears in this area to estimate rates of survival, reproduction, and population growth. Annual survival rates of bears other than dependent young averaged 95% for females and 81–85% for males. Although this area was largely unhunted, humans caused 75% of female mortality and 86% of male mortality. Females produced their first surviving litter at 6–12 years of age ($\bar{x}$ = 8.4 years). Litters averaged 1.84 cubs spaced at 4.4-year intervals. Adult (6+ year-old) females produced 0.24 female cubs per year and were expected to produce an average of 1.7 female cubs in their lifetime, based on rates of reproduction and survival. Cub survival was 79%, yearling survival was 91%, and survival through independence at 2.5–5.5 years of age was 72%, as no dependent young older than yearlings died. Although this is the slowest reproducing grizzly bear population yet studied, high rates of survival seem to have enabled positive population growth (λ=1.04, 95% CI = 0.99–1.09), based on analyses using Leslie matrices. Current management practices, instituted in the late 1980s, focus on alleviating human-caused bear mortality. If the 1970–80s style of management had continued, we estimated that an average of 1 more radiomarked female would have been killed each year, reducing female survival to the point that the population would have declined.

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

Glenn D. DelGiudice

SUMMARY OF FINDINGS

During 8 February–23 March 2005, we had 91 captures, which included 66 initial captures (26 adult and 11 fawn females, 10 adult and 19 fawn males), and 25 recaptures (9 adult and 9 fawn females, 7 fawn males). As of 31 March 2005, a total of 452 female deer, including 43 female newborns, have been recruited into the study. The fawn:doe capture ratio during winter 2004–05 was the highest of the 15-year study (111 fawns:100 does); it followed 3 consecutive mild winters. Previously, the highest fawn:doe capture ratio (105:100) occurred during moderately severe winter 2000–01, which also followed an historically unprecedented 3 consecutive mild winters. The fawn:doe ratio was as low as 32:100 (winter 1996–97), attributable primarily to historically severe winter 1995–96. After the first year of the study, mean age of females remained stable and ranged from 5.0 (± 0.4 [SE], n = 90) in 2001 to 7.1 (± 0.6, n = 62) years old in 1993. During 2004, mean age was 6.5 (± 0.4) years old, compared to 6.0 (± 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 (95.2%, n = 218) with pregnancy rates for does 1.5–15.5 years old of 87.5 to 100%. There was a difference ($P \leq 0.05$) in mean body mass at capture for pregnant (63.0 ± 0.7, range = 45.7–82.5 kg, n = 171) versus non-pregnant (54.6 kg ± 2.8, range = 43.3–69.1 kg, n = 10) does, which is indicative of an effect of inadequate nutrition on conception during the breeding season. The winter severity index (WSI) for our study sites has now ranged from 38 (winters 2003–04) to 185 (winter 1995–96) during the past 15 winters. The WSI of winter 2004–05 (108) was attributable in part to 57 snow-days, whereas, during 5 of the past 7 winters accumulated snow-days were only 0–6. Winter mortality of adult females (≥1.0 year old) has ranged from 2.0 to 29.3% during 1990–91 to 2004–05, and it is significantly related to WSI (r^2 = 0.47, P = 0.005). Annual mortality of females (including fawns) has ranged from 9.1 to 47.6% through 2004. Wolf predation (24.5%), hunter harvest (21.8%), and "censored" (38.2%, i.e., lost to monitoring or still alive) accounted for the fates of most of the collared females through 2004.

INTRODUCTION

The goal of this long-term investigation is to assess the value of conifer stands, as winter thermal cover/snow shelter, to 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 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, and distribution of 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: (1) monitor deer movements between seasonal ranges by aerial radio-telemetry, and more importantly, within winter ranges, for determination of home range size; (2) determine habitat composition of winter home ranges and deer use of specific vegetation types; (3) monitor winter food habits; (4) monitor winter nutritional restriction and condition via sequential examination of deer weights, body composition, blood and bladder urine profiles, and urine specimens suspended in snow (snow-urine); (5) monitor age-specific survival and cause-specific mortality of all study deer; and (6) 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

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 Lake, Dirty Nose Lake) and 2 treatment sites (Inguadona Lake, Shingle Mill Lake), 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 are located in the Grand Rapids-Remer-Longville area of north central Minnesota and are 10.4-22.0 km^2 (4.0-8.5 mi^2) in area. The study began with the Willow Lake and Inguadona Lake sites during winter 1990-91 with the Shingle Mill Lake and Dirty Nose Lake sites included beginning in winter 1992-93.

The objective of the experimental treatment (impact) was to reduce moderate ($\geq$40-69% canopy closure) and optimum ($\geq$70% canopy closure) conifer thermal cover/snow shelter to what is considered a poor cover class (< 40% canopy closure). We just completed our 15th winter 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 along the eastern and southern boundaries of the Chippewa National Forest, Minnesota (46°52'–47°08'N and 93°45'–94°08'W). We augmented our capture efforts during some winters (not in 2004–05) with rocket-netting (Hawkins et al. 1968) and net-gunning from helicopter (Wildlife Capture Services, Marysvale, Utah). Generally, handling of each deer included chemical immobilization (intramuscular injection of a 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 last incisor for age-determination (Gilbert 1966), various morphological measurements, and administration of a broad-spectrum antibiotic. All does were checked for pregnancy by dop-tone or visual ultrasound. Female fawns and does were fitted with VHF radiocollars (Telonics, Inc., Mesa, Arizona[1]) for monitoring their movements and survival 9 does also were fitted with global positioning system (GPS) radiocollars (Advanced Telemetry Systems, Inc ., Isanti, Minnesota[1]). Upon completion of handling, all deer immobilizations were

[1] disclaimer

reversed with an intravenous injection of yohimbine HCl. Additional details of deer capture and handling are provided elsewhere (DelGiudice et al. 2001, 2005*b*, Carstensen Powell 2004).

We live-captured wolves (*Canis lupus*) with Newhouse no. 14 steel leghold traps during May–September 1993–2004 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 and Handling of Study Deer

During this study, we have had 1,208 deer captures, including recaptures. Because the study focuses on females, male fawns (< 1.0 year old in their first winter) and adult (≥1.0 year old) males were eartagged and released. As of 31 March 2005, a total of 452 female deer, including 43 female newborns, have been recruited into the study. Additionally, 47 male newborns were 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 in Carstensen Powell (2004).

During 8 February–23 March 2005, we had 91 captures, which included 66 initial captures (26 adult and 11 fawn females, 10 adult and 19 fawn males), and 25 recaptures (9 adult and 9 fawn females, 7 fawn males). This winter's winter severity index (WSI = 108) was the highest since winter 2000–01 (WSI = 153). Consequently, nearly all of our radiocollared does that are seasonal migrators (i.e., mean distance between winter and spring-summer-fall home ranges is 8–16 km [5–10 miles]), which is about 72% of the total (DelGiudice, unpublished data), were induced to move to our winter range study sites this winter. This indicates that many more uncollared deer migrated to winter ranges and facilitated our relatively high capture success.

The fawn:doe capture ratio was the highest of the 15-year study (111 fawns:100 does), which was likely attributable to age-specific pregnancy rates of 90–100% for does ≥1.5 years old (DelGiudice, unpublished data), and positive effects on survival of 3 consecutive very mild winters (2001–02 to 2003–04, WSIs = 38–58) previous to this winter. Previously, the highest fawn:doe capture ratio (105:100) occurred during winter 2000–01, which was moderately severe (WSI = 153), but similarly followed an historically unprecedented 3 consecutive mild winters (WSI range = 45–57) (P. Bouley, State Climate Office, personal communication). Although the mortality rate of winter 2000–01 was relatively high (16.2%), its weather conditions had only a moderate negative effect on the subsequent reproductive success in spring 2001, as the fawn:doe capture ratio of winter 2001–02 remained relatively high (81:100). Actual fawn:doe capture ratios for 2000–01, 2001–02, and 2003–04 would be expected to be somewhat higher, as a portion of the deer were captured by net-gun, which involves a level of selection for adult females. During the study, the fawn:doe capture ratio has declined to as low as 32:100 (winter 1996–97), likely attributable to the preceding historically severe winter (1995–96, WSI = 183), during which the highest mortality rate (29.3%) of collared does occurred. Further, observations indicated that reproductive success of surviving does following severe winter 1995–96 was exceptionally low, thus a small number of fawns would have entered winter 1996–97.

Of the 91 deer captured during winter 2004–05, 35 new females (11 fawns, 24 adults) were recruited into the radiocollared study cohort. Including does already radiocollared when this winter began, 82 females have been monitored during December 2004–May 2005.

Ages and Reproductive Status of Study Deer

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 phase (1991–1995), the 4-year treatment phase (1996–1999), and thus far during the 6-year post-treatment phase (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). Equally as important, after 1991, mean age of deer on all 4 sites (pooled) also remained stable, and has ranged from 5.0 (± 0.4 [SE], n = 90) in 2001 to 7.1 (± 0.6, n = 62) years old in 1993 (Figure 1). During 2004, mean age was 6.5 (± 0.4) years old, compared to 6.0 (± 0.1) years old during the remainder of the study overall.

According to progesterone concentrations (≥1.6 ng/ml, Wood et al. 1986, DelGiudice, unpublished data), the pregnancy rate of captured adult (≥1.0 years old) females has remained consistently high (95.2%, n = 218) throughout the study, ranging from 79 to 100% during winters 1990–91 to 2001–02. Only 1 fawn has been assessed as pregnant by this method. However, pregnancy rates for does 1.5–15.5 years old have ranged from 87.5 to 100% (Figure 2). Mean serum progesterone concentrations differed (P < 0.05) between pregnant (3.8 ± 0.09, range = 1.6–8.9 ng/ml, n = 218) and non-pregnant (0.7 ± 0.16, range = 0–1.4 ng/ml, n = 11) does. There was no relationship (r^2 = 0.01, P = 0.52) between progesterone concentrations and julian day. However, there was a difference (P ≤ 0.05) in mean body mass at capture for pregnant (63.0 ± 0.7, range = 45.7–82.5 kg, n = 171) versus non-pregnant (54.6 kg ± 2.8, range = 43.3–69.1 kg, n = 10) does, which may be indicative of an effect of inadequate nutrition on conception during the breeding season.

Capturing the Variability of Winter Severity

Weather is one of the strongest environmental forces impacting wildlife nutrition, populations, and their numbers. For northern deer in the forest this becomes most evident during winter when diminished quantity, availability and quality of food resources, and severe weather conditions impose the most serious challenge to their survival. This long-term study continues to document highly variable winter weather conditions, which permits 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 are investigating (e.g., movements, distribution, food habits, cause-specific mortality, and age-specific survival). We are examining the variability of weather conditions in several different ways. Specifically, Figure 3 illustrates the Minnesota Department of Natural Resources' (MNDNR) WSI, which is calculated by accumulating a point for each day (temperature-days) with an ambient temperature ≤ -17.8° C (0° F), and an additional point for each day (snow-days) with a snow depth ≥ 38.1 cm (15"). The WSI for our study sites has now ranged from 38 (winters 2003–04) to 185 (winter 1995–96) over the past 15 winters. The WSI of winter 2004–05 was attributable in part to 57 snow-days, whereas, during 5 of the past 7 winters, accumulated snow-days were only 0–6. The biological significance of this is that depth of snow cover is the component of the WSI that has the greatest negative effect on deer survival (DelGiudice et al. 2002). However, the average snow depth just exceeded the WSI snow threshold (38 cm) throughout much of winter 2004–05; depth of cover was actually rather moderate compared to all other winters (Figure 4). The wide range of winter weather conditions captured during this study will enhance the value of all data interpretations relative to deer survival, other aspects of their ecology, and management implications. A severe winter during the post-treatment phase of the study remains elusive, and would undoubtedly prove valuable.

Mean daily minimum temperatures by month have been highly variable (Figure 4). To relate the variability of ambient temperature to deer in a more biologically meaningful or functional way, I calculated the *effective critical temperature* for an averaged size adult female deer (-7° C or 19.4° F), and the number of days per month when the maximum ambient temperature was at or below this threshold (Figure 5). At or below this temperature threshold, heat losses may exceed energy expenditure for standard metabolism and activity, with additional heat generated to maintain homeothermy (McDonald et al. 1973). On these days, a physiological (e.g., accelerated mobilization of fat reserves) or behavioral response (e.g., change in habitat use) by the deer would be necessary to meet this environmental challenge. Interestingly, the potential physiological challenges of ambient temperatures during January–March 2005 were greater than during this interval in any other year of the study (Figure 5). Similarly, I used a snow depth threshold of ≥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). This threshold is slightly higher than that used for the WSI, but similarly, this winter's snow conditions are considered moderate. Importantly, these snow-days (6) were minimal during March, which can be a most critical time relative to deer survival (DelGiudice et al. 2002). Clearly, there has been a pronounced variability of days during the study's 15 winters when it is biologically reasonable to expect that there were potentially serious energetic implications associated with ambient temperature or snow depth. It is noteworthy that extensive statistical analyses of age-specific survival and weather data from the first 6 years of this study (DelGiudice et al. 2002) showed that snow conditions (depth and density) impose a far greater challenge to survival than ambient temperature. However, during a very severe winter (e.g., 1996), the consequences of cold temperatures on individual deer with rapidly depleting or exhausted fat reserves should not be underestimated. Our analyses of 13 years of data have shown that variation in winter severity has direct and indirect influences on the age-specific hazard (i.e., instantaneous probability of death) of deer (DelGiudice et al. 2005*a*).

Status and Cause-Specific Mortality of Study Deer

The status/fate of study deer through 31 December 2004 is shown in Figure 6. 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 Dec–May) by the total number of deer that were collared and monitored during that period. 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 2004, annual mortality rates of collared females ranged from 9.1 to 47.6% (Figure 7). The mortality rate for 2004 was rather typical at 23.3%. As has been mentioned in previous reports, the atypical 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 older females, severe weather conditions, and wolf predation contributed to the higher mortality rates (Figure 7). The number of antlerless permits issued varied considerably from 1991 to 2004. As reflected by the hunter-caused mortality rates in Figure 7, 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, in 1999 there was an increase in hunter-caused mortality, and this increased further to the study's second highest level during 2000 (19.4%, Figure 7). During 2003 and 2004, antlerless permits were unlimited, and hunter-caused mortality rates were among the highest of the study (17.0 and 16.1%). Although hunter harvest mortality is primarily a function of antlerless permit numbers, the more than 2 times higher percent harvest mortality in 1992 compared to 2003 was likely influenced by the markedly smaller sample of collared does entering the 1992 hunting season (n = 35) than in 2003 and 2004 (n = 53 and 62). Wolf-caused mortality of females in 2004 was the second lowest of the study (Figure 7). 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%, with the maximum occurring during 2001. 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 female deer killed by wolves during 11 of the first 14 winters of the study was 6.0 (+ 1.8, n = 9)– 11.7 (+ 1.7, n = 8) years old Mean age of deer killed by wolves during winter 2003–04 was 9.9 ($\pm$ 2.7, n = 4) versus 7.9 ($\pm$ 0.6, n = 63) during the previous 13 years.

Most of the annual non-hunting mortality of study deer occurred during winter. Typically, winter mortality of collared adult female deer has been low (2.0–12.5%, Figure 8). The highest winter mortality rates (16.2–29.3%) of does have occurred during 3 of the 4 most severe winters (1993–94, 1995–96, and 2000–01, Figure 8). Mortality during winter 2004–05 was among the lowest of the study (5.4%). The relationship between WSI and percent winter mortality of adult female deer continued to be reasonably strong (r^2 = 0.47, P = 0.005, Figure 9). Predation, and wolf predation specifically, were responsible for a mean 77.1% ($\pm$ 7.2, range = 0.00–100%, n = 15) and 68.05% ($\pm$ 7.8, range = 0–100%, n = 15), respectively, of the winter (Dec–May) mortality of collared fawn and adult females throughout the 15-year study period. Monthly wolf predation of females was greatest during March and April (Figure 10).

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 the 1950–60s, but just 5–6 years prior to initiation of the study, had re-entered and became re-established. When the study began in winter 1990–91, this area was on the leading edge of wolf range expansion in Minnesota. Since spring 1993, we have captured and radiocollared 50 (28 females, 22 males) wolves from 7–9 packs which 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 11 wolf deaths versus 5 deaths by natural causes (Figure 11).

Based on aerial observations, pack sizes have ranged from 2 to 7 members. Current status of each of the collared wolves is listed in 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. We will capture and radiocollar additional wolves this summer. 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 are updating the coverage to account for any changes in type classification associated with succession during the past 14 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 Lake and Shingle Mill Lake 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 winters, locations were more concentrated in dense conifer cover. Data will be analyzed more rigorously in the upcoming year.

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Table 1. History of radiocollared gray wolves, north central Minnesota, 1993–2005. (Ad=adult, juv=juvenile).

WOLF NO.	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	F	JUV	ON THE AIR	
		MAY 2004				
2060	SHINGLE MILL	AUG 1996	F	JUV	LOST SIGNAL	FEB 1, 2000
		JUL 1998 - RECAPTURED				
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	F	AD	DISPERSED	MAR 17, 2004
		RECAPTURED JUN 2003				
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	ON THE AIR	
2146	WILLOW	AUG 2003	F	JUV	ON THE AIR	
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	DISPERSED	JAN 2005
2268	INGUADONA	MAY 2004	M	AD	UNKNOWN MORTALITY	JAN 19, 2005
2269	WILLOW	MAY 2004	M	AD	DISPERSED	JUN 2004

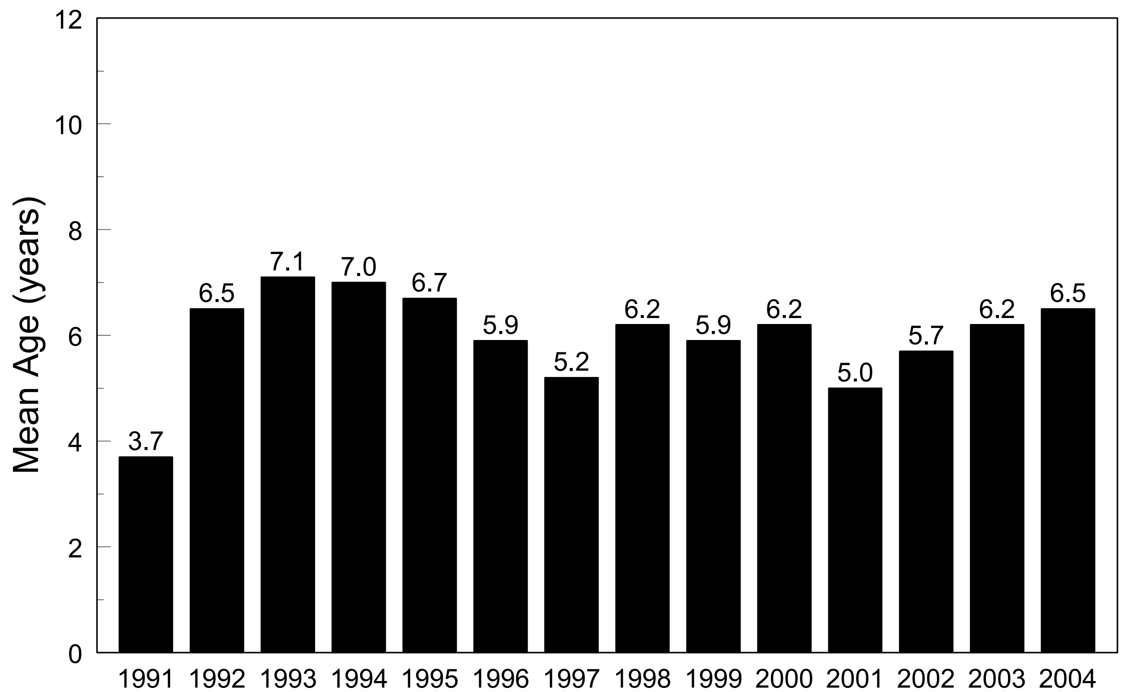


Figure 1. Mean age of radiocollared female white-tailed deer among years, north central Minnesota, 1 January 1991–31 December 2004. (Sample sizes were 22, 34, 62, 66, 54, 76, 74, 49, 55, 48, 90, 84, 75, and 81, respectively.)

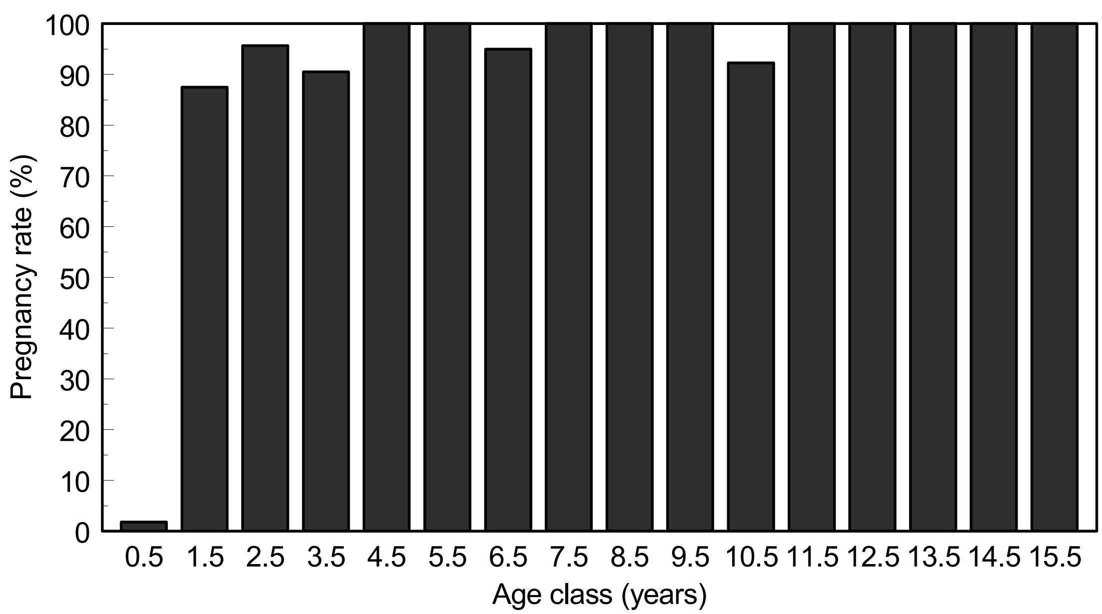


Figure 2. Age-specific pregnancy rate of radiocollared white-tailed deer (4 study sites pooled) in north central Minnesota, winters 1991–2002. (Sample sizes were 55, 48, 23, 21, 18, 21, 20, 13, 9, 11, 13, 8, 11, 5, 4, and 4 for yearly age classes, respectively.)

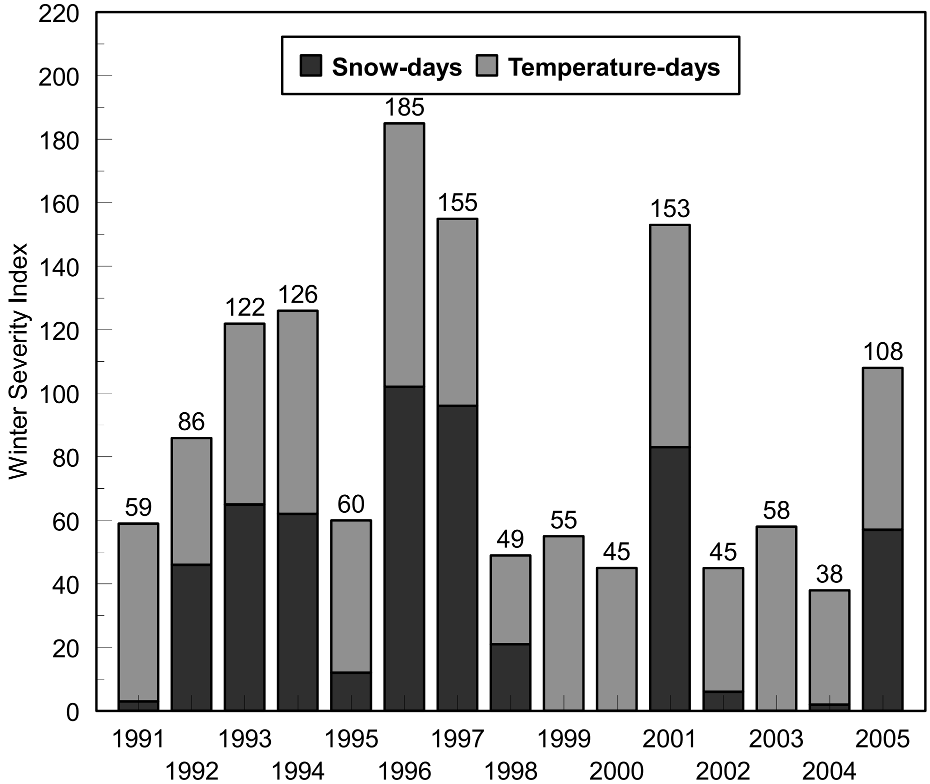


Figure 3. Winter severity index for white-tailed deer study sites, north central Minnesota, winters 1990–91 to 2004–05. One point is accumulated for each day with an ambient temperature $\leq$ -17.8^{o} C (temperature-day), and an additional point is accumulated for each day with snow depths $\geq$38.1 cm (snow-day).

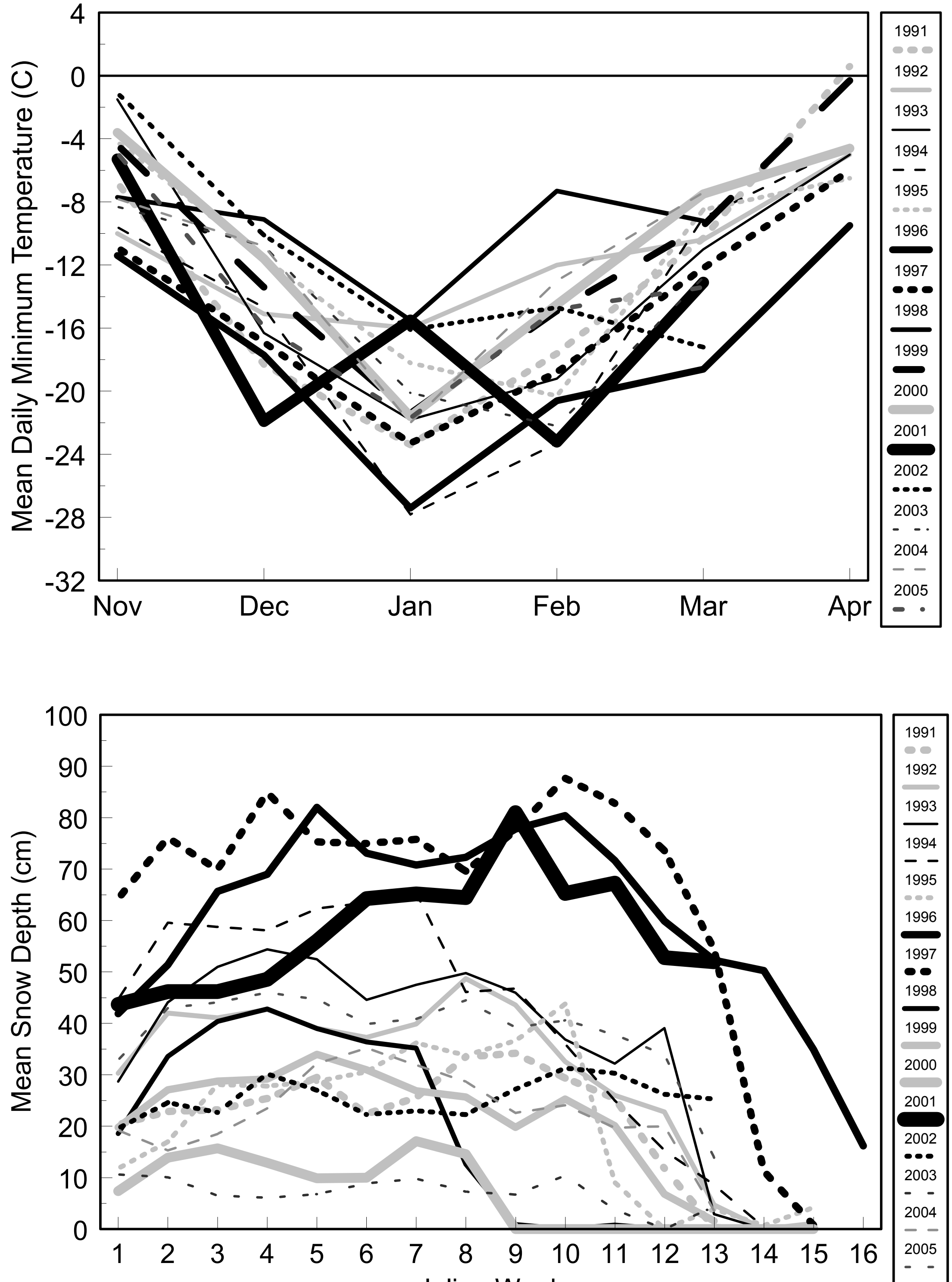
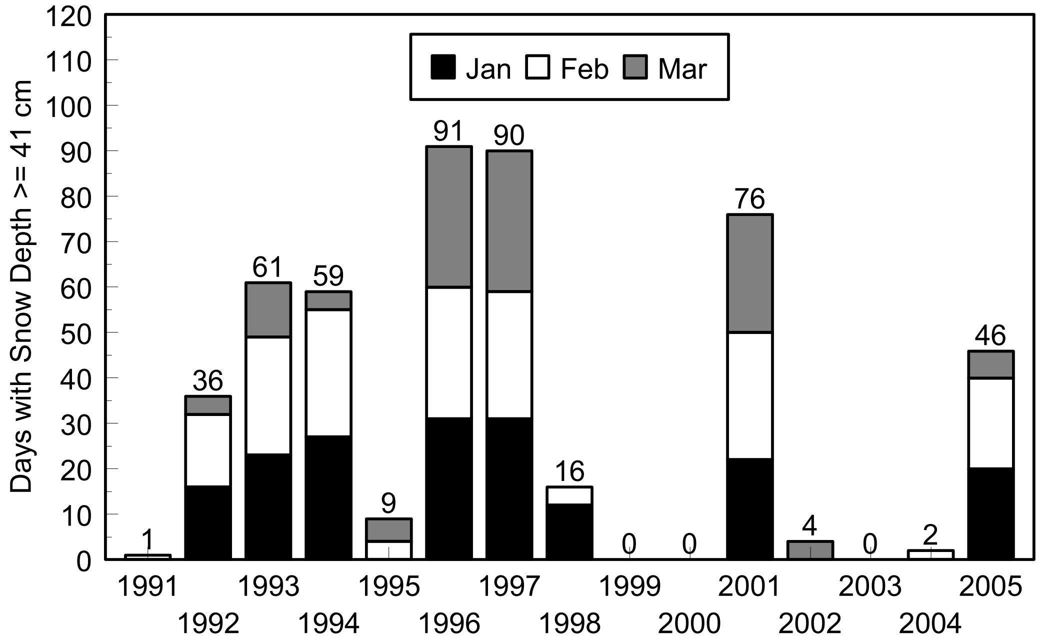


Figure 4. Mean daily minimum ambient temperature (top, Nov–Apr 1990–2005) and mean weekly (julian) snow depths (bottom, Jan–Apr 1991–2005) for white-tailed deer study sites, north central Minnesota.



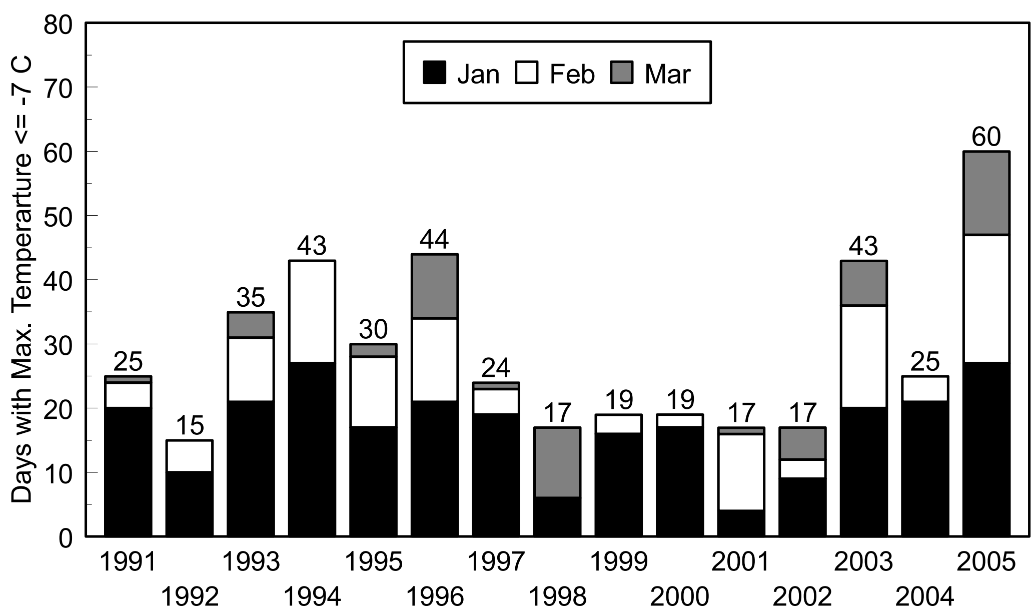


Figure 5. Number of days with snow depths ≥41 cm (top) and maximum ambient temperatures ≤ -7° C (bottom, *effective critical temperature* for an average size doe [60 kg]), north central Minnesota, January–March 1991–2005.

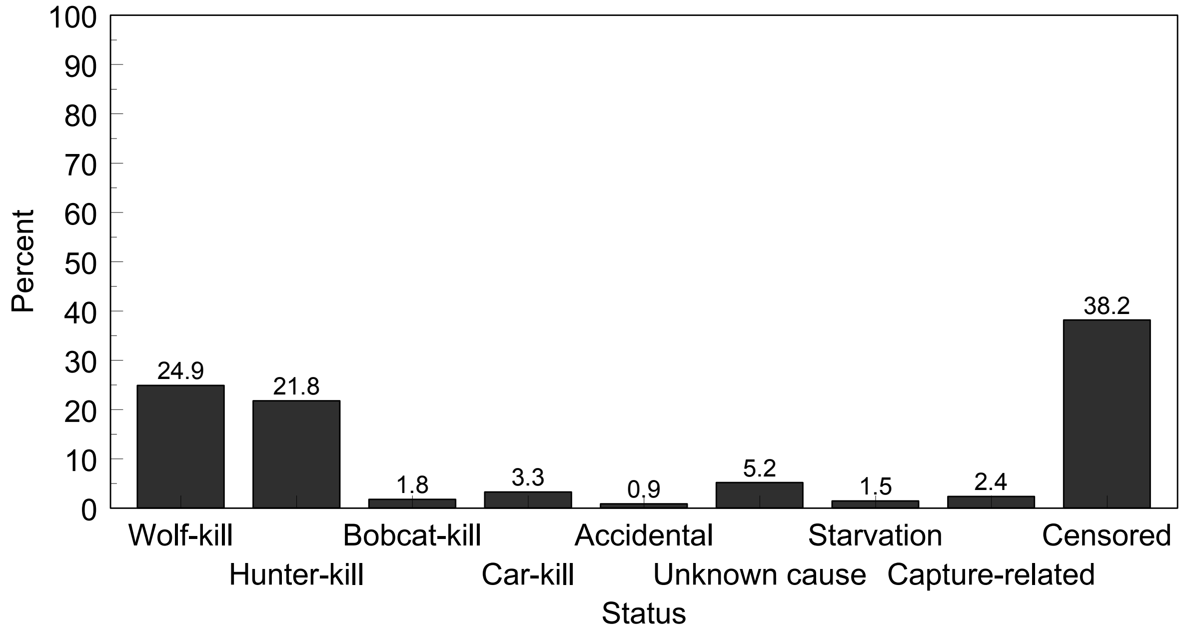


Figure 6. Status of radiocollared female deer, north central Minnesota, January 1991–December 2004. Censored deer include those that were still alive on 31 December 2004, or whose radio signals have been lost to monitoring (e.g., radio failure, dispersal from region of the study sites).

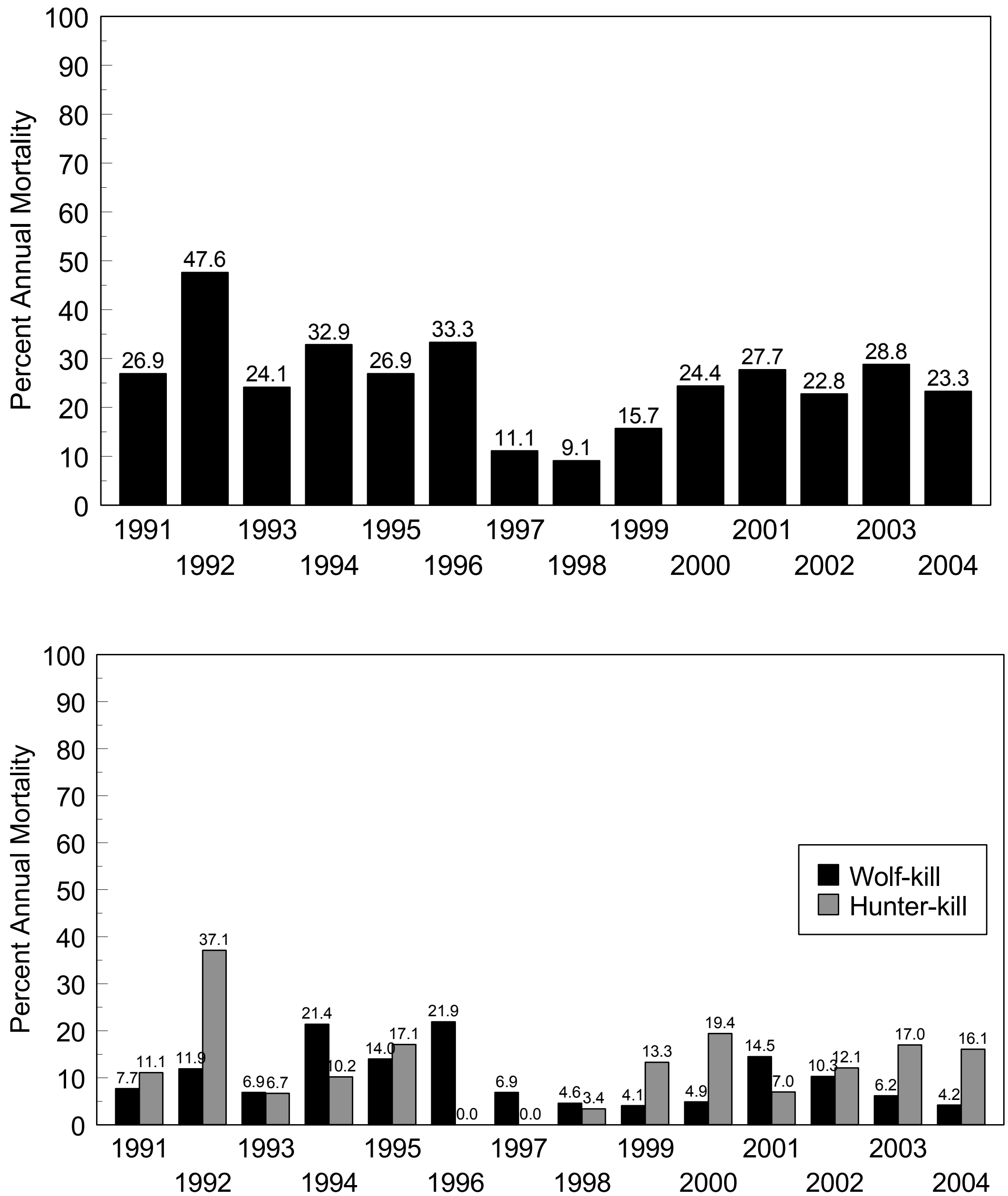


Figure 7. Annual (1 Jan–31 Dec) percent mortality of radiocollared, female white-tailed deer (top) and annual percent mortality attributable to wolf predation and hunter harvest (bottom, 4 sites pooled), north central Minnesota, 1991–2004. (Sample sizes were 26, 42, 58, 70, 52, 66, 72, 44, 51, 41, 83, 79, 66, and 73, respectively. Hunter harvest was calculated with the maximum number of collared females entering November; no antlerless permits were issued in 1996 and 1997, and very few were issued in 1998.)

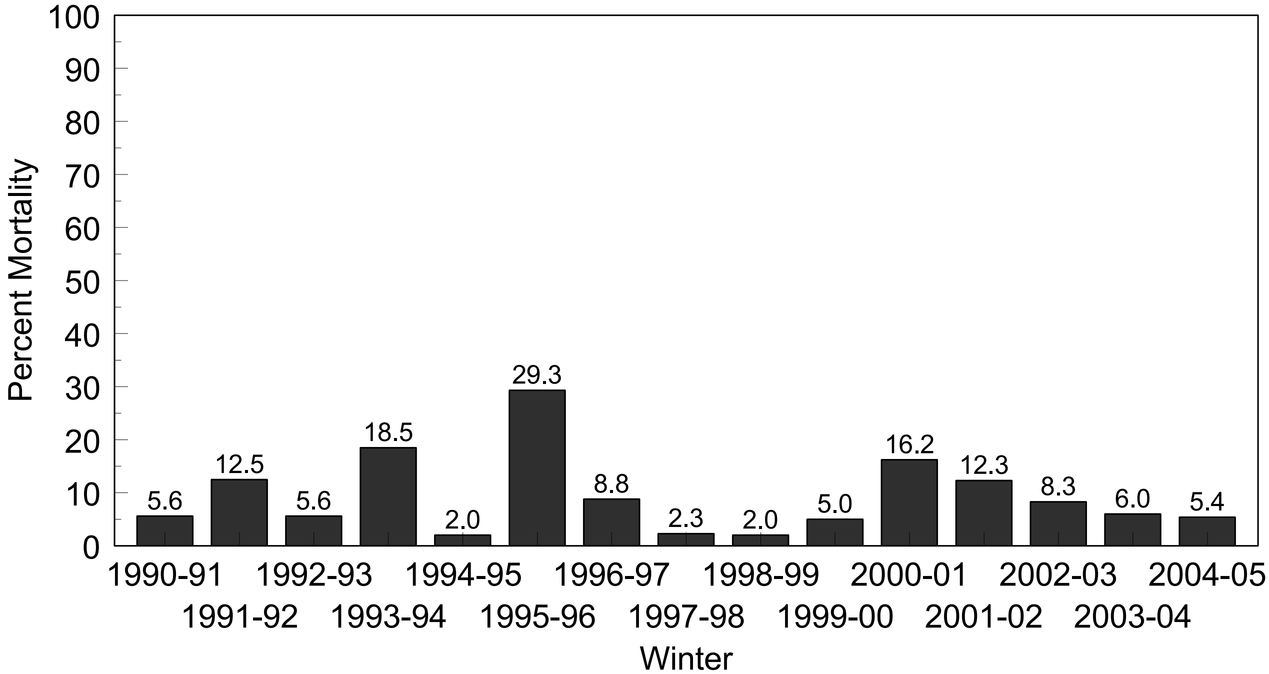


Figure 8. Percent winter mortality (Dec–May) of radiocollared, adult (≥1.0 year old) female white-tailed deer (4 sites pooled), north central Minnesota, winters 1990–91 to 2004–05. (Sample sizes were 18, 40, 54, 65, 50, 58, 68, 43, 49, 40, 68, 73, 60, 67, and 74, respectively; no deer were radiocollared during Dec 1990.)

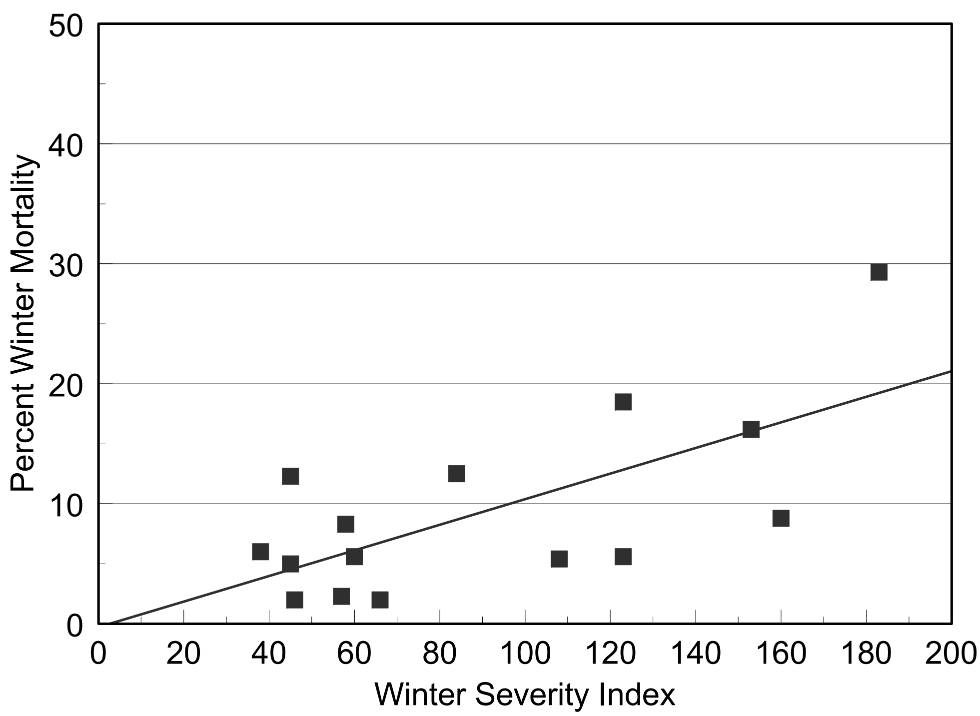


Figure 9. Relationship between MNDNR winter severity index (Nov–May) and percent winter (Dec–May) mortality (Y = -0.2820 + 0.1068x, r^2 = 0.47, P = 0.005) of radiocollared, adult (≥1.0 year old), female white-tailed deer (4 sites pooled), north central Minnesota, winters 1990–91 to 2004–05.

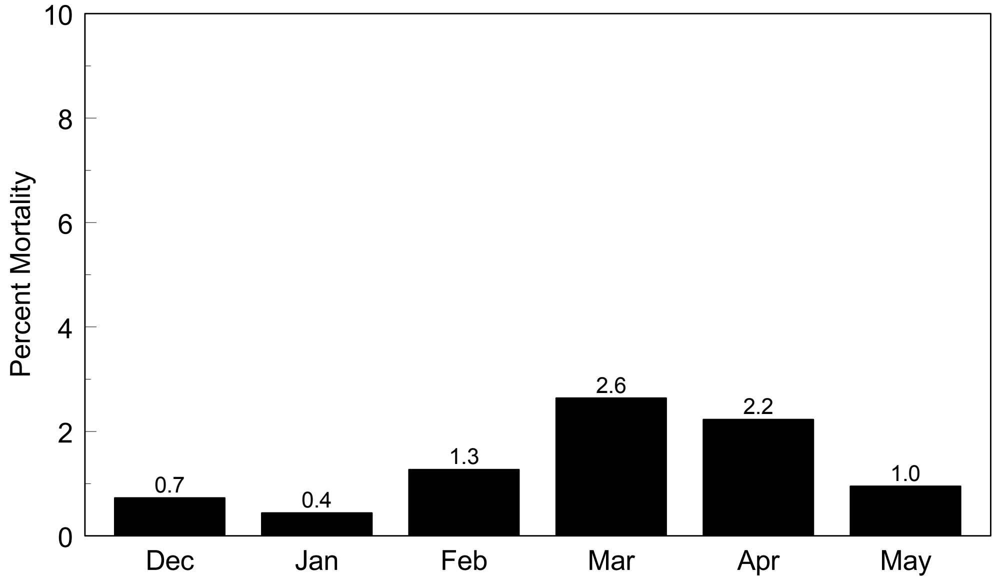


Figure 10. Monthly mortality of radiocollared female (fawns and adults) white-tailed deer by wolves (4 sites pooled), north central Minnesota, winters 1990–91 to 2004–05.

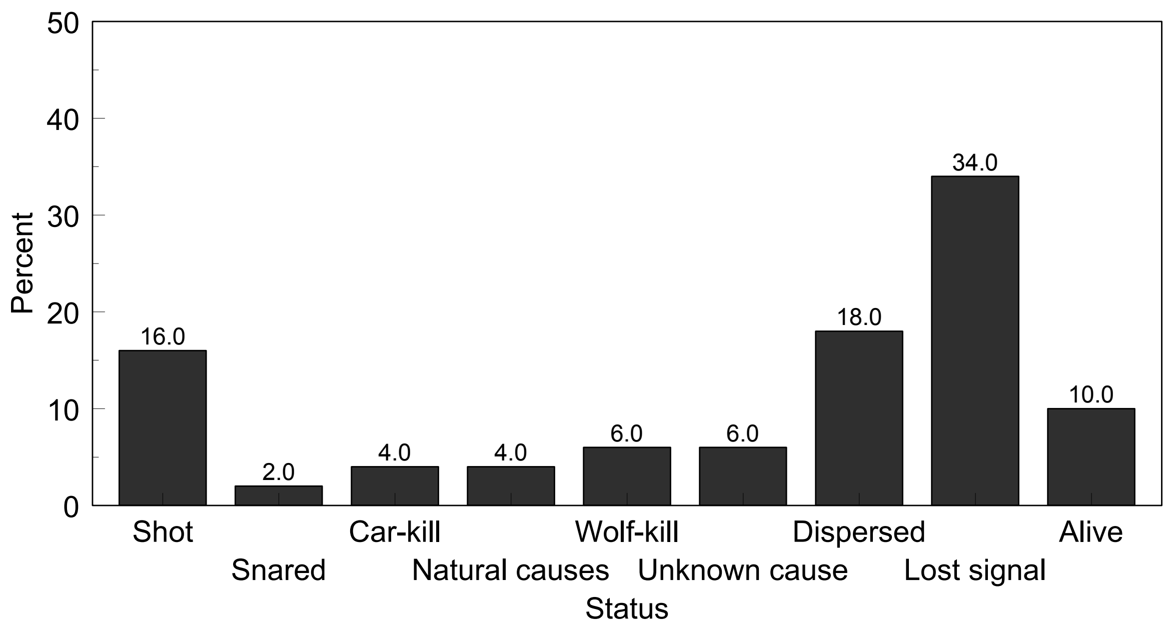


Figure 11. Status of radiocollared wolves, north central Minnesota, 1993–2005.

WINTER SEVERITY, BODY COMPOSITION, AND SURVIVAL OF WHITE-TAILED DEER

Michelle Carstensen Powell, and Glenn D. DelGiudice

Abstract: Understanding the relation between winter severity and survival of free-ranging deer (*Odocoileus* spp.) requires close examination of the functional relation between environmental conditions and the nutritional status of deer. In this study, we determined the body composition of free-ranging adult (>1.0 year old) female white-tailed deer (*O. virginianus*) and fawns (< 1.0 year old) during 2 winters of varying severity; winter 2000–01 was severe (Winter Severity Index [WSI] = 153), and winter 2001–02 was mild (WSI = 45). We used deuterium-dilution to estimate total body water of 39 adult females and 37 fawns (18 females, 19 males) and employed predictive equations derived from 15 sacrificed deer (7 adults, 8 fawns) to calculate their ingesta-free body composition (i.e., total body water, fat, protein, and ash). Mean total fat (%) decreased from mid- to late-winter for adults and fawns in 2001 (adults = 7.9, ± 0.1% [SE] to 7.2 ± 0.3%; fawns = 6.9 ± 0.4% to 5.3 ± 0.4%) and 2002 (adults = 7.9 ± 0.3% to 6.5 ± 0.2%; fawns = 6.2 ± 0.3% to 5.0 ± 0.7%). These changes were accompanied by declines in protein mass (0.6 and 1.0 kg for adults and 1.6 and 0.2 kg for fawns in 2001 and 2002, respectively). Fat reserves did not differ between years for either adults or fawns. Similarly, there was a minimal effect of winter severity on blood profiles of deer; however, cholesterol (in combination with julian day) was inversely related (R^2 = 0.43) to fat (%) of adults and serum urea nitrogen (in combination with julian day) was inversely related (R^2 = 0.36) to fat (%) of fawns. Survival of adult females was similar between years with the majority of deaths occurring between February and April; wolf (*Canis lupus*) predation was the primary cause of death. Age appeared to influence adult survival as 70% of adults that died were >10 years old. Winter severity may have played a role in fawn survival. Nearly half (47%) of the fawns died during late-winter to early-spring of 2001, while all survived in the same interval of 2002. Fat reserves were not reliable predictors of deer survival in this study. Eighty-two percent and 71% of adults in 2001 and 2002 that were determined to have low fat reserves in mid- or late-winter (i.e., below the median body fat percentage of animals sampled) survived winter and early-spring; 75% of fawns with low fat reserves in 2001 also survived. Absence of a biologically significant relation of body condition of deer relative to winter severity may be a result of a cumulative effect of several mild winters preceding 2000–01, that enabled deer to accumulate sufficient energy reserves to withstand prolonged and severe climatic stress. Further study of the relation between winter severity and body condition of northern deer during several successive winters of varying severity may be warranted, as well as consideration of other potentially influential factors (e.g., migration behavior, predator density, habitat quality).

**From the abstract of Chapter 1 of Michelle Carstensen Powell's Ph.D. Dissertation , Department of Fisheries, Wildlife, and Conservation Biology, University of Minnesota, St. Paul.*

SURVIVAL AND CAUSE-SPECIFIC MORTALITY OF WHITE-TAILED DEER NEONATES IN RELATION TO WINTER SEVERITY AND NUTRITIONAL CONDITION OF THEIR DAMS

Michelle Carstensen Powell, and Glenn D. DelGiudice

Abstract: Winter severity is thought to play a key role in newborn white-tailed deer (*Odocoileus virginianus*) survival, yet few studies of free-ranging ungulate populations have been able to establish a link between maternal body condition and subsequent survival of offspring. Free-ranging deer neonates (n = 66) were captured from radiocollared dams that survived winters of varying severity; winter 2000–01 was severe (Winter Severity Index [WSI] = 153) and 2001–02 was historically mild (WSI = 42). Mean dates of birth (26 May ± 1.7 [SE] days and 26 May ± 1.3 days) and estimated birth-mass (2.8 ± 0.1 kg and 3.0 ± 0.1 kg) of neonates 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, 1–4, and 4–12 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 predator-related deaths of neonates in springs 2001 and 2002; whereas, 50% of neonate mortality in 2002 was caused by bobcats (*Felis rufus*). Wolves (*Canis lupus*) played a minor role in neonate mortality, accounting for only 5% of predator-related deaths. Birth characteristics and blood profiles of neonates were examined as possible predictors of survival. Serum urea nitrogen:creatinine (SUN:C) ratio was associated with neonate survival to 1, 4, and 12 weeks of age; with elevated levels reported in survivors (28.6–35.2) compared to nonsurvivors (22.1–27.0). No relation between winter fat reserves (i.e., percent ingesta-free body fat) of dams and survival of their neonates the subsequent spring was observed; however, dams (n = 5) of neonates that died within 4 weeks of age had greater (P< 0.05) concentrations of SUN (19.8 ± 3.4 vs 11.1 ± 1.1 mg/dL), C (2.7 ± 0.1 vs 2.3 ± 0.1 mg/dL), and SUN:C (7.2 ± 0.9 vs 4.8 vs 0.4) 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 suggested that body mass at birth 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 allow a greater understanding of the factors that may predispose neonates to natural sources of mortality.

**From the abstract of Chapter 2 of Michelle Carstensen Powell's Ph.D. Dissertation, Department of Fisheries, Wildlife, and Conservation Biology, University of Minnesota, St. Paul.*

BIRTH-SITE CHARACTERISTICS, HABITAT USE, AND SPATIAL RELATIONS OF WHITE-TAILED DEER NEONATES

Michelle Carstensen Powell, and Glenn D. DelGiudice

Abstract: Little is known about birth-site characteristics of free-ranging white-tailed deer (*Odocoileus virginianus*) in northern climates, or whether use of fawning habitat is related to neonate survival. In this study, we located 31 birth-sites of fawns in north central Minnesota during springs 2001 (n = 17) and 2002 (n = 14), and captured 41 neonates. Seven dams lost 1 or more fawns within 1 week of birth; 5 neonates were killed by predators and 3 died of unknown causes. Birth-site characteristics, including vegetative cover-types, distances to roadways and water, and concealment cover were highly variable; and none of these factors had an apparent influence on neonate survival. Neonates used a variety of cover-types within 3 weeks post-parturition, including hardwood and conifer stands, open areas, and they also used residential communities. Spatial relations of neonates to their dam, birth-site and siblings were assessed.

**From the abstract of Chapter 3 of Michelle Carstensen Powell's Ph.D. Dissertation,*
Department of Fisheries, Wildlife, and Conservation Biology,
University of Minnesota, St. Paul.

A LONG-TERM AGE-SPECIFIC SURVIVAL ANALYSIS OF FEMALE WHITE-TAILED DEER

Glenn D. DelGiudice, John Fieberg, Michael R. Riggs[1], Michelle Carstensen Powell, and Wei Pan[2]

Abstract: We conducted a 13-year survival (i.e., time survived since birth) and cause-specific mortality study, divided into 2 phases (Phase I = yr 1-6; Phase II = yr 7-13), of 302 female white-tailed deer (*Odocoileus virginianus*) ≥0.6 years old at capture. The study spanned a period of extreme variability in winter severity (maximum winter severity indexes of 45-195) and hunting pressure. Most studies of survival and cause-specific mortality of northern deer have assumed constant survival rates for adults (≥1.0 yr old pooled) of a sex and examined fawns (0.6≤ x ≤ 1.0 yr old) separately. We observed U-shaped hazard (i.e., instantaneous risk of death) curves for both phases of the study, indicating the risk of death is highest for younger and older individuals. The estimated hazard for Phase II was generally lower and relatively constant for adults 2-10 years old compared to Phase I, where the curve began ascending at age 6 years. This difference likely reflected differences in winter severities, associated changes in the magnitude of wolf (*Canis lupus*) predation, and changes in hunting pressure between the 2 phases. The age distribution of our study cohort was relatively stable over the study period. Subsequently, when 76 neonates (i.e., ≤ 0.6 yr old) were included in the study cohort, the descending arm of the all-causes hazard began its descent at a hazard rate of 2.3 (vs 1.0 without neonates), clearly demonstrating that the greatest risk of mortality occurs in the first 1 year of life. We compared cumulative survival estimates for these data using the generalized Kaplan-Meier (GKM) and the iterative Nelson estimator (INE) and illustrate the potential for bias when applying the GKM to left-truncated data. Median age of survival for females was 0.8 years old (90% confidence interval [CI] = 0.79-1.45 yr old) using the INE and 0.4 years old (90% CI = 0.8-1.4 yr old) using the GKM. Lastly, we use a simulation approach to examine the potential for bias resulting from pooling adults (using the U-shaped hazard function to determine reasonable "age classes" for pooling). These simulations suggest that models using the constructed discrete time variable give nearly unbiased estimates and provide support for using age-specific hazards to determine the reliability of adult age-pooled survival estimates. However, we caution that assessed cause-specific mortality may vary with environmental variability, variation of human-related activities, and age distribution of the study cohort.

**Abstract of paper accepted by the Journal of Wildlife Management.*

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MYCOBACTERIUM AVIUM SUBSP PARATUBERCULOSIS FROM FREE-RANGING DEER AND RABBITS SURROUNDING MINNESOTA DAIRY HERDS

Eran A. Raizman[1], Scott J. Wells[1], Peter A. Jordan[2], Glenn D. DelGiudice, and Russell R. Bey[3]

***Abstract*:** The objectives of this study were to (1) estimate the prevalence of *Mycobacterium avium subsp paratuberculosis* (MAP) among white-tailed deer (*Odocoileus virginianus*) and eastern cottontail rabbits (*Sylvilagus floridanus*) surrounding infected and noninfected Minnesota dairy farms using fecal culture and (2) describe the frequency of use of farm management practices that could lead to potential transmission of infection between these species. Fecal samples from cows and the cow environment were collected from 108 Minnesota dairy herds, and fecal pellets of free-ranging deer and rabbits were collected from locations surrounding 114 farms; all samples were tested using bacterial culture. In addition, a questionnaire was administered to 114 herd owners. Sixty-two percent of the dairy herds had at least 1 positive fecal pool or environmental sample. A total of 218 rabbit samples were collected from 90% of the herds, and 309 deer samples were collected from 47% of the herds. On each of 2 sampled farms (4%), 1 deer fecal sample was MAP positive. Both farms had culture positive cow fecal pool and cow environment samples. On each of 2 other farms (2%), 1 rabbit fecal sample was culture positive to MAP, with 1 of these farms having positive fecal pools and environmental samples. Pasture was used on 79% of the study farms as a grazing area for cattle, mainly for dry cows (75%) and bred or prebred heifers (87%). Of the 114 farms, 88 (77%) provided access to drylot for their cattle, mainly for milking cows (77/88; 88%) and bred heifers (87%). Of all 114 farms, 20% and 25% estimated as daily the probability of physical contact between cattle manure and deer or rabbits, respectively. Possible contact between cattle manure and deer or rabbits was estimated to occur primarily from March through December. The frequency of pasture or drylot use and manure spreading on crop fields may be important risk factors for transmission of MAP among dairy cattle, deer, and rabbits. Although the MAP prevalence among rabbits and deer is low, their role as MAP reservoirs should be considered, especially in proximity to cattle herds with very low or zero prevalence.

**Abstract of paper accepted by the Canadian Journal of Veterinary Research.*

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CHRONIC WASTING DISEASE SURVEILLANCE PROGRAM 2002–2004

Michael DonCarlos, Michelle Carstensen Powell, and Lou Cornicelli

SUMMARY OF FINDINGS

In response to the discovery of chronic wasting disease (CWD) in wild Wisconsin white-tailed deer (*Odocoileus virginianus*) and a Minnesota captive elk (*Cervus elaphus*) herd in 2002, the Minnesota Department of Natural Resources (MNDNR) developed a comprehensive wild deer CWD monitoring program that included surveillance of targeted animals (e.g., suspect or potentially sick deer exhibiting clinical signs or symptoms consistent with CWD), opportunistic surveillance (e.g., vehicle-killed deer), and hunter-killed deer surveillance. From 2002–2004, nearly 28,000 deer have been tested for CWD statewide with no positive results. The MNDNR will continue to monitor for the presence of CWD in suspect deer statewide, and may revisit the need to sample hunter-killed and opportunistic deer if the disease is detected in captive or wild deer in the future.

INTRODUCTION

Chronic wasting disease is a transmissible spongiform encephalopathy (TSE) that affects elk, mule deer (*Odocoileus hemionus*), and white-tailed deer (Spraker et al. 1997, Miller et al. 2000). TSEs are infectious diseases that alter the morphology of the central nervous system, resulting in a "sponge-like" appearance of this tissue (Williams and Young 1993). The etiological agent of CWD is believed to be an infectious protein, called a "prion." A healthy animal exposed to these prions may develop CWD (Miller et al. 1998); however, precise mechanisms and rates of CWD transmission are poorly understood. Incubation time of the disease, from infection to clinical signs, can range from a few months to nearly 3 years (Williams and Young 1980, Spraker et al. 1997, Miller et al. 1998). Clinical signs may include a loss of body condition and weight, excessive salivation, ataxia, and behavioral changes. Currently, there is no known treatment for the disease, and it is always fatal.

Chronic wasting disease was first discovered in captive mule deer in 1967, and then recognized in captive white-tailed deer and elk in 1978 (Williams and Young 1980). The disease has been diagnosed in captive cervid populations from Nebraska, Oklahoma, Kansas, Montana, Colorado, Wyoming, South Dakota, Wisconsin, New York, and Minnesota, USA, and Alberta and Saskatchewan, Canada (United States Animal Health Association 2001, Canadian Food Inspection Agency 2002). Within wild populations, CWD was historically confined to free-ranging deer and elk in the endemic area of northeast Colorado and southeast Wyoming (Miller et al. 2000, Williams et al. 2002). Recently CWD has been detected west of the continental divide in Colorado, and within wild deer populations of Nebraska, Wisconsin, Illinois, South Dakota, Utah, New Mexico, and New York. Generally, wild cervid CWD occurrences outside the endemic area have been located in close proximity to captive cervid facilities with past or present infected animals, except for 4 positive deer located at White Sands Missile Base, New Mexico.

Public health officials and the Centers for Disease Control in Atlanta, Georgia, have concluded there is no link between CWD and any neurological disease in humans. Furthermore, there is no evidence that CWD can be naturally transmitted to animals other than deer and elk. Experimental and circumstantial evidence suggests that transmission of the disease is primarily through direct contact with infected animals (Miller et al. 1998). However, because of the possibility of persistence of prions in the environment, transmission in a contaminated environment may also be possible.

Wildlife disease control strategies need to be based on an understanding of specific disease etiology and epidemiology. Once established, most infectious diseases are extremely difficult to eliminate from wild populations. Because the epidemiological attributes of CWD remain uncertain, the MNDNR has positioned itself to acquire as much information as possible about CWD, including effective control strategies. Since CWD has not been detected in Minnesota's free ranging white-tailed deer population, the opportunity to assess the progress of the disease in other states, and observe the outcomes of selected management alternatives is of great value. Given the extended incubation period associated with CWD, the apparent capacity for lateral transmission, and the potential contributions from environmental contamination (Miller et al. 2004), it is imperative that CWD be identified, isolated, and controlled as rapidly as possible following detection within a population.

OBJECTIVES

- Determine if CWD is present in Minnesota's wild deer population; and
- Continue to monitor occurrences of CWD in wild or captive cervids in bordering states.

METHODS

Hunter-Killed Deer Surveillance

Power analysis was used to determine sample sizes for each Deer Permit Area (DPA) to ensure a $\geq$ 95% probability of detecting CWD, given a 1% infection rate (assuming a random distribution of the disease among individuals within each sampling area). Approximately 300 deer were needed in each sampling area (Table 1). Due to the prolonged incubation period of CWD, only deer $\geq$ 1.5 years of age were selected. Also, an attempt was made to collect samples equally across sex classes, as both sexes are susceptibility to CWD (Miller et al. 2000). To optimize the time spent collecting samples, collections occurred primarily during the Minnesota firearms deer season. Hunters voluntarily submitted all samples.

During the 2002 Minnesota deer hunting seasons, 16 sampling areas consisting of 17 DPAs were selected for CWD monitoring of hunter-killed deer (Figure 1a). Sampling areas were selected based on the following criteria: 1) proximity to cervid farms with known or suspected CWD positive animals, 2) proximity to CWD infected states, and 3) a statewide distribution. Approximately 100 registration stations within the selected DPAs were staffed for sample collection. Staff were trained to collect hunter data, including the specific harvest location, and to remove deer heads. All heads were given a unique ID number and transported to "extraction" sites. A total of 57 DNR wildlife research staff and veterinary students were trained to extract a portion of the brain, called the obex. All samples were transported to the Farmland Wildlife Population and Research Station in Madelia where they were inventoried, entered into a database, and sent to the University of Minnesota Veterinary Diagnostic Laboratory (VDL) for immunohistochemical (IHC) testing of the obex tissue for the presence of the abnormal prion protein.

During the 2003 Minnesota deer hunting seasons, 37 sampling areas consisting of 59 DPAs were selected for CWD monitoring of hunter-killed deer (Figure 1b). The sampling plan was modified from the 2002 scheme to include blocks of adjacent DPAs that enabled greater utilization of available personnel, and enhanced the efficiency of collection efforts. Approximately 130 registration stations within the selected DPAs were staffed for collection of deer heads, which were than transported to 11 extraction sites. Over 90 DNR wildlife management and research staff and veterinary students were trained to extract the medial retropharyngeal lymph nodes (MRPLN) from collected deer heads. The removal of MRPLN instead of obexes was another enhancement from the 2002 collection year, which resulted in a faster extraction process and less expensive testing protocol. All samples were transported to the Farmland Wildlife Population and Research Station in Madelia where they were inventoried, entered into a database, and sent to the University of Minnesota VDL for enzyme-linked immunosorbent assay (ELISA) testing of the lymph node tissue for the presence of the abnormal prion protein.

During the 2004 Minnesota deer hunting seasons, 50 sampling units consisting of 60 DPAs were selected for CWD monitoring of hunter-killed deer (Figure 1c). The sampling plan was meant to include all DPAs that were not previously sampled during the 2002 and 2003 collections. Nearly 130 registration stations within the selected DPAs were staffed for collection of MRPLN. This was a major change from previous collection years, where deer heads were removed at the registration stations and then transported to extraction sites for subsequent MRPLN removal. Removal of MRPLN on-site marked a vast improvement in the efficiency of sample collection and use of personnel (see Surveillance Costs section). Over 500 DNR wildlife staff, veterinary students, and volunteers were trained to extract MRPLN and gather relevant sample information from the hunters. Samples were transported to either the Farmland Wildlife Population and Research Station in Madelia or the Carlos Avery Wildlife Management Area in Forest Lake where they were inventoried, entered into a database, and sent to the University of Minnesota VDL for ELISA testing of the lymph node tissue for the presence of the abnormal prion protein.

Suspect Deer Surveillance

Suspect deer, which included animals in any DPA that displayed clinical symptoms thought to be consistent with CWD, as well as escaped captive cervids, were also tested for CWD. Obexes (2002) or MRPLNs (2003 and 2003) were extracted from suspect animals and submitted to the

RESULTS

Hunter-Killed Deer Surveillance

A total of 4,533 usable samples were collected from the selected sampling areas in 2002 and all were negative for the presence of CWD. Approximately 4.8% of collected samples were not usable. Females composed 40% of the samples, while males contributed the remaining 60%. Assuming that the samples were randomly collected from each DPA, results indicate that CWD infection rates $\geq$1% would have been detected in 7 of 16 sampling areas with $\geq$95% confidence, in 4 of 16 sampling areas with 92-95% confidence, and in 5 of 16 sampling areas with $\leq$90% confidence (Figure 2).

No positive results were detected in the 10,054 usable samples collected from the selected sampling areas in 2003. Approximately 2.8% of collected samples were not usable. Females composed 44% of the samples, while males contributed the remaining 56%. Assuming that the samples were randomly collected from each DPA, results indicate that CWD infection rates $\geq$ 1% would have been detected in 19 of 37 sampling areas with $\geq$ 95% confidence, in 13 of 37 sampling areas with 90-95% confidence, and in 5 of 37 sampling areas with $\leq$ 90% confidence (Figure 2).

In 2004, there were no positive results detected in the 13,038 usable samples collected from the selected sampling areas. The percentage of unusable samples (0.7%) marked a vast improvement in sample quality compared to the previous collection years. Females and males comprised 35% and 64% of the samples, respectively. Sex was not recorded in the remaining 1% of samples. Assuming that the samples were randomly collected from each DPA, results indicate that CWD infection rates $\geq$ 1% would have been detected in 21 of 50 sampling areas with ≥95% confidence, in 13 of 50 sampling areas with 90-95% confidence, and in 16 of 50 sampling areas with ≤90% confidence (Figure 2).

Suspect Deer Surveillance

From 2002 to 2004, 120 deer were sampled as suspects (Figure 3). All suspect samples were negative for the presence of the abnormal prion protein.

Surveillance Costs

The MNDNR conducted CWD surveillance for three deer seasons. Over that time, the protocol was changed to reflect new information about CWD (e.g., extraction of the obex versus MRPLN), and an overall desire to increase efficiency and decrease costs. In 2002, obexes were removed at centralized extraction stations. In total (including diagnostic fees), $857,600 was expended to collect 4,533 samples ($189/sample). In 2003, MRPLN were removed (again at centralized extraction stations) and $1.14 million was expended to collect 10,054 samples ($113/sample). In 2004, the protocol was changed to collect MRPLN at the registration stations, which eliminated the need to remove deer heads and transport them to centralized extraction stations. Ultimately, we expended $1.046 million and collected 13,038 samples ($80.50/sample). Diefenbach et al. (2004) reported spending $56/sample to remove heads and transport them to a centralized location in Pennsylvania for CWD testing. In Minnesota, excluding the diagnostic fees ($25/sample) and the veterinary student contract ($7.70/sample), it cost an estimated $48/sample to collect the sample (there were no head removal or transportation costs).

FUTURE CWD SURVEILLANCE EFFORTS

The MNDNR's effort to sample hunter-killed deer for the presence of CWD was highly successful, with the collection of nearly 28,000 samples statewide from 2002–2004. As the disease has not been detected in wild deer, there is no immediate need to continue the surveillance of hunter-killed deer. However, the sampling of suspect animals will continue throughout the state. If CWD is detected in wild or captive deer in the future, the MNDNR may revisit the need to sample hunter-killed or opportunistic deer.

We have attempted to collect CWD samples from the metro permit areas (228 and 337), but have been unsuccessful in obtaining statistically significant numbers. Consequently, an effort will be made again in 2006 to collect approximately 500 metro deer samples.

ACKNOWLEDGEMENTS

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Table 1. 2004 CWD sampling areas and sample size required to detect an infection rate of 1% with 95% confidence (98% confidence in combined DPAs)

Sampling Area (DPA)	Modeled Pre-Fawn Population Size	CWD Sample Size
Block 1		
104	14.564	295
107	15,160	295
110/283	5,940	377
205/214	6,000	377
211	10,986	294
213 (Red Lake)	18,500	300
Block 2		
167	10,560	294
168	12,308	294
170	17,095	295
172	15,785	295
197	9,600	293
242	14,003	295
243	9,734	294
245	16,907	295
246	19,708	296
Block 3		
244/251/287	23,594	386
297/298	9,997	382
402	1,939	276
407	2,657	282
408	2,519	281
409	3,936	287
420/421	3,261	367
Block 4		
152/157	23,287	386
156	13,216	295
159	12,496	295
174	10,020	294
183	10,605	294
222	5,562	290
225	9,656	294
249	9,538	293
337/338/339	5,442	376
228	3,555	286

Table 1. Continued.

Sampling Area (DPA)	Modeled Pre-Fawn Population Size	CWD Sample Size
Block 5		
440	2,515	281
442	3,143	284
443	1,852	275
448	3,263	285
449	4,187	288
450	1,795	275
451	2,198	279
452	1,592	272
453	2,770	283
454/455	4,134	372
456	2,349	280
457	1,964	275
458	1,644	273
459	3,701	286
461	2,597	282
463	1,494	270
464	1,885	276
466	2,979	284

Table 2. Summary of CWD samples collected by sampling area. Sample numbers include hunter-killed and opportunistic deer.

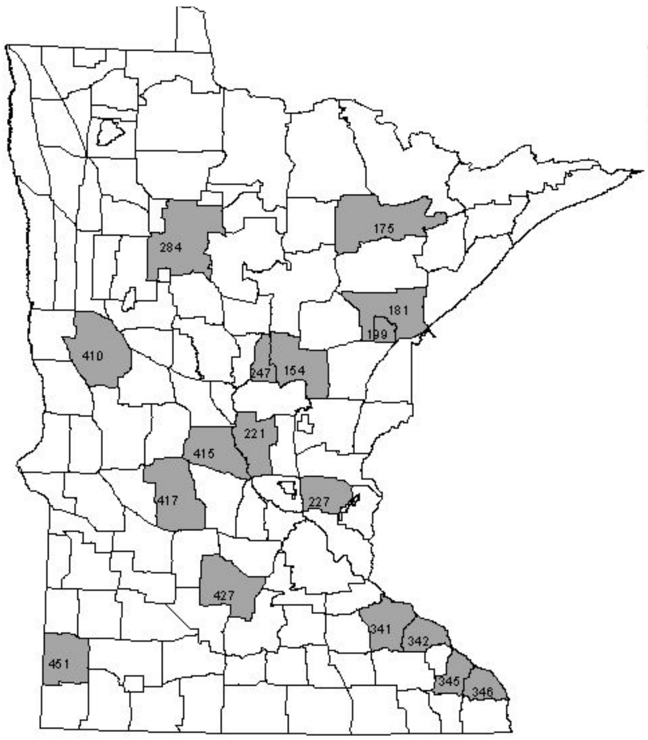
Sampling Area (DPA)	Total # of Samples Collected	Total # of Usable Samples	Negative Samples	Positive Samples	Unusable Samples	Female (%)	Male (%)	Unknown Sex (%)	Total Unusable Samples (%)	Confidence Level (1% Infection Rate)
BLOCK 1										
104	321	316	316	0	5	32.30	66.15	1.55	1.56	95.82
107	325	325	325	0	0	26.15	72.00	1.85	0.00	96.19
110/283	327	327	327	0	0	46.20	53.80	0.00	0.00	96.26
205/214	352	351	351	0	1	34.66	63.35	1.99	0.28	97.06
211	301	300	300	0	1	29.90	69.10	1.00	0.33	95.10
213	11	11	11	0	0	9.09	90.91	0.00	0.00	10.47
BLOCK 2										
167	245	245	245	0	0	28.16	69.80	2.04	0.00	91.48
168	344	344	344	0	0	33.72	64.53	1.75	0.00	96.85
170	358	358	358	0	0	30.45	69.55	0.00	0.00	97.26
172	381	376	376	0	5	37.53	61.68	0.79	1.31	97.72
197	253	252	252	0	1	28.02	67.70	4.28	0.43	92.06
242	191	191	191	0	0	46.07	53.40	0.53	0.00	85.33
243	292	289	289	0	3	43.15	56.85	0.00	1.03	94.52
245	328	328	328	0	0	36.67	63.03	0.30	0.00	96.30
246	399	399	399	0	0	40.35	59.40	0.25	0.00	98.19
BLOCK 3										
244/251/287	412	412	412	0	0	35.59	63.44	0.97	0.00	98.41
297/298	312	312	312	0	0	36.42	61.98	1.60	0.00	95.65
402	171	168	168	0	3	43.27	54.97	1.76	1.75	81.52
407	345	345	345	0	0	45.00	54.44	0.56	0.00	96.88
408	258	257	257	0	1	42.25	57.36	0.39	0.39	92.44
409	315	315	315	0	0	39.05	60.95	0.00	0.00	95.78
420/421	330	327	327	0	3	44.35	55.65	0.00	0.91	96.26
BLOCK 4										
152/157	514	513	513	0	1	30.29	68.93	0.78	0.19	99.42
156	295	295	295	0	0	34.69	64.97	0.34	0.00	94.84
159	290	290	290	0	0	40.34	57.93	1.73	0.00	94.58
174	296	294	294	0	2	34.46	65.54	0.00	0.68	94.79

Table 2. Continued.

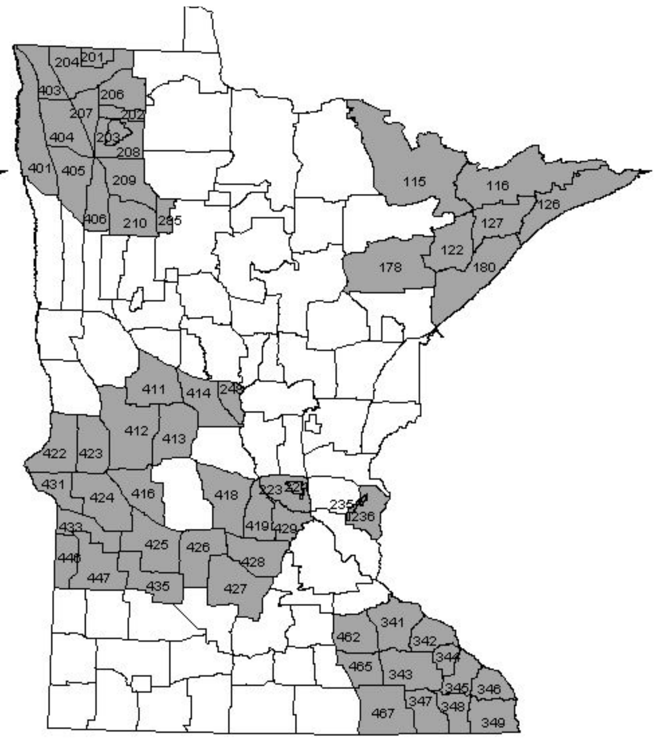
Sampling Area (DPA)	Total # of Samples Collected	Total # of Usable Samples	Negative Samples	Positive Samples	Unusable Samples	Female (%)	Male (%)	Unknown Sex (%)	Total Unusable Samples (%)	Confidence Level (1% Infection Rate)
183	359	359	359	0	0	29.53	69.64	0.83	0.00	97.29
222	291	291	291	0	0	33.68	65.98	0.34	0.00	94.63
225	299	299	299	0	0	33.78	65.22	1.00	0.00	95.05
249	296	295	295	0	1	40.54	57.09	2.37	0.34	94.84
337/338/339										
228										
BLOCK 5										
440	198	193	193	0	5	39.00	61.00	0.00	2.53	85.63
442	302	300	300	0	2	29.14	69.21	1.65	0.66	95.10
443	181	178	178	0	3	28.73	70.17	1.10	1.66	83.29
448	186	185	185	0	1	34.41	63.44	2.15	0.54	84.42
449	288	288	288	0	0	26.48	72.47	1.05	0.00	94.47
450	128	127	127	0	1	29.46	70.54	0.00	0.78	72.10
451*	324	300	300	0	24	29.33	70.67	0.00	7.41	95.10
452	145	144	144	0	1	37.93	62.07	0.00	0.69	76.48
453	173	170	170	0	3	27.96	71.51	0.53	1.73	81.89
454/455	377	373	373	0	4	31.65	66.75	1.60	1.06	97.65
456	294	293	293	0	1	42.81	57.19	0.00	0.34	94.74
457	237	234	234	0	3	35.86	63.29	0.85	1.27	90.48
458	167	167	167	0	0	23.35	76.65	0.00	0.00	81.33
459	284	281	281	0	3	24.30	75.35	0.35	1.06	94.06
461	226	223	223	0	3	42.15	57.40	0.45	1.33	89.37
463	135	133	133	0	2	41.61	58.39	0.00	1.48	73.73
464	86	84	84	0	2	37.21	61.63	1.16	2.33	57.01
466	184	181	181	0	3	36.67	62.78	0.55	1.63	83.78
2004 Totals	**13,126**	**13,038**	**13,038**	**0**	**88**	**35.0**	**64.0**	**1.0**	**0.67**	

*Includes 215 samples (191 usable, 24 unusable) samples collected in 2002.

a) 2002 DPAs



b) 2003 DPAs



c) 2004 DPAs

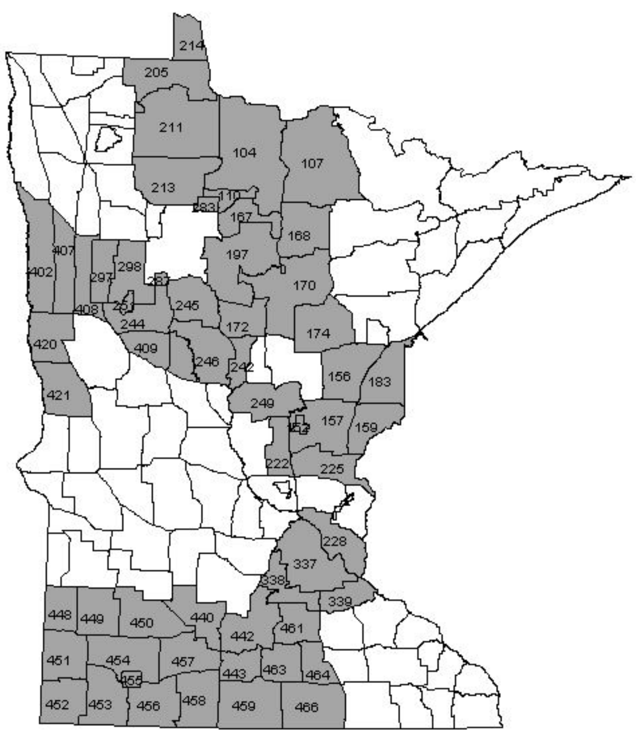


Figure 1. Sampling areas, denoted by Deer Permit Area (DPA), selected for chronic wasting disease surveillance of hunter-killed deer in 2002 (a), 2003 (b, and 2004 (c).

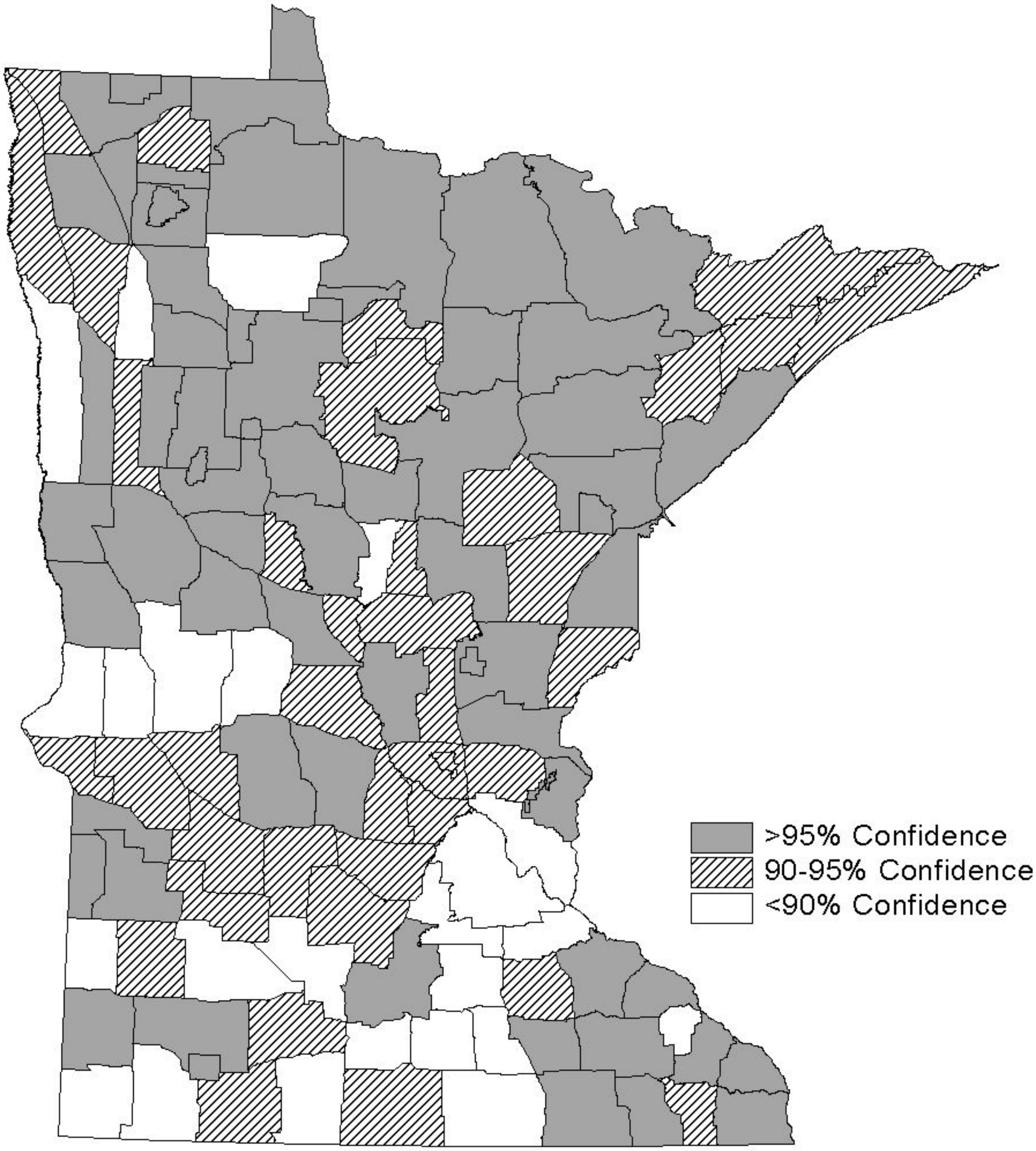


Figure 2. Probability of detecting the presence of chronic wasting disease CWD), given a 1% infection rate, in white-tailed deer sampled in Deer Permit Areas in Minnesota, 2002–2004. Confidence level was based on the assumption of a random distribution of CWD among individuals within each sampling area.

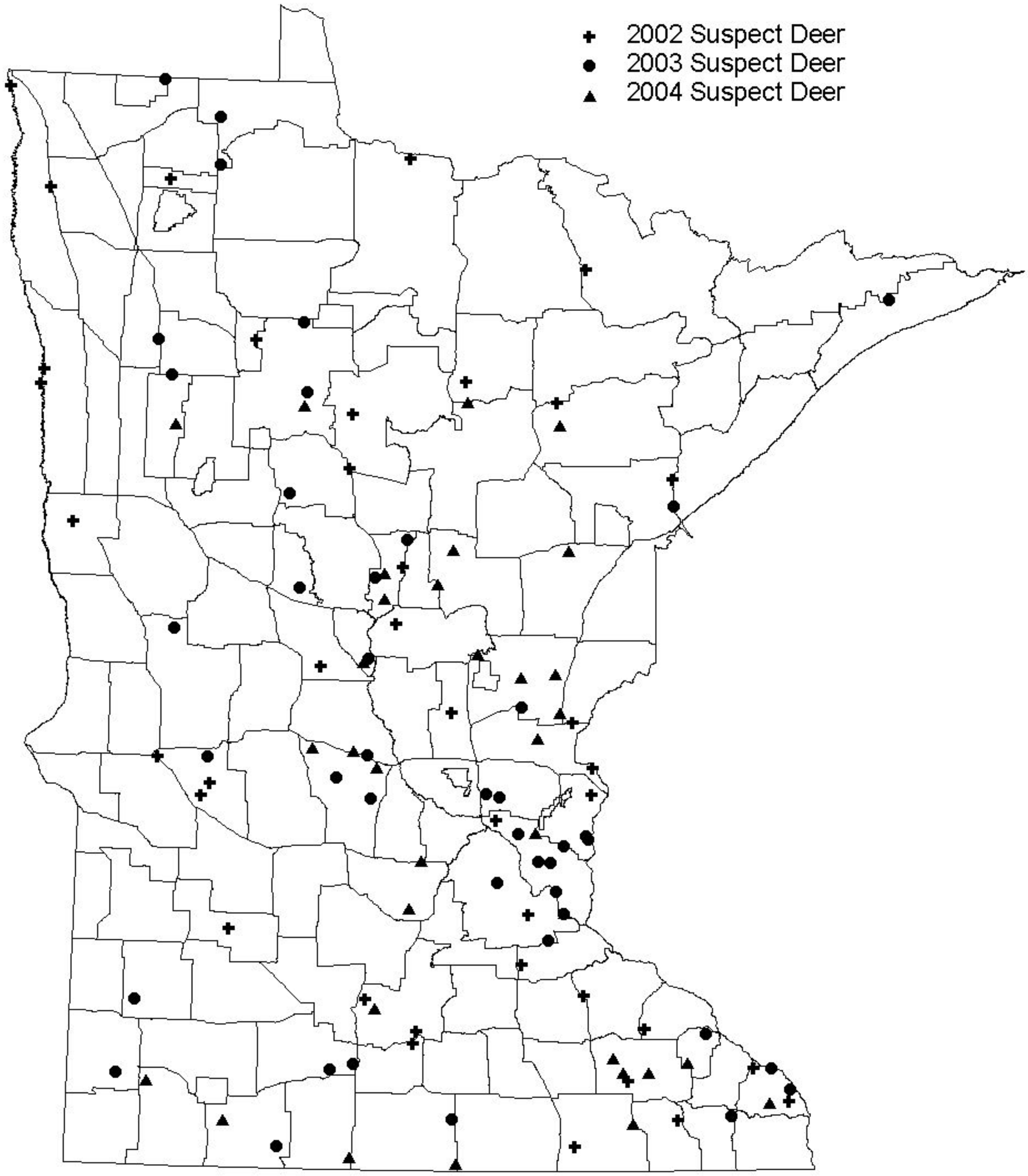


Figure 3. Locations of suspect deer sampled for chronic wasting disease in Deer Permit Areas of Minnesota, 2002–2004.

SURVIVAL AND CAUSE-SPECIFIC MORTALITY OF A PROTECTED POPULATION OF RIVER OTTERS IN SOUTHEASTERN MINNESOTA*

Thomas A. Gorman[1], Brock R. McMillan[1], and John D. Erb

Abstract: Determining causes of mortality and estimating survival rates can provide insight into the status of a species whose population trends are not well understood. Legal harvest of river otters (*Lontra canadensis*) has been prohibited in southern Minnesota since 1917. Thus, this region provided an opportunity to examine the influences of incidental trapping and natural causes of mortality on a protected population of river otters. From October 2001 through April 2004, 39 (13 adult males; 6 sub-adult males; 8 adult females; 12 sub-adult females) river otters were captured and radio-marked along a portion of the Mississippi River watershed to estimate survival and determine causes of mortality. For each mortality event, we determined the cause of death (e.g., incidental captures from trappers, automobile collisions, and natural mortality). To assess which factors were most influential on survival, we developed a suite of a priori models incorporating age (i.e., sub-adults < 2 years old and adults > 2 years old), sex, and/or season. Program MARK was used for model selection and survival estimation and we estimated population growth using a Leslie projection matrix. Human induced mortalities, including accidental captures by fur-harvesters targeting other species (n = 6) and automobile collisions (n = 1), accounted for the majority of deaths, while natural mortality was low (n = 1). Annual survival of adult females (S = 0.733, SE = 0.122) was similar to survival of sub-adult females (S = 0.709, SE = 0.132), but survival of adult males (S = 0.889, SE = 0.086) and sub-adult males (S = 0.891, SE = 0.088) was higher than females. The population was estimated to be increasing (λ = 1.146) despite females having a lower overall survival rate than males. A perturbation analysis of the matrix indicated that the survival of juvenile and adult females has the greatest influence on λ. River otters and other furbearers need to be monitored to assess population status, and measures should be taken to ensure that demographic parameters are sufficient for the population to persist.

** From the Abstract of Chapter 1 of Thomas A. Gorman's Master's Thesis, Biology Dept., Minnesota State University, Mankato.*

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HOME RANGE AND SPATIAL CHARACTERISTICS OF RIVER OTTERS IN SOUTHEASTERN MINNESOTA*

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***Abstract*:** The river otter (*Lontra canadensis*) has been re-occupying portions of its native range across North America for more than 25 years due to extensive reintroduction programs, improved water quality, and wetland restoration. In southeastern Minnesota, there is a native population of river otters that is believed to be increasing in distribution and numbers. We examined the spatial characteristics, overlap, and interactions between river otters in southeastern Minnesota. We captured 39 river otters (13 adult males; 6 sub-adult males; 8 adult females; 12 sub-adult females) and equipped them with radio transmitter implants. Otters were monitored from spring 2002 to spring 2004 along portions of the Mississippi River, the Whitewater River, and the Zumbro River. We estimated annual and seasonal home ranges and annual core areas for individual otters, and compared home range characteristics between sexes and among age classes. Further, we evaluated the static and dynamic interactions between individuals to evaluate the social structure of the population. Annual home ranges of male river otters were 2.73 times larger than females, and annual core areas of males were 2.52 times larger than females. Within each sex, we did not detect a difference in home range size between seasons, but there was a seasonal difference between sexes ($F = 14.419$; $p = 0.0003$). The static interactions (home range overlap) between river otters were extensive, and occurred between 94.9% of the individuals analyzed at the 95% home range scale, and occurred between 69.2% of the individuals analyzed, at the 50% core area scale. Dynamic interactions between male: female comparisons and female: female comparisons were positive (78.5% and 75.5% were positive, respectively), revealing that one animal's movements influenced the others. River otter sociality and space use varied between the sexes, with minimal influence of age and season.

** From the Abstract of Chapter 2 of Thomas A. Gorman's Master's Thesis, Biology Dept., Minnesota State University, Mankato.*

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NATAL DEN SITE CHARACTERISTICS OF RIVER OTTERS IN SOUTHEASTERN MINNESOTA*

Thomas A. Gorman[1], Brock R. McMillan[1], and John D. Erb

Abstract: Several factors may influence the selection of natal dens by female river otters (*Lontra canadensis*). Den use may be influenced by the availability of dens sites as well as specific den characteristics that protect young from other river otters and weather extremes. Otters have been reported to use a variety of existing burrows created by other animals. We monitored 8 adult (>2 years old) female river otters during the natal denning season (March – May). We measured 12 micro- and macro-habitat characteristics to best describe the physical characteristics of their natal dens. Females began to actively den on 27 March, with a mean denning date of 31 March, and were in natal dens for an average of 49 days (SE = 3.03). Two females used natal dens that consisted of brush piles, 4 females located dens in caves that may have been improved by other animals, but otherwise were natural features that occurred in the limestone bluffs, 1 female used a den that was created by the roots of a big- toothed aspen (Populus grandidentata), and 1 female placed her den in a beaver bank den. Dens were located an average of 315.9 m (SE = 78.5, n = 9) from the nearest body of water and were on average at 274.1 m (SE = 15.7m, n = 9) of elevation above sea level. All females used natal dens that were protected from rapid changes in water levels, and 7 of 8 females placed dens outside of their normal activity areas. Management for river otters should not only consider habitats within normal areas of activity, but also adjacent uplands likely to be used for natal denning habitat.

** From the Abstract of Chapter 3 of Thomas A. Gorman's Master's Thesis, Biology Dept., Minnesota State University, Mankato.*

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IDENTIFYING PLOTS FOR SURVEYS OF PRAIRIE CHICKENS IN MINNESOTA

Michael A. Larson

SUMMARY OF FINDINGS

Collection of data for this project will begin on 4 April 2005, so no results are available.

INTRODUCTION

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

The availability of 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. A study design that selects all samples randomly or at least allows for concurrently estimating the proportions of locations that are and are not occupied by leks, as I propose below, would preclude those problems (Keating and Cherry 2004).

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 will build 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 will also serve 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.
- To evaluate potential within-year sources of variation in the probability of detecting prairie-chicken leks in Minnesota.

METHODS

In Minnesota, prairie chickens occur in 3 distinct ranges (i.e., Northwest, Southwest, and Central; Giudice 2004; Figure 1). The study area will be established in the Northwest prairie chicken range because the Northwest range contains the largest population of prairie chickens, is where the hunting permit areas are, and is the focus of all current prairie chicken monitoring effort by the DNR. The Northwest prairie chicken range is linear, and most known lek locations are between southwest Red Lake County in the north and eastcentral Wilkin County in the south (Figure 1). The study area includes the northern 96% of the Northwest range as defined by Giudice (2004) based upon land type associations of the Ecological Classification System. The size of the study area was limited by a maximum distance of 90 km from Crookston and Moorhead, where field technicians reside. All study areas will encompass many large areas of open grasslands that could serve as suitable habitat for prairie grouse leks.

Methods for this study are based on recently developed analytical techniques for estimating the probability of site occupancy (MacKenzie et al. 2002). Multiple visits to sample plots, only a portion of which will be occupied by the object of interest, are the basis of such studies. The main benefit of this method for estimating the probability of site occupancy is that the models used to analyze the data simultaneously account for covariates of site occupancy and the fact that the probability of detecting the object of interest is <1. Although logistic regression may seem simpler and more straightforward than occupancy modeling, it does not account for errors in the response variable (occupied or unoccupied) due to detection probabilities <1, and it is not necessary because occupancy models incorporate the same logistic function to relate covariates to the probabilities of detection and occupancy.

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).

A sampling unit, or plot, will be defined as a Public Land Survey (PLS) section, most of which are 1.6×1.6 km squares (i.e., 2.59 km^2 = 1 mi^2). In portions of the prairie chicken range in Minnesota some PLS sections are more rectangular and much smaller than 2.59 km^2. Variability in the size of plots is accounted for by the possible inclusion of habitat area within a plot as a covariate for ψ. The spatial scale of sampling units is a trade-off between being relatively large, so a reasonable proportion of them can be surveyed each year and ψ is "large" (Figure 1), and being relatively small, so that each unit can be surveyed rapidly and is likely to contain $\leq$1 lek (few PLS sections contain >1 lek [DNR, unpublished data]).

Access to and within plots by automobiles may be limited or infeasible in some areas. Time constraints will prevent extensive surveying by foot, so failing to detect a lek within a plot, even after multiple visits and accounting for detection probabilities <1, will not ensure that the plot is not occupied. That will cause the estimated probability of occupancy to be biased relative to true occupancy throughout the study area. Inferences, therefore, will be limited to portions of each study area that are within some distance of roads that are accessible by automobiles during spring. The distance will be equal to the maximum distance at which leks may be detected by sight or sound.

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). Dual frame sampling is appropriate for this situation because an area frame is necessary for sample plots to be representative of other plots in the population, and a list frame is useful for focusing adequate sampling effort in plots where leks are 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. Dual frame sampling is essentially a form of stratification because parameter estimates and variances are weighted by the size of each frame and then added across frames (Haines and Pollock 1998), just as they are for stratified random samples.

An observer will visit each sample plot once during each of T = 3 consecutive biweekly periods from 4 April until 15 May 2005 (Svedarsky 1983). A visit will consist of a 20-minute interval between 0.5 hours before until 2 hours after sunrise (Cartwright 2000) during which a plot is surveyed with the purpose of detecting the presence of a lek (i.e., ≥2 male prairie chickens) by sight or sound. During each visit the observer will record whether or not a lek was detected, the value of time-dependent covariates that may affect p, and, if time is still available, the value of covariates of p and ψ that vary only spatially. Observers will also compare printed maps of land cover with actual land cover in and near sample plots (i.e., within 1.6 km) and mark corrections, including whether or not roads are paved, on the maps.

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 the following 2 steps also will be taken to prevent the sampling design from introducing heterogeneity during this study. First, each observer will visit a different set of plots during each biweekly survey period, so differences among observers in their ability to detect leks will not be correlated with specific plots. Second, 1 visit to each plot will occur during each of 3 periods of the morning—early (-30–20 minutes from sunrise), middle (21–70 minutes after sunrise), and late (71–120 minutes after sunrise). This will minimize the correlation between plot-specific p's and differences in detection rates caused by time of day.

The probability of detecting a lek, p, may be affected by many different covariates (Table 1). The value of some time-dependent covariates of p will be recorded during each visit, whereas the value of other covariates that vary only spatially will be recorded only once for each plot.

The probability that a plot is occupied by a lek, ψ, also may be affected by many different covariates (Table 2). Prairie chickens may respond to landscape characteristics at several spatial scales (Niemuth 2000), so each covariate will be quantified at 2 different spatial scales (Keppie and Kierstead 2003)—the 2.59-km^2 sampling plot and a 9-plot area (≤23.3 km^2) centered on the sample plot. The sampling plot roughly corresponds to home range sizes of prairie chickens (<400 ha; Robel et al. 1970) during spring. The larger scale roughly corresponds to areas of nesting and brood-rearing, which usually occur within 1.6 km from a lek (Schroeder and Braun 1992, Ryan et al. 1998). Most of the covariates of ψ will be measured using a GIS, but some will need to be verified by observers in the field.

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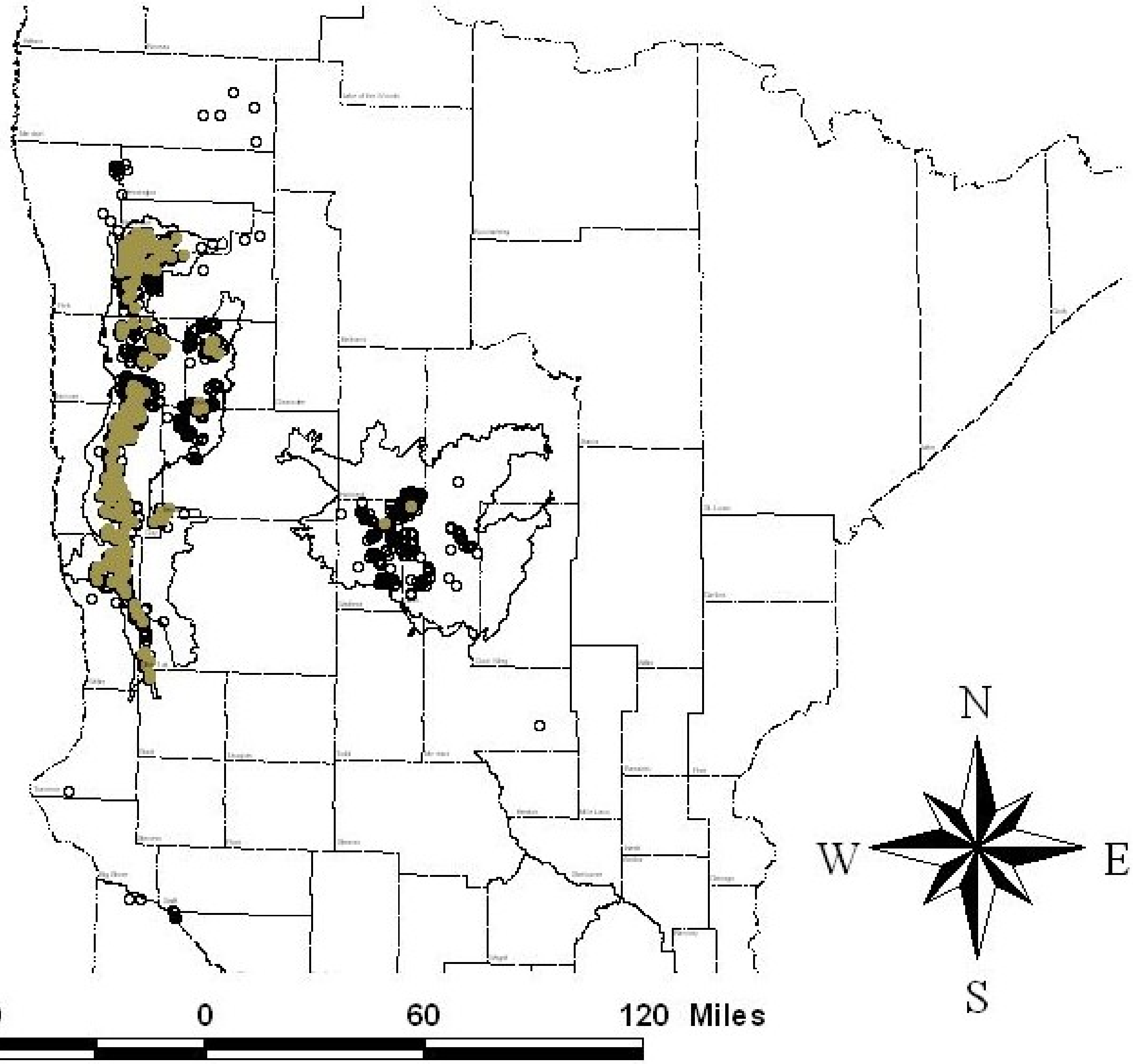

Figure 1. Locations of recent (gray circles) and historical (open circles) leks of greater prairie-chickens in Minnesota. Boundaries of the Northwest and Central ranges are based on land type associations of the Ecological Classification System. The Southwest range includes areas near the Minnesota River and in Big Stone County.

Table 1. Covariates that may affect the probability of detecting a lek, given that a lek was present within a sample plot (p).

Type[a]	Abbreviation	Description
t	OBS	observer
t	RECAP	whether or not a lek was detected previously this year
t	BIRDS	number of prairie grouse observed
s	LEKS	number of leks observed
t	DAY	day of the survey
t	TIME	minutes before or after sunrise
t	TEMP	ambient temperature
t	WIND	wind velocity
t	PREC	presence or absence of precipitation; type and intensity recorded also
t	CLOUD	proportion of the sky obscured by clouds
s	ROAD	density of roads accessible to a vehicle (km/km^2)
s	RDINT	density of accessible roads to the interior of the plot only
s	VIS	proportion of suitable land cover types that is visible to the observer
t	COV	proportion of suitable cover types under snow cover or water

[a] Time-dependent covariates (t) will be quantified during each visit to a plot, whereas covariates that vary only spatially (s) during the study will be quantified only once for each plot.

Table 2. Covariates that may affect the probability of a plot being occupied by a lek (ψ).

Abbreviation	Description
HABAREA	area (ha) of all suitable land cover types combined
PROTECT	area of land in a conservation program or owned by a conservation organization
PRAIRIE	proportion of area in the Prairie cover type
GRASS	proportion of area in the Grassland cover type
SEDGE	proportion of area in the Sedge Meadow cover type
CROP	proportion of area in the Cropland cover type
SHRUB	proportion of area in the Lowland Deciduous Shrub & Upland Shrub cover types
BOGCON	proportion of area in the Stagnant Black Spruce & Stagnant Tamarack cover types
FOREST	proportion of area in the Forestland cover type
EDGE	density of edges (m/ha) between suitable and unsuitable cover types
LEKDIST	distance (m) to the nearest known lek, measured between plot centers
HOMES	number of occupied human residences (v)
TREE	presence of trees within suitable cover types (v)
ROAD	density of all roads (km/km^2) as an indicator of disturbance (v)[a]
PAVE	density of paved roads (v)
DISTURB	evidence of disturbance (e.g., prescribed burning, livestock grazing) (v)

[a] The last 5 covariates will be verified (v) by observers in the field.

LINKING POPULATION VIABILITY, HABITAT SUITABILITY, AND LANDSCAPE SIMULATION MODELS FOR CONSERVATION PLANNING

Michael A. Larson, Frank R. Thompson, III[1], Joshua J. Millspaugh[2], William D. Dijak[1], Stephen R. Shifley[1]

Abstract: Methods for habitat modeling based on landscape simulations and population viability modeling based on habitat quality are well developed, but no published study of which we are aware has effectively joined them in a single, comprehensive analysis. We demonstrate the application of a population viability model for ovenbirds (Seiurus aurocapillus) that is linked to realistic landscape simulations using a GIS-based habitat suitability index (HSI) model. We simulated potential future characteristics of a hardwood forest in southern Missouri under 2 tree harvest scenarios using LANDIS. We applied 3 different versions of the HSI model (lower, best, and upper estimates) to output from the landscape simulations and used RAMAS GIS to link estimates of temporally dynamic habitat suitability, through fecundity and carrying capacity, to ovenbird population viability. Abundances and viability differed more between the upper and lower HSI estimates than between the 2 forest management scenarios. The viability model was as sensitive to the relationship between reproductive success and habitat suitability as it was to rates of first-year survival and reproductive success itself. Habitat-based viability models and the wildlife studies they support, therefore, would benefit greatly from improving the accuracy and precision of habitat suitability estimates.

Combining landscape, habitat, and viability models in a single analysis provides benefits beyond those of the individual modeling stages. A comprehensive modeling approach encompasses all components and processes of interest, allows direct comparison of the relative levels of uncertainty in each stage of modeling, and allows analysis of the economic benefits and costs of different land use plans, which may be affected by landscape management, habitat manipulation, and wildlife conservation efforts. Using population viability, habitat suitability, and landscape simulation models in an integrated analysis for conservation planning is an important advancement because habitat quality is a critical link between human land use decisions and wildlife population viability.

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COMPARABILITY OF THREE ANALYTICAL TECHNIQUES TO ASSESS JOINT SPACE USE

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Abstract: The degree of space-use overlap among adjacent individuals is a central focus of many wildlife investigations. We studied the comparability of minimum convex polygon and fixed-kernel home-range overlap indices and Volume of Intersection (VI) scores using simulated data. We simulated pairs of point patterns to represent telemetry locations of adjacent individuals and varied the amount of potential overlap in the simulation region (100%, 50%, and 10%) and the point distribution (random, loosely clumped, and tightly clumped). We created 1,000 pairs of point sets (60 points in each individual set) for each of the 9 potential overlap and point distribution combinations. In all 9 treatment combinations, VI scores were highest followed by kernel and then polygon estimates. Raw differences among estimates within a treatment were greatest when there was 50% potential overlap, and overlap indices decreased as the degree of clumping increased. The relative differences among overlap indices within a treatment were affected most by potential overlap; differences generally were greatest at 10% and least at 100%. Correlation between index values was lowest for random point patterns, and highest for loosely clumped and tightly clumped point patterns. Although the VI tended to indicate the most overlap and minimum convex polygon the least, there was no consistent correction factor among techniques because of the interacting effects of the overlap index, distribution pattern, and potential overlap. Interpretation of overlap measures requires careful consideration of assumptions and properties of animals under study.

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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 (54 bulls and 60 cows) have been captured and collared since beginning the study in 2002. As of 31 March 2005, 36 collared moose (20 bulls and 16 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 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 have stabilized around 4,000 animals.

That moose numbers in northeast Minnesota have not increased in recent years is an enigma. Research in Alaska and northern Canada has indicated that 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: i) average non-hunting mortality rate for moose in northeastern Minnesota is considerably higher and/or more variable than measured in previous studies; ii) recruitment rates estimated from the aerial surveys and used in the model are biased high; and/or iii) moose numbers estimated by the aerial survey are biased low.

OBJECTIVES

- Determine annual rates of non-hunting mortality for northeastern moose;
- 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 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 to77 radio-collared moose. The transmitter in each radio-collar contained a mortality sensor that increased the pulse rate (mortality mode) if it remained stationary for more than 6 hours. When a transmitter was

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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 progesterone levels (Haigh et al. 1981). Following birth, the presence/absence of a calf with a radio-collared cow was determined, when possible during the telemetry flights.

During the aerial moose survey in January 2005, a sightability model (Anderson and Lindzey 1996, Quayle et al. 2001) was developed using the radio-collared moose. Following each relocation flight, a square test plot (6.7 mi^2) that surrounded one or more collared moose 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

During 7-11 February 2005, 30 adult moose (20 bulls and 10 cows) were immobilized to increase the sample of collared moose. A total of 114 moose (60 cows and 54 bulls) have been captured in northeastern Minnesota since February 2002 (Figure 1). No additional moose will be collared.

As of 31 March 2005, 36 radio-collared moose (20 bulls and 16 cows) have died. The cause of death in 16 cases could be identified (8 hunter kill, 1 poached, 1 train, 3 trucks, 2 wolf predation, and 1 natural accident). 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 remains of 2 additional moose that died within BWCAW. Fifteen appear to have died from unknown non-traumatic causes. In 8 cases, scavengers had consumed the carcasses, but evidence suggested predators did not kill them. In the remaining 7 cases, moose had little or no body fat (rump, kidney, abdominal, or heart), and were often emaciated. Moose dying of unknown causes died throughout year (1 - January, 2 - April, 4 - May, 1 - June, 1 - July, 3 - August, 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, 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).

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 18 additional radio-collared moose were tested for the presence of *P. tenuis*-specific antibodies using an enzyme-linked immunosorbent assay procedure (ELISA) (Ogunremi et al. 1999). Thirteen (11 cows and 2 bulls) of the 79 collared moose tested to date were sero-positive for antibodies against P. tenuis. Subsequently 3 died of unknown causes and 1 was killed by a hunter.

Pregnancy rate was estimated at 92% in 2002, 57% in 2003, and 100% in 2004, based on serum progesterone. The samples from 2005 have not been analyzed yet. Similar estimates for the northwest moose population between 1996 and 1999 averaged 48% (Cox et al. *In press*).

Radio collared moose were located 41 times in the process of developing a sightability model. In 21 cases, the collared moose was observed using the standard survey protocol. In 17 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, grouped into 6 equal intervals (Giudice and Fieberg, unpublished). Total population size based on this sightability model was 6,481±26%, higher than previous estimates calculated using the "Gasaway" protocol (Gasaway et al. 1986) and likely more accurate (Lenarz, unpublished). Ultimately, with additional data, this model will improve the accuracy and precision of the aerial survey.

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

Non-Hunting Mortality			
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)
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)

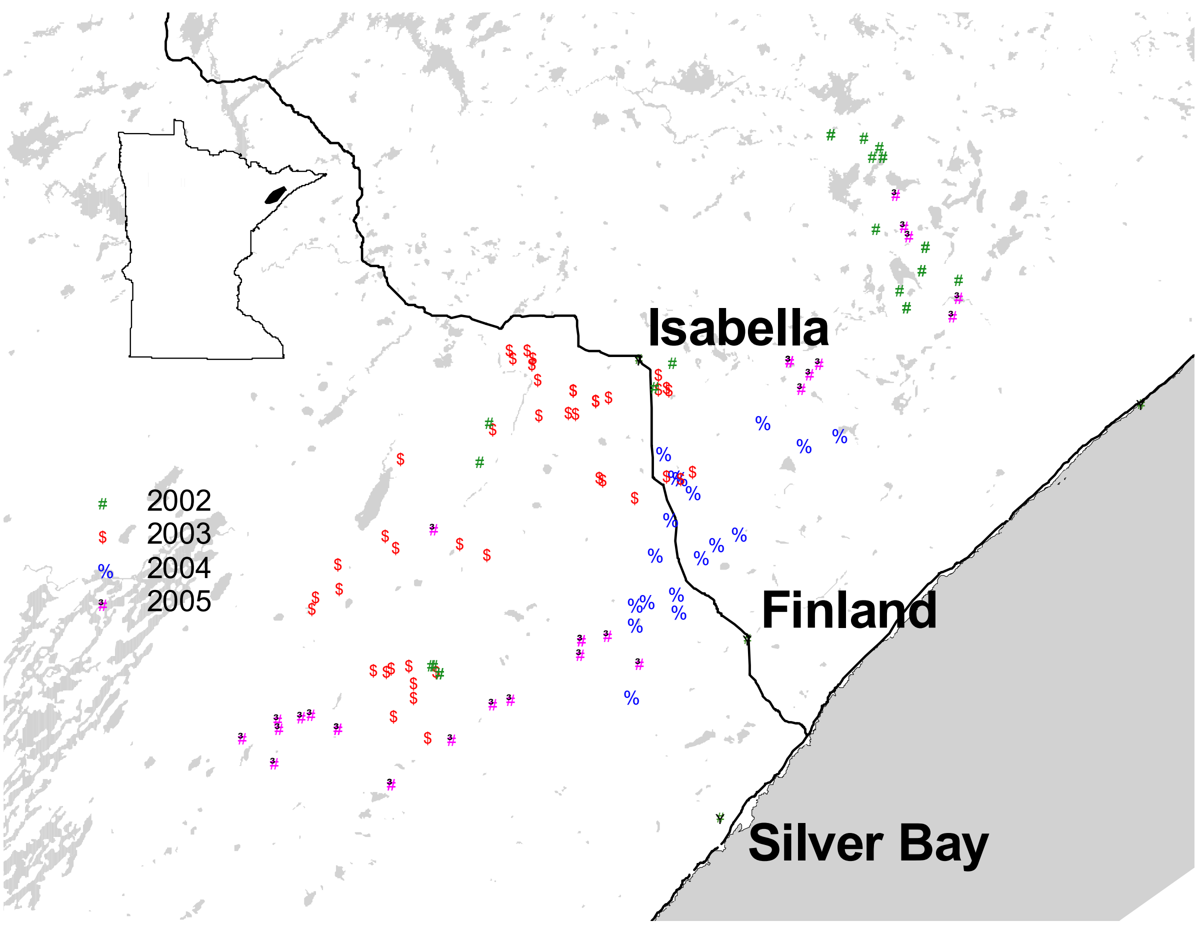

Figure 1. Capture locations of moose radio collared, 2002-2004.

CONDITION OF MOOSE *(ALCES ALCES)* IN NORTHEASTERN MINNESOTA

Glenn D. DelGiudice, Mark S. Lenarz, Michael Schrage[1], Andrew Edwards[2] Michael E. Nelson[3], and Barry A. Sampson

SUMMARY OF FINDINGS

During winters 2002–03, 2003–04, and 2004–05, body condition of 37, 17, and 26 free-ranging moose (*Alces alces*), respectively, was assessed by at least 1 of the following: (1) ultrasonic measurement of rump fat thickness, (2) Franzmann's condition classification (FCC), or (3) visual and palpation assessment of fat repleteness of the rump (BCS_r). Mean maximum rumps fat (Maxfat) thickness was 1.6 (SE = 0.16), 2.1 (SE = 0.38), and 2.2 (SE = 0.18) cm during these 3 winters, whereas mean ingesta-free body fat (IFBFAT) estimates were 8.9% (range = < 5.6–13.4%), 9.9% (range = 6.8–15.0%), and 10.1% (range = 6.5–14.2%). Maxfat and IFBFAT were less in bulls than in cows during winters 2003–04 and 2004–05. Mean FCC and BCS_r scores were 7.2 (scale of 10) and 3.4 (scale of 5), 7.3 and 3.8, and 6.5 and 3.2, respectively, during winters 2002–03 to 2004–05. Both scores tended to be lower in bulls than in cows during the first 2 winters, and were significantly ($P \leq 0.05$) lower during winter 2004–05. There were significant correlations between the FCC and BCS_r for all moose during all 3 winters (r = 0.83, 0.75, and 0.61; $P \leq 0.002$). Additionally, Maxfat was correlated to FCC scores (r = 0.56, 0.71, and 0.56; $P \leq$ 0.01) and BCS_r scores (r = 0.53, 0.68, 0.71; $P \leq 0.02$).

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INTRODUCTION

A study of moose in northeastern Minnesota was begun in 2002, because aerial survey estimates suggested the population was stable despite a very conservative harvest (Lenarz et al., unpublished data). The study's goal is to generate data that will provide a clearer understanding of the ecological mechanism(s) underlying the population dynamics observed (Lenarz et al., unpublished data). One of the primary objectives was to "determine annual rates of non-hunting mortality…" for moose in this part of the state (Lenarz et al., unpublished data). Winter is the most nutritionally challenging season of the year for northern cervids, and nutrition has been shown to be a mechanistic link between environmental variation (e.g., winter tick [*Dermacentor albipictus*] infestation) and variation of moose populations (DelGiudice et al. 1997).

OBJECTIVES

- Assess winter condition of moose recruited into the study;
- Relate condition to winter severity; and
- Relate condition to sex and age.

METHODS

Logistical constraints and considerations associated with capture and handling of free-ranging moose during the study's first winter field season (2001–02) precluded condition assessments; however, such evaluations during capture operations of winter's 2002–03, 2003–04, and 2004–05 were feasible and successful.

During 26 February–2 March 2003, 9–11 February 2004, and 7–10 February 2005 adult (≥1.5 year old) moose were immobilized with a carfentanil-xylazine combination delivered by a dart rifle from a helicopter. Details of the capture/chemical immobilization procedure, as well as a description of the study area, are provided elsewhere (Lenarz et al., unpublished data).

Condition of moose was assessed by the following 3 methods: (1) ultrasonic measurements of rump fat thickness (Stephenson et al. 1998, 2002), (2) Franzmann's condition classification (FCC), developed specifically for moose (Franzmann 1977), and (3) the portion of a body condition scoring system developed for elk (*Cervus elaphus*), which concentrates on visual and palpation assessments of fat repleteness of the rump (BCS_r, Cook et al. 2001). We measured subcutaneous rump fat thickness (cm) with a portable ultrasound device (Sonovet 600 model, Universal Medical Systems, Inc., Bedford Hills, N.Y.1) and a 5-MHz 8-cm linear-array transducer. Measurements were made at the midway point ("Mid") between the tips of the iliums and the right or left tuber ischium (pin bone), and at the point of maximum fat thickness ("Maxfat"), which we located by scanning laterally along the sacral ridge towards the pin bone. Location of Maxfat was immediately anterior to the cranial process of the pin bone. Due to differences in body size of males and females, application of a scaling factor (0.83) to Maxfat measurements of males permitted comparison to adult females (Stephenson et al. 1998). The FCC and the BCS_r are described in Tables 1 and 2. Compared to the BCS_r, the FCC system includes a more complete assessment of the conformation of the moose's entire body related to condition. Captured moose were aged in the laboratory by counting of cementum annuli on the last incisor (extracted during capture). Age determinations of moose captured during winter 2004–05 had not been made prior to the writing of this summary.

RESULTS AND DISCUSSION

By at least 1 of the 3 methods, we assessed the condition of 37 (19 females, 18 males) of the 42 adult moose captured and handled during winter 2002–03, 17 (12 females, 5 males) of 18 moose in winter 2003–04, and 26 (8 females, 18 males) of 30 moose in winter 2004–05. Overall, mean Maxfat was 1.6 (SE = 0.16, range = 0–3.8 cm), 2.1 cm (SE = 0.38, range = 0.58–4.6 cm), and 2.2 cm (SE = 0.18, range = 0.42–4.2 cm) during these 3 winters. In captive moose, Maxfat measurements have ranged between 0 and 7.0 cm, and were directly related ($Y = 5.61 + 2.05\ x$, $r^2 = 0.96$, $P < 0.0001$) to ingesta-free body fat (IFBFAT) contents of approximately 2.5–17.5% (Stephenson et al. 1998). Applying the regression of Stephenson et al. (1998), Maxfat measurements of our free-ranging moose indicated an estimated mean IFBFAT of about 8.9% (range of < 5.6–13.4%), 9.9% (range = 6.8–15.0%), and 10.1% (6.5–14.2%) during winters 2002–03, 2003–04, and 2004–05, respectively. Studies of captive moose (and other cervids) have shown that at 5–5.6% IFBFAT, rump fat will be depleted (i.e., Maxfat = 0 cm). Maxfat and IFBFAT were less in bulls than in cows during all 3 winters with the difference significant ($P \leq 0.05$) in winter 2003–04 and 2004–05 (Table 3).

The mean FCC and BCS_r scores were 7.2 (range = 3–10, scale of 10) and 3.4 (range = 2–4.5, scale of 5) in winter 2002–03, 7.3 (range = 4–9) and 3.8 (range = 2.5–5.0) in winter 2003–04, and 6.5 (range = 4–8) and 3.2 (range = 2–4.5) in winter 2004–05. According to both of these scoring systems, although not significantly, mean condition scores were apparently lower for bulls than cows during winters 2002–03 and 2003–04 (Table 3). However, during winter 2004–05, the FCC and BCS_r scores were significantly lower ($P \leq 0.05$) in bulls than

[1] disclaimer

in cows (Table 3). There was no difference in age of moose between winters 2002–03 and 2003–04, and we observed no relation between moose condition, as assessed by FCC, BCS_r, Maxfat or estimated IFBFAT, and moose age during winters 2002–03 and 2003–04. There were significant correlations between the FCC and BCS_r scores for all moose during winters 2002–03 (r = 0.83, P< 0.0001), 2003–04 (r = 0.75, P = 0.002), and 2004–05 (r = 0.61, P< 0.0001). Additionally, during all 3 winters, Maxfat was significantly correlated to FCC scores (r = 0.56, 0.71, and 0.56; $P \leq$ 0.01) and BCS_r scores (r = 0.53, 0.68, 0.71; $P \leq$ 0.02). The strength of the statistical relationship between the scoring systems and Maxfat measurements is inherently limited, because the scoring systems are characterized by discrete scores, whereas the Maxfat measurements are continuous. Consequently, a range of Maxfat measurements may be associated with a given condition score.

The late winter, mean Maxfat measurements (2002–03, 1.6 cm and 95% confidence limits [CL] = 1.3, 1.9 cm; 2003–04, 2.1 cm and 95% CL = 1.4, 2.9 cm; 2.2 cm and 95% CL = 1.8, 2.6) and associated estimated IFBFAT contents (roughly 9–10%) of our free-ranging moose indicate that most of them were in good condition, which was consistent with the unusually mild (2002–03) and moderate (2003–04 and 2004–05) weather conditions of these winters in northeastern Minnesota. It is noteworthy that snow depths were only 30–36 cm until mid-January 2004, but by early February when we conducted moose capture operations, snow depths were typically approaching 80 cm. Similarly, during early February 2005, snow depths were 74–91 cm; however, when assessing moose condition in late February–early March 2003, snow cover was only 23–33 cm. Clearly, the preponderance of bulls in the capture sample of 2005 lead to the overall increase in the animals assessed to be in fair-poor condition (46.2%) compared to moose assessed in 2003 and 2004 (24.3–25.0%, Table 4). Breeding bulls particularly are entering winter in poorer condition than females due to their diminished consumption of food during the fall rut (Schwartz and Renecker 1997).

The potential value of the condition assessments of the radiocollared moose may occur at the individual and population scales. They may provide insight relative to the survival or fate (i.e., cause of mortality) of each individual moose. There was a tendency for moose that died of non-hunting causes to be younger and exhibit lower winter condition scores than moose that survived; however, overall, we noted no significant differences. This may be attributable in part to a relatively small sample size of such mortalities.

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Table 1. Franzmann's condition classification for moose, used to assess winter condition of free-ranging adult moose during winters 2002–03 (19 females, 18 males), 2003–04 (11 females, 5 males), and 2004–05 (8 females, 18 males), northeastern Minnesota.

10	A prime, fat animal with thick, firm rump fat by sight. Well fleshed over back and loin. Shoulders round and full
9	A choice, fat moose with evidence of rump fat by feel. Fleshed over back and loin. Shoulders round and full.
8	A good, fat moose with slight evidence of rump fat by feel. Bony structures of back and loin not prominent. Shoulders well fleshed.
7	An average moose with no evidence of rump fat, but well fleshed. Bony structures of back and loin evident by feel. Shoulders with some angularity.
6	A moderately fleshed moose beginning to demonstrate **one** of the following conditions: (A) definition of neck from shoulders; (B) upper foreleg (humerous and musculature) distinct from chest; or (C) rib cage prominent.
5	A condition in which **two** of the characteristics listed in Class 6 are evident.
4	A condition in which **three** of the characteristics listed in Class 6 are evident.
3	A condition in which the hide fits loosely about neck and shoulders. Head is carried at a lower profile. Walking and running postures appear normal.
2	Signs of malnutrition are obvious. The outline of the scapula is evident. Head and neck are low and extended. The moose walks normally but trots and paces with difficulty, and cannot canter.
1	A point of no return. A generalized appearance of weakness. The moose walks with difficulty and can no longer trot, pace or canter.
0	Dead

Table 2. Body condition scoring system modified from Cook et al. (2001), used to assess the condition of free-ranging adult moose during winters 2002–03 (19 females, 18 males), 2003–04 (10 females, 4 males), and 2004–05 (8 females, 18 males), northeastern Minnesota.

5	Sacral ridge, ilium, ischium are virtually discernible.
4	Sacral ridge is discernible from ilium approximately midway to base of tail, Ischium and sacro-sciatic ligament are discernible.
3	Entire sacral ridge is discernible, but not prominent.
2	Sacral ridge is prominent to base of tail.
1	Sacral ridge, ilium, ischium, tuber coxae, and sacro-sciatic ligament (entire top of rump) are prominent.

Table 3. Mean (±SE) maximum rump fat (Maxfat) thickness measured by portable ultrasonography, and body condition scores (Franzmann's condition classification [FCC] and rump portion of body condition scoring system [BCS_r] modified from Cook et al. 2001) of free-ranging adult moose during winters 2002–03 (19 females, 18 males), 2003–04 (11 females, 5 males), and 2004–05 (8 females, 18 males), northeastern Minnesota.[a] Range of values occurs in parentheses.

Sex	Maxfat (cm)		FCC		BCS_r	
	Mean	**SE**	**Mean**	**SE**	**Mean**	**SE**
Winter 2002–03						
Females	1.7[b]	0.24	7.4	0.4	3.6	0.2
	(0.0–3.8)		(3.0–10.0)		(2.0–4.5)	
Males	1.5[b]	0.20	7.0	0.3	3.2	0.1
	(0.3–2.6)		(4.0–9.0)		(2.0–4.3)	
Winter 2003–04						
Females	2.9[c]	0.42	7.8	0.3	4.1[d]	0.2
	(1.5–4.6)		(5.0–9.0)		(3.0–5.0)	
Males	1.1[c]	0.38	6.2	0.8	3.1[d]	0.3
	(0.6–2.6)		(4.0–8.0)		(2.5–4.0)	
Winter 2004–05						
Females	2.9	0.24	7.4	0.3	3.8	0.2
	(2.0–4.2)		(6.0–8.0)		(3.0–4.5)	
Males	1.8[e]	0.18	6.1	0.3	2.9	0.1
	(0.4–2.8)		(4.0–7.5)		(2.0–3.5)	

[a] Descriptions of the FCC and BCS_r systems are provided in Tables 1 and 2, respectively.
[b] *n* = 16 for females and males due to temporary malfunctioning of portable ultrasound.
[c] *n* = 7 and 5 for females and males, respectively, due to unavailability of portable ultrasound.
[d] *n* = 10 and 4 for females and males, respectively; assessor did not have access to moose.
[e] *n* = 15 due to unavailability of portable ultrasound.

Table 4. Qualitative condition assessment according to Franzmann's condition classification of free-ranging adult moose during winters 2002–03 (19 females, 18 males), 2003–04 (11 females, 5 males), and 2004–05 (8 females, 18 males), northeastern Minnesota.

Franzmann's Condition Score				
	≥8	**7 ≤ x < 8**	**< 7**	Total
	(Very Good)	**(Good)**	**(Fair-Poor)**	
Winter 2002-03				
Number of moose	15	13	9	37
Percent of total	40.54	35.14	24.32	100
Winter 2003-04				
Number of moose	9	3	4	16
Percent of total	56.25	18.75	25.00	100
Winter 2004-05				
Number of moose	4	10	12	26
Percent of total	15.38	38.46	46.15	100

[a] A description of Franzmann's condition classification is provided in Table 1.

PARASITE-MEDIATED DECLINE IN A MOOSE POPULATION AT THE SOUTHERN RANGE PERIPHERY

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Abstract: Several potential proximate causes may be implicated in a recent (1984-2001) decline in moose (*Alces alces andersoni*) numbers from their southern range periphery in northwest Minnesota, including increased: i) predation by wolves (*Canis lupus*) and black bears (*Ursus americanus*), ii) mortality from legal or illegal hunting, iii) malnutrition due to high intraspecific competition, iv) malnutrition from increased food competition with white-tailed deer (*Odocoileus virginianus*), v) deleterious effects of parasites and diseases, some of which are associated with deer, and vi) negative effects of climate change on survival and production. Ultimate causes potentially contributing to the moose decline include factors associated with marginal habitat (leading to malnutrition, immunosuppression, etc.). We examined survival among radiocollared (*n* = 152) adult cow and juvenile moose in 3 northwest Minnesota study areas during 1995–2000. We assessed cause of death and pathology through carcass necropsy of radioed animals, with additional necropsies being conducted on nonradioed animals collected opportunistically. Pregnancy and twinning rates were determined through radioimmunoassay of reproductive hormones in blood and feces, and calf observations post partum, respectively.Aerial moose surveys suggested that hunting was an unlikely source of the numerical decline because the level of harvest was relatively low (i.e., 3-25% per year) and the population usually grew in years following a hunt. The bull:cow and calf:cow ratios were markedly high throughout the population decline period but remained low following hunting cessation.

The majority of mortalities (62% of radioed moose [*n* = 76)] 54% of non-radioed moose [*n* = 94]) ere related to pathology associated with parasitism, infectious disease, and perhaps starvation, with few mortalities being associated with predation or poaching. Liver fluke infections, apparently the greatest single cause of death, were associated with pathology in the liver, thoracic and peritoneal cavities, pericardial sac, and lungs. Mortality due to meningeal worm (*Parelaphostrongylus tenuis*) appeared to be less prevalent. Bone marrow fat was lower for moose dying of natural causes than for those dying of anthropogenic factors or accidents, implying that acute malnutrition contributed to moose mortality. Blood profiles indicated that animals dying in the subsequent 18 months were chronically malnourished.

Average annual survival rates for adult cows (0.79 [0.74, 0.84; 95% CI]) and yearlings (0.64 [0.48, 0.86]) were low, whereas for calves (0.66 [0.53, 081]) survival rates were higher than in many other moose populations, with female calve survival rates being higher than for males. Moose exhibited low pregnancy (48%) and twinning (24%) rates, with reproductive senescence being observed as early as age 8 years among adult cows. Pregnancy status was related to indices of acute (bone marrow fat) and chronic (blood condition indices) malnutrition. Carcass recovery indicated that there likely were few prime-aged bulls (> 5 years old) in the population.

Analysis of protein content for the predominant browse species indicated that food quality was probably adequate to support moose over winter. Trace element analysis from necropsied moose livers revealed apparent deficiencies in copper and selenium concentrations, but there was limited association between trace elements and moose

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disease, pathology, or mortality

Time series analysis of regional moose population censuses (1961–2000) suggested that annual growth rate in the surveyed population was negatively related to mean summer temperature, with winter and summer temperatures increasing by about 6.8°C and 2.1°C, respectively, during the 40-year period. This change may have contributed to increased moose thermoregulatory costs and disruption of energy balance. Population rate of change also was associated positively with population size, implying inverse density-dependence and the absence of resource limitation in the study population.

We concluded that the decline in moose numbers in northwest Minnesota likely was caused principally and proximally by fluke parasitism, with additional mortality and reduced productivity being related to infectious disease and poor nutritional status; these factors likely interacted synergistically. Climatic changes also may have contributed to the population decline, and when combined with recent increases in deer numbers and parasite transmission rates, may have rendered northwest Minnesota inhospitable to moose. Our results imply that the southern distribution of moose may become restricted in the future if the phenomena observed in northwest Minnesota are common elsewhere in the southern range.

Abstract of paper accepted by Wildlife Monographs.

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DDE, PCB, AND MERCURY RESIDUES IN MINNESOTA COMMON GOLDENEYE AND HOODED MERGANSER EGGS: A FOLLOWUP

Michael C. Zicus and David P. Rave

SUMMARY OF FINDINGS

We collected 11 common goldeneye (*Bucephala clangula*) and 16 hooded merganser (*Lophodytes cucullatus*) eggs from northeastern Minnesota in spring 2004 to augment a sample of 45 goldeneye and 42 merganser eggs collected from northern Minnesota in 2003. Eggs were collected for contaminant assays, and to determine eggshell thicknesses. Contaminant assays have not begun yet, whereas eggshells have been measured. Mean eggshell thickness for the combined sample was 0.401 mm (SE = 0.003) and 0.606 mm (SE = 0.008) for common goldeneye and hooded merganser eggs respectively. This was 9.0 and 6.0% greater than in 1981 but still 7.8 and 3.5% less than that measured prior to the use of DDT (Zicus et al. 1988). Ratcliffe indexes for the combined goldeneye sample increased proportionately less than did eggshell thickness, and remained 4.8% less than the pre-1900 value. The index for mergansers was unchanged from 1981, and remained 5.6% less than the pre-DDT value, suggesting that eggshell density has not improved since 1981. Overall, these eggshell thickness/density metrics suggest a possible decrease in exposure to contaminants causing eggshell thinning for both mergansers and goldeneyes. Continued concern over mercury in the environment, and new concerns about polybrominated diphenyl ethers indicate contaminant assays of the collected eggs would be prudent because food habits of these species might cause them to be vulnerable to these contaminants. Funding is needed for the chemical assays.

INTRODUCTION

The Minnesota Department of Natural Resources' (MN DNR) fall use plan (Restoring Minnesota's Wetland and Waterfowl Hunting Heritage) attributes a major hurtle in attaining 16% of the Mississippi Flyway duck harvest to decreased waterfowl harvest from forested parts of the state. The plan states "*Total harvest has been below the 16% objective, as Minnesota harvested 9.5% of the flyway duck harvest in 1997-99. Also, distribution objectives are not being met in Minnesota. All major species were below the objective proportion of harvest in the forested portion of the state".* Further, the plan states, "*Maintaining a sizable population of Minnesota-breeding ducks is the cornerstone to improving fall duck use. These birds are important measures of the health of the ecosystem, and provide a substantial portion (25-33%) of Minnesota's duck harvest."*

Staff in the MN DNR Wetland Wildlife Populations and Research Group have been concerned about the status of Minnesota's common goldeneyes (*Bucephala clangula*) and hooded mergansers (*Lophodytes cucullatus*). These concerns were first voiced by area wildlife managers in the late 1970s, and prompted Zicus et al. (1988) to examine contaminants in eggs of these species. Although organochlorine pesticides and polychlorinated biphenyls (PCBs) were modest in both species, geometric mean mercury (Hg) levels in merganser eggs were considered high. Eggshells of both species were thinner than historic measurements with eggshell thickness in 1981 being 15.4% and 9.6% thinner for common goldeneye and hooded merganser eggs, respectively, than that measured around 1900 (Figure 1). Further, cracked or broken eggs were 8.5 times more common in successful goldeneye nests than in either successful wood duck (*Aix sponsa*) or hooded merganser nests.

Common goldeneyes have been identified in the MDNR's *Strategic Conservation Document* as an indicator species for the Forest Province, and the eggshell thinning in Minnesota common goldeneyes that had occurred by 1981 could have been contributing to significant loss in production from successful nests. In addition, mercury levels in some hooded merganser eggs were at levels in 1981 that cause neurological problems in mallards. Furthermore, a historic survey (1958-1990) in the Bemidji area (Figure 2) suggested a possible continuing decline in breeding common goldeneyes (Zicus and Rave 2003), which prompted us to reinstate the historic survey and to follow up the earlier contaminant study conducted by Zicus et al. (1988).

OBJECTIVES

- Determine the extent to which contaminant loads and eggshell thicknesses in common goldeneyes and hooded mergansers might have changed since 1981, and
- Restrict egg collection to northeastern Minnesota in 2004 to improve the sample distribution in the Laurentian Mixed Forest Province.

METHODS

Sample size estimation suggested that 40-50 eggs of each species collected from different nests would result in reasonable precision for the parameters of interest (J. Fieberg, MN DNR, unpublised data). We attempted to collect one unincubated egg randomly from each common goldeneye and hooded merganser nest primarily within the Laurentian Mixed Forest Province of Minnesota (Figure 3). Egg length, width, and mass were determined when each egg was collected. In the lab, egg contents were removed and frozen in chemically pre-cleaned jars for later chemical assay. Eggshells were dried, their mass determined, and thickness at the equator of each egg was measured in 3 random locations.

RESULTS

We collected 11 common goldeneye and 16 hooded merganser eggs from northeastern Minnesota in spring 2004. This sample augmented the 45 goldeneye and 42 merganser eggs collected from northern Minnesota in 2003. Cooperators collected most samples in 2004 and about one-half of the eggs in 2003. Mean eggshell thickness (Figure 4) measured at the equator for the combined sample was 0.401 mm (SE = 0.003) and 0.606 mm (SE = 0.008) for common goldeneye and hooded merganser eggs, respectively. These values are 9.0 and 6.0% greater than those measured in 1981 (Table 1), but still 7.8 and 3.5% less than those measured prior to the use of DDT.

Ratcliffe indexes, which are the eggshell mass divided by the product of the length and width of the egg (Ratcliffe 1967), changed proportionately less than did eggshell thicknesses. Mean Ratcliffe index (Figure 5) for the combined sample was 2.521 (SE = 0.021) and 3.778 (SE = 0.042) for common goldeneye and hooded merganser eggs respectively. The goldeneye index for the combined sample was 4.8% greater than in 1981, but still 4.8% less than for a sample of eggs collected prior to 1900 (Table 2). In contrast, there was no change in the hooded merganser index from 1981, which was 5.6% less than that of eggs collected prior to the use of DDT.

DISCUSSION

Organochlorine pesticides and PCBs in the environment are believed to have declined, but concentrations may still be high enough to cause problems for sensitive species. Although the amount of Hg being released into the atmosphere has declined, it is still being deposited in aquatic ecosystems of northern Minnesota in many locations, and has been identified as a concern in the federal Clear Skies Initiative http://www.epa.gov/air/clearskies/basic.html).

Mean eggshell thickness for both goldeneyes and mergansers increased significantly between 1981 and 2003, but eggshell density did not increase commensurately. This suggests a probable decreased exposure to compounds related to eggshell thinning during this period. This study will provide evidence of the extent to which organochlorine pesticides, PCBs, and Hg affecting common goldeneyes and hooded mergansers has changed since 1981. Further, polybrominated diphenyl ethers (PBDEs), a class of chemicals used extensively in fire retardants, have been detected recently in biological samples at unexpected rates (M. Briggs, MN DNR, personal communications). PBDEs are lipophilic and chemically similar to PCBs (http://www.ourstolenfuture.org/NewScience/oncompounds/PBDE/whatarepbdes.htm). As such, they are highly persistent and bioaccumulative. PBDEs are potent thyroid disrupters, and also may cause problems similar to those of PCBs. Investigations into PBDE levels in Great Lakes Region water birds have begun (http://dnr.wi.gov/org/es/science/inventory/Cormorants.pdf). Thus, we believe assays for PBDEs in goldeneye and merganser eggs would be prudent because their food habits might cause these species to be vulnerable to these contaminants.

Funding is needed before we can proceed with the chemical assays. We investigated some possible federal programs that might provide cost sharing for the analyses. Funding from these programs is awarded on a competitive basis, but the qualifying criteria are restrictive (D. Warburton, U.S. fish and Wildlife Service, personal communications). Qualifying points are awarded based in part on the share of the project cost funded by the non-federal partner. However, the time period during which MN DNR in kind costs would qualify and could be used as matching funds is short, and precludes most of our field collection efforts from qualifying.

Assay costs will vary depending on whether they could be done in partnership with the U.S. Fish and Wildlife Service (USFWS) or another agency. Costs also vary among the contracting labs doing the assays (D. Warburton, U.S. Fish and Wildlife Service, personal communications). Different contract labs working with the Patuxent Analytical Control Facility perform the USFWS assays. USFWS costs for organochlorine (OC) scans range from $400-460 per sample (non-USFWS costs - $480-550) with mercury assays ranging from $67-155 per sample (non-USFWS costs- $80-185). One contract lab will analyze for PBDEs. If PBDE analyses were part of a requested OC scan, the additional cost would be $150 per sample. If the assays were done through the Minnesota Department of Agriculture's (MDA) chemistry lab, the cost would be approximately $250/sample for selected OCs and mercury (M. Briggs, MN DNR, personal communications). However, MDA does not assay for PBDEs. Assays most comparable to the previous work (Zicus et al. 1988) but including PBDE analysis could be done through the Wisconsin Hygiene lab for $613 per egg. We would need to assay ~30 eggs of each species for precision comparable to the earlier work. Of course, fewer eggs could be assayed if less precise estimates were acceptable. PBDE assays seem particularly important in light of their harmful potential.

ACKNOWLEDGMENTS

We'd like to thank Mark Briggs for providing the collection jars for the chemical assays. Keith Backer, Kevin Carlisle, Dave Dickey, Walt Gessler, Mike Huber, Dave Ingebrigtsen, Dave Johnson, Perry Loegering, and Rich Staffon collected eggs for us.

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Table 1. Mean common goldeneye and hooded merganser eggshell thicknesses (mm ± 95% confidence interval) measured in Minnesota in 2003 – 2004 were greater than those measured in 1981 but still less than those measured ~1900.

Species	~1900	1981	2003-2004
Common goldeneye	0.435 ± 0.012[a]	0.368 ± 0.008[a]	0.401 ± 0.007
Hooded merganser	0.628 ± 0.049[b]	0.568 ± 0.014[a]	0.606 ± 0.015

[a]Zicus et al. 1988.
[b]Data from 1880 - 1927 (White and Cromartie 1977).

Table 2. Mean Ratcliffe indexes (± 95% confidence interval) for common goldeneye eggshells measured in Minnesota in 2003 – 2004 were greater than those measured in 1981 but still less than those measured ~1900 whereas hooded merganser indexes measured in 1981 and 2003 – 2004 were similar and remained less than those measured prior to 1947.

Species	~1900	1981	2003-2004
Common goldeneye	2.648 ± 0.176[a]	2.405 ± 0.045[a]	2.521 ± 0.040
Hooded merganser	4.000 ± 0.110[b]	3.757 ± 0.065[a]	3.778 ± 0.082

[a]Zicus et al. 1988.
[b]Data from pre-1947 (Faber and Hickey 1973).

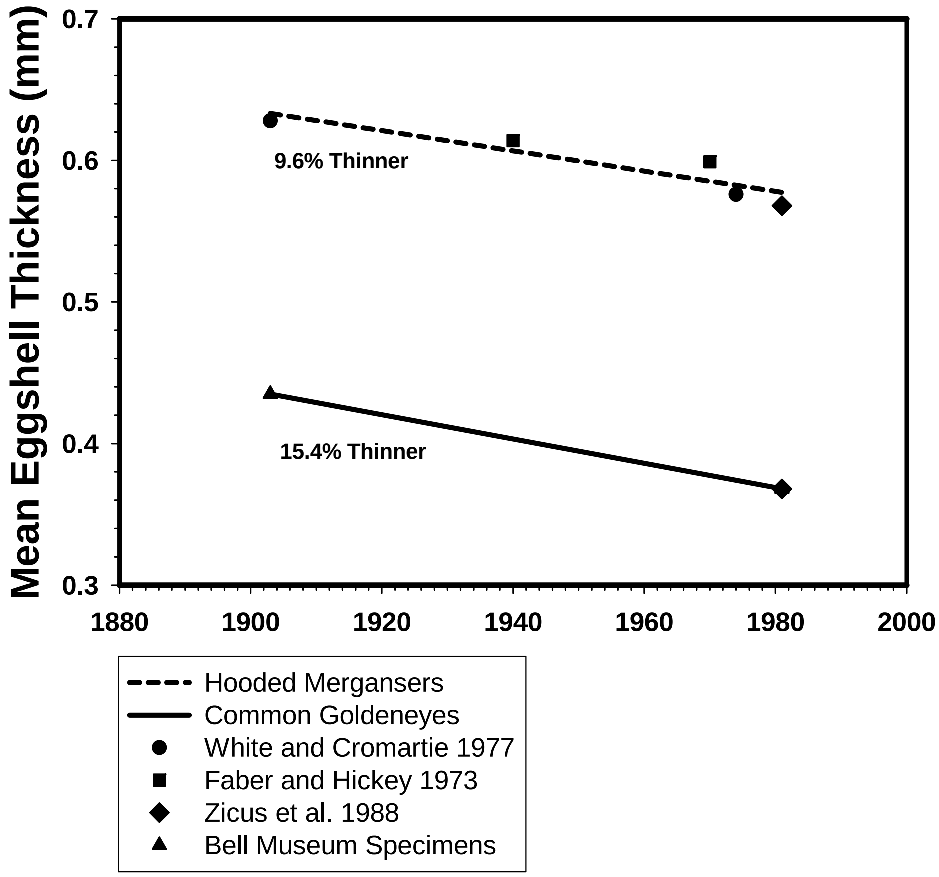

Figure 1. Mean eggshell thickness for common goldeneyes and hooded mergansers declined between 1900 and 1981.

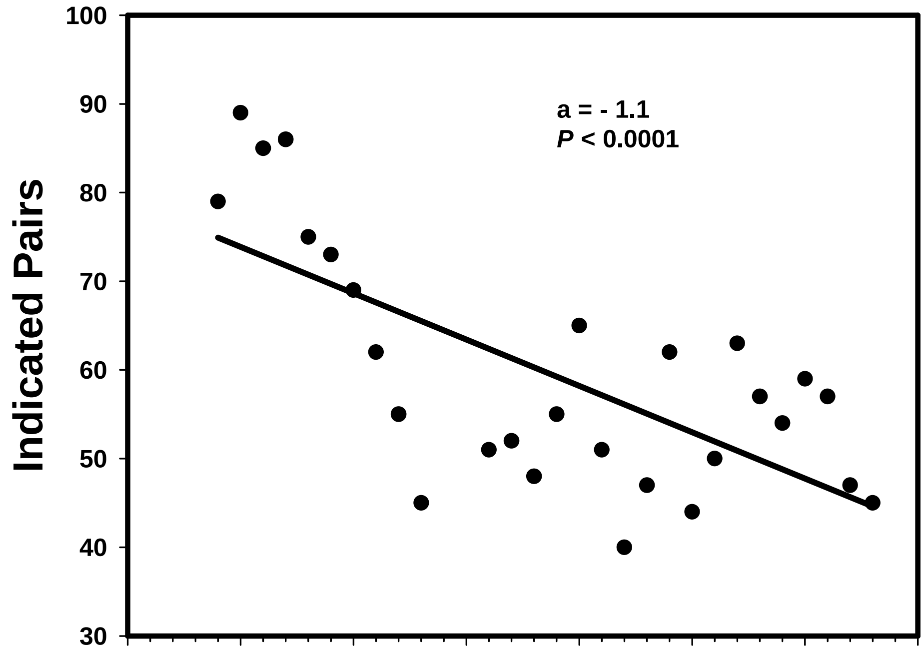

Figure 2. Indicated breeding common goldeneye pairs counted on the Bemidji Area Pair Survey declined during the period 1959-88.

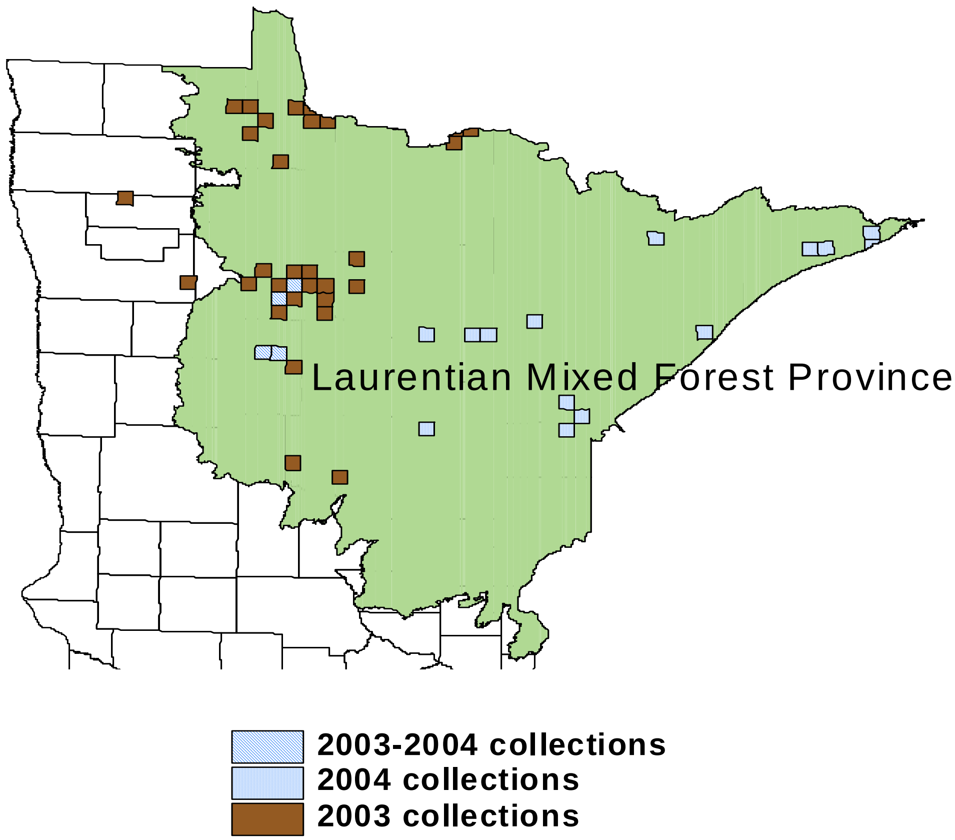

Figure 3. Minnesota townships where common goldeneye and hooded merganser eggs were collected in 2003 – 2004.

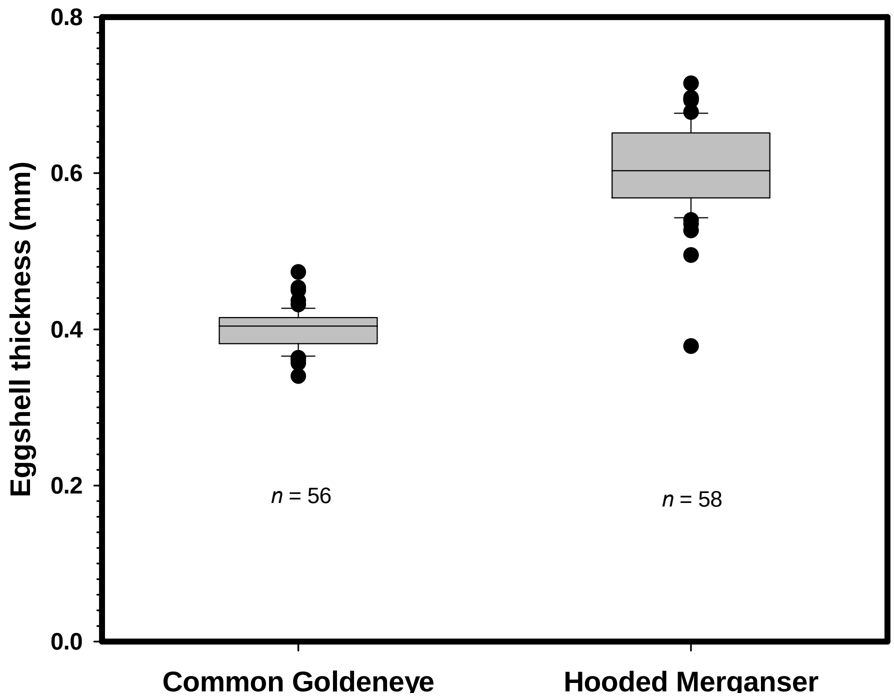

Figure 4. Box and whisker plots of eggshell thickness measured at the equator for common goldeneye and hooded merganser eggs collected from northern Minnesota, 2003 – 2004.

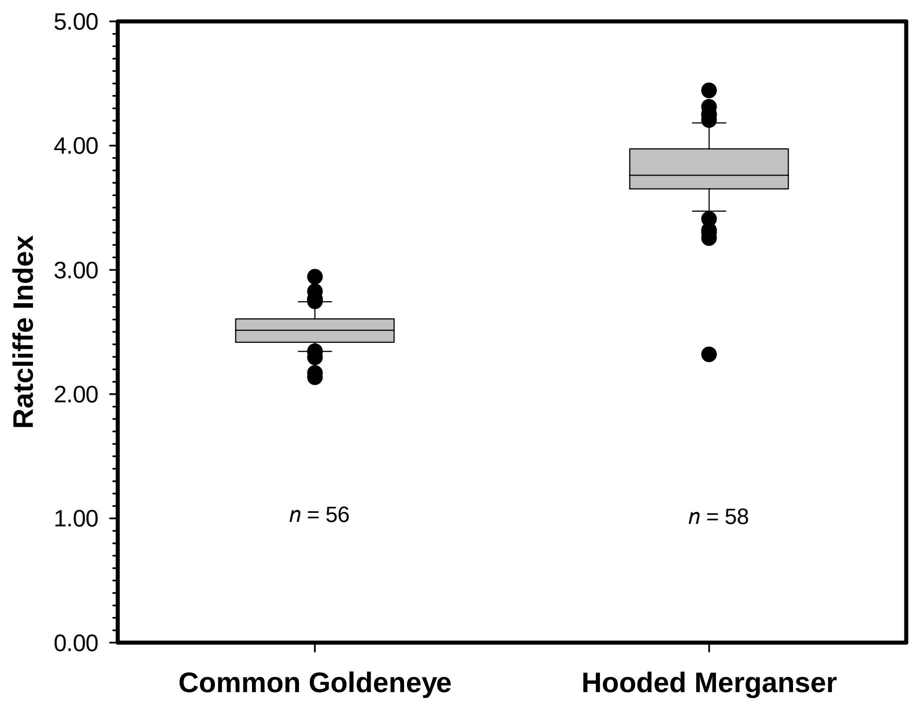

Figure 5. Box and whisker plots of Ratcliffe indexes (i.e., eggshell mass divided by the product of the length and width of the egg) for common goldeneye and hooded merganser eggs collected from northern Minnesota, 2003 – 2004.

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

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

SUMMARY OF FINDINGS

Ring-necked ducks (*Aythya collaris*) have been identified by the Minnesota Department of Natural Resources as an indicator species for the Forest Province. Little is known about the distribution and relative abundance of breeding ring-necked ducks in Minnesota because current waterfowl breeding pair surveys are inadequate for the species. In 2004, a pilot survey was conducted from 6 – 17 June in a portion of Minnesota considered primary breeding range. The helicopter survey entailed approximately 13 survey-crew days. Minnesota Department of Natural Resources' MN-GAP data were used to quantify presumed ring-necked duck nesting cover in Public Land Survey (PLS) section-sized survey plots, and 4 habitat classes were defined based on the amount of nesting cover in each plot. We apportioned 200 plots among 12 strata (i.e., 6 Minnesota Department of Natural Resources' Ecological Classification System sections x 2 habitat classes) using a stratified random sampling design. Plots in 2 habitat classes were not sampled because we believed that few ring-neck pairs would occur on these plots. The population of indicated breeding pairs was estimated to be ~ 9,000. Exploratory analyses were conducted to examine assumptions regarding duck visibility and absence of ducks on plots in the habitat classes that were not sampled, to examine estimation bias and plot size efficiency, and to assess the value of the stratification used. Similar numbers of ducks were counted from the air and the ground suggesting visibility was similar, but plots in habitat classes that were not sampled were misclassified, likely resulting in underestimation of breeding pairs. Plot misclassification resulted both from the way we used the MN-GAP data and from data limitations. PLS quarter sections might be a more efficient sampling unit than PLS sections;

however, additional analyses are required that would consider travel time and cost/sample unit. The stratification we used accounted for geographical and habitat based differences in ring-necked duck abundance.

INTRODUCTION

Staff in the Minnesota Department of Natural Resources (MNDNR) Wetland Wildlife Populations and Research Group has been developing a forest wetlands and waterfowl initiative. The status of ring-necked ducks (*Aythya collaris*) has been among the topics considered because the species has been considered an important forest resident, and it has been identified as an indicator species for the Forest Province (Minnesota Department of Natural Resources 2003).

Little is known about the current distribution and abundance of breeding ring-necked ducks in Minnesota. Moyle (1964) described the species as nesting primarily in the northern-forested portions of the state with appreciable numbers in the forest-prairie transition zone. At the time, ring-necks were believed to be the second most abundant species (to mallards) breeding in the forest zone. More recently, Hohman and Eberhardt (1998) described the primary breeding range as including areas south to approximately the Minnesota River. They also acknowledged local breeding to the Iowa border. In comparison, the MNDNR's *Gap Analysis Project* (MN-GAP) defined ring-neck breeding range as including any MNDNR Ecological Classification System (ECS) subsection where ring-necked duck reproduction had been documented (~87% of the state) (Minnesota Department of Natural Resources, Minnesota GAP Analysis Project, unpublished report).

Continentally, numbers of breeding ring-necks have been increasing, but this might not be the case in Minnesota (Figure 1). Current Minnesota waterfowl breeding pair surveys are inadequate for monitoring resident ring-necked ducks. The *Bemidji Area Ring-necked Duck Breeding Pair Survey* has been conducted in the Bemidji vicinity since 1969, and the survey includes lakes that were believed to be some of the best ring-necked duck lakes in north-central Minnesota when the survey was designed (Zicus et al. 2004). Unfortunately, the geographic extent of the survey is limited to the Bemidji vicinity. In contrast, the *Minnesota May Waterfowl Breeding Population and Habitat Survey* has a wider coverage that is directed primarily at mallards (*Anas platyrhynchos*), but the survey does not include much of the northern and eastern portion of the ring-neck breeding range (Maxson and Pace 1989). Further, this survey is conducted too early to provide useful information because ring-necked ducks arrive on breeding areas and begin nesting later than mallards (Hohman and Eberhardt 1998).

Sizable populations of breeding ducks in Minnesota are the cornerstones to improving fall duck use (Minnesota Department of Natural Resources 2001). Properly designed breeding population surveys are needed to monitor the status of all species of resident forest waterfowl; however, the biology of different species precludes the ability to survey all species with a single survey.

OBJECTIVES

- Initiate a pilot study to evaluate the feasibility of conducting a separate breeding-pair survey of ring-necked ducks in Minnesota; and
- determine the most appropriate sampling design and allocation for an operational survey, although this will in part depend on survey objectives (i.e., population estimates, population trends, distribution) and desired precision levels.

METHODS

We used a stratified random sampling design with 2 stratification variables: ECS sections and presumed nesting-cover availability (i.e., a surrogate for predicted breeding ring-necked duck density). This design is similar to that used for Minnesota's resident Canada geese (S. Maxson, Minnesota Department of Natural Resources, personal communication). We used a helicopter for the survey because visibility of ring-necked ducks from a fixed-wing airplane is poor in most ring-neck breeding habitats. We considered each pair, lone male, and males in flocks of fewer than 6 to indicate a breeding pair (J. Lawrence, Minnesota Department of Natural Resources, personal communications).

Statistical Population and Sampling Frame

The survey was restricted to the primary breeding range of ring-necked ducks in Minnesota (Figure 2) for logistical efficiency. Data from the U.S. Fish and Wildlife Service Habitat and Populations Evaluation Team's (HAPET) *4-square Mile Survey* were used to identify ECS subsections in the MN-GAP breeding range that represented peripheral breeding areas (D. Hertel, HAPET, unpublished data). Generally, we excluded subsections from the primary range if none of the 4-square mile plots in the subsection had at least an average of 1 ring-necked duck pair/year during a 10-year period. We also excluded plots if pairs were not counted on plots in at least 5 of the 10 years (based on data from HAPET plots). The Minnesota River Prairie subsection qualified as primary breeding range under these criteria, but it was excluded. Only 2 of the 97 4-square mile plots in this subsection had the required numbers of ring-necks and both plots were near the boundary with the

Hardwood Hills subsection, which was considered to be primary breeding range. The Boundary Waters Canoe Area and Twin Cities metropolitan counties were also excluded from the sampling frame because of flight restrictions and other logistical considerations.

Design and Sample Allocation

Preliminary observations during the spring 2004 Canada goose survey, where plots based on Public Land Survey (PLS) quarter-sections are used, suggested that it would be feasible to count ring-necked ducks on section-sized plots without redistributing ring-necked ducks on the plot. Therefore, we used PLS sections (~2.6-km^2 plots, range = 1.2 – 3.0 km^2) as the primary sampling units. Data were recorded by quarter sections to facilitate exploratory analyses regarding plot size and potential sources of bias. Presumed ring-necked duck nesting cover was defined as fine-leaf sedge and/or broad-leaf sedge-cattail cover within 250 m of and adjacent to open water (Minnesota Department of Natural Resources, Minnesota GAP Analysis Project, unpublished report). ArcInfo and ArcView GIS software (Environmental Systems Research Institute, Inc., Redlands, California, USA), and MN-GAP land cover data (Minnesota Department of Natural Resources 2004, U. S. Geological Survey 1989) were used to assign each PLS section to one of 4 habitat classes (Table 1). PLS sections at the periphery of the survey area that were less than 299 acres in size were removed from the sampling frame to reduce the probability of selecting these small plots.

The sampling frame consisted of 24 strata (i.e., 6 ECS sections x 4 habitat classes), but plots in habitat classes 3 and 4 were not sampled because the probability of ring-neck pairs occurring on these plots was assumed to be low. Thus, initial population estimates were based on 12 strata (i.e., 6 ECS sections x 2 habitat classes). Sample allocation was a 2-step process. For the pilot survey, 200 plots (i.e., PLS sections) were apportioned among ECS sections in proportion to the relative amount of presumed nesting cover within each ECS section. Within an ECS section, plots were then apportioned between habitat class 1 and 2 based on the proportion of total plots in each habitat class (i.e., proportional allocation). Survey plots were selected randomly from all plots in each stratum. Much is unknown regarding the usefulness of MN-GAP data as a stratification variable and the most efficient plot size. Therefore, breeding-pair observations were recorded by quarter section within survey plots to evaluate the validity of the assumption that ring-neck densities in habitat classes 3 and 4 were low and to assess questions about plot size efficiency for operational surveys.

Data Analyses

Estimated Population Size. – We estimated the population size for the survey area using 2 approaches. First, SAS Proc SURVEYMEANS (SAS 1999) was used to estimate population totals for each ECS section (i.e., a domain analysis) and the entire survey area. In this analysis, PLS sections were the primary sampling unit in a stratified random sampling design. Secondly, we estimated the population size for the entire survey area using ratio estimators to account for differences in plot size and nesting-habitat availability among plots (Cochran 1977).

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

Assumptions Regarding Plots in Habitat Classes 3 and 4. – Plots in habitat classes 3 and 4 were not sampled because we assumed that they would have few if any ring-necked duck pairs. We examined this assumption 2 different ways. First, PLS quarter sections that had no presumed nesting cover (classes 3 and 4) were sampled during the survey when they were part of a sampled PLS section. If ring-necked ducks were observed in these quarter sections, it would indicate the potential for having missed birds in PLS sections in habitat classes 3 and 4. However, these quarter sections were not sampled randomly and were near at least one other quarter section that had nesting cover (since the PLS section was sampled). Possibly, "no cover" quarter sections that were next to others with nesting cover would be more likely to have ring-necked ducks present than "no cover" quarter sections surrounded by other "no cover" quarter sections. To examine this possibility, we first calculated the number of quarter sections in sampled PLS sections that had at least some nesting cover (range = 1 to 3 quarter sections). Next, we constructed a frequency table of the number of indicated pairs in each "no cover" quarter section versus the number of quarters in the PLS section with available nesting cover. We then used the correlation statistic (Stokes et al. 2000) to test whether more indicated pairs were seen in those "no cover" quarters that were next to more quarter sections with nesting cover.

It was also possible that the number of ring-necked ducks observed on "no cover" quarter sections differed among ECS sections, or that the number observed on habitat class 3 quarter sections differed from that seen on habitat class 4 quarter sections. We tested these possibilities by comparing the distribution of indicated pairs in "no cover" quarter sections across ECS sections, and by comparing the distribution of indicated pairs in class 3 versus class 4 PLS quarter sections using row mean score tests (Stokes et al. 2000).

Further, we estimated the rate at which habitat classes were correctly assigned to PLS quarter section- and section-sized plots in habitat classes 3 and 4. We assessed classification accuracy by randomly selecting 100 plots for each plot size and habitat class, and visually inspecting aerial photos and National Wetlands Inventory data for the plots. When plots appeared to be incorrectly classified, we examined the MN-GAP data for the plot to determine why classifications were wrong.

Estimation Bias. – We estimated the number of indicated ring-necked duck pairs that might have been missed by not surveying PLS sections in habitat classes 3 and 4 in 2 different ways. To get a rough idea of how many birds might have been missed by the current sampling design, we multiplied the mean number of indicated pairs in "no cover" quarter sections by the total number of quarter sections in the survey area that were in PLS sections in habitat classes 3 and 4. We also estimated the number of indicated ring-necked duck pairs that might have been missed using a Monte Carlo simulation approach (Manly 1997). First, quarter-section samples were drawn randomly with replacement from habitat classes 1 and 2 (i.e., 12 strata = 6 ECS sections x 2 habitat classes). Second, quarter-section samples were drawn randomly with replacement from quarter sections in all habitat classes (i.e., 24 strata = 6 ECS sections x 4 habitat classes). Sample size was doubled in the second simulation to account for additional sampling effort in the habitat class 3 and 4 strata. The difference between the population estimates from the 2 simulations provided a second estimate of the bias in the pilot survey estimate.

Plot Size and Efficiency. – We estimated the approximate sample size required to estimate the ring-necked duck population size with a 25% bound (Scheaffer et al. 1996:137). We estimated sample sizes for both PLS section-sized plots and quarter section-sized plots. For both sampling units, we assumed that no ducks occupied plots in habitat classes 3 and 4. Additionally, we assumed the sample of quarter sections was independent.

Stratification Evaluation. – If stratification performed well, then it would account for differences in indicated ring-necked duck pairs seen on plots among the strata in the survey. We used SAS Proc GLM to evaluate the stratification that we used by testing for differences in the mean number of indicated pairs seen among the different ECS sections and within the habitat classes in the ECS sections.

RESULTS

The pilot survey was conducted 6 – 17 June and entailed approximately 13 survey-crew days. Survey plots were concentrated somewhat in the central and western parts of the survey area (Figure 3). The most plots (78) were located in the Northern Minnesota Drift and Lake Plains Section, while the fewest plots (13) were located in the Northern Superior Uplands Section (Table 2). The highest sampling rate occurred in the Lake Agassiz, Aspen Parklands Section with the lowest rate occurring in the Northern Superior Uplands Section. The amount of presumed nesting cover in the sample plots was highly skewed (Figure 4). Plots in habitat class 1 contained from 3.23 – 86.88 ha of cover while those in habitat class 2 contained 0.03 – 3.17 ha of presumed ring-necked duck nesting cover. Pairs represented 57% of the indicated pairs tallied during the survey (Table 3).

Estimated Population Size

Estimates of the total number of indicated breeding pairs in the survey area ranged from 8,449 – 9,059 and had similar precision (Table 4). Exploratory scatter plots and smoothed trend lines did not support the need for ratio estimators to adjust population estimates for differences in plot size or nesting cover among sample plots. All 3 estimates would be biased low if plots in habitat classes 3 and 4, which were not sampled, contained uncounted ring-necked duck pairs.

Indicated breeding pairs of ring-necked duck were most abundant in the Northern Minnesota Drift and Lake Plains Section and least abundant in the combined Western and Southern Superior Uplands Section (Table 5). The number of indicated breeding pairs seen on survey plots was notably greater in the Lake Agassiz, Aspen Parklands Section, northwestern portion of the Northern Minnesota Drift and Lake Plains Section, and the northern portion of the Minnesota and North East Iowa Morainal Section than the remainder of the survey area (Figure 5).

Aerial Visibility

Boat counts and the air counts of indicated breeding pairs differed somewhat for the individual lakes included in the *Bemidji Area Ring-necked Duck Pair Survey* (Figure 6). This was expected as ring-necked duck pairs are mobile and surveys of individual lakes were separated in time. In total, similar numbers of indicated ring-necked duck pairs were seen in both surveys. Furthermore, regression analysis suggested both surveys detected an equal proportion of the population (air to ground slope = 0.92, 95% confidence interval = 1.29 – 0.55).

Assumptions Regarding Plots in Habitat Classes 3 and 4

Indicated ring-necked duck pairs were observed on quarter sections that would have been in habitat classes 3 and 4 if the survey had used quarter section-sized plots (Table 6). There was no indication ($\chi_1^2 = 0.51$, $P = 0.47$) that more ducks were seen in those quarter sections that were next to quarter sections containing nesting cover (Table 7). However, the power to detect an effect of neighboring quarter sections was likely low. The distribution of counts across ECS sections (Table 8) appeared to differ ($\chi_5^2 = 12.1$, $P = 0.034$). Nonetheless, 90-100% of the quarter sections in habitat classes 3 and 4 had no indicated pairs regardless of the ECS section. Further, the distributions of indicated pair counts among quarter sections in habitat classes

3 and 4 (Table 9) were similar (χ_1^2 = 0.80, P = 0.37).

Public Land Survey quarter section- and section-sized plots having a habitat classification of 3 or 4 in the Minnesota survey area were misclassified at a high rate (Table 10). Classification errors were more common for plots initially placed in habitat class 3 (56 – 58%) using MN-GAP data than for those placed in habitat class 4 (33 – 40%). More misclassifications resulted from our use of MN-GAP data than from data limitations (Table 11).

Estimation Bias

The observed density of indicated ring-necked duck pairs in sampled PLS quarter sections that would have been placed in habitat classes 3 and 4 (Table 12) varied among ECS sections. This indicated the potential for uncounted pairs in most ECS sections. Based on the overall density of indicated pairs in these PLS quarter sections and an unstratified design, an estimated 10,092 (95% CI = 4,784 – 15,379) indicated pairs were not counted. The number of uncounted indicated pairs (9,338) estimated using Monte Carlo simulations (Table 13) was similar.

Plot Size and Efficiency

Plot size efficiency was examined only from the point of sample sizes and total area surveyed that would be needed to estimate the ring-necked duck population with 25% bounds when surveying plots in habitat classes 1 and 2 (Table 14). At this point, more than twice as many PLS section-sized plots and nearly 4 times as many PLS quarter section-sized plots would be needed to achieve the desired precision. Section-sized plots would require that twice as much total area be surveyed. In comparison, quarter section-sized plots would require surveying only as much area as included in this year's pilot survey.

A complete examination of plot size efficiency will require consideration of the time required to fly to and among plots in the sample as well as the number of refueling stops required. Time required to survey a plot varied (Figure 7), ranging from 1 – 29 minutes (mean = 7.2 minutes).

Stratification Evaluation

Analysis of variance indicated that the stratification used in the pilot survey performed well. Indicated pairs were related significantly to ECS sections ($F_{5,188}$ = 2.29, P = 0.049) and to habitat classes within the ECS sections ($F_{1,188}$ = 7.19, P = 0.008). Counts of indicated pairs were not related to an interaction between ECS section and habitat class ($F_{5,188}$ = 0.89, P = 0.487). Pair density was greatest in the Lake Agassiz, Aspen Parkland habitat class 1 stratum plots. In contrast, no indicated pairs were observed in any Northern Minnesota and Ontario Peatlands habitat class 2 plots (Table 15).

DISCUSSION

Information gained from the pilot survey has provided us with a better understanding of the issues involved in designing and conducting a survey to estimate the abundance and describe the distribution of breeding ring-necked ducks in Minnesota. Survey dates appeared appropriate because 57% of the indicated pairs were counted as paired birds, and survey timing is considered optimal when most birds are counted as pairs and not in flocks (Smith 1995). The stratified random sampling design that we employed seemed to perform well, but survey plots in habitat classes 3 and 4 were misclassified at an unacceptably high rate. We did not sample plots in habitat classes 3 and 4 because these classes were defined as having little or no nesting habitat. Thus, we had assumed that few if any ring-necked duck pairs would occur on these plots. Post-hoc classification of 400 habitat class 3 and 4 plots using aerial photography indicated that >25% would have been correctly classified as habitat class 1 or 2 plots. As a result, the population estimate (~9,000 indicated pairs) derived from the survey is almost certainly biased low. The magnitude of the bias could be substantial (9,000-10,000 missed pairs) because >79% of the survey area was placed in habitat classes 3 and 4. Plot misclassification occurred both because of limitations in the MN-GAP data (~40%) that we used and because of the way we used the data (~60%). There was no indication that indicated ring-neck duck pair estimates based on helicopter counts would be biased because of incomplete visibility.

Preliminary analysis indicated PLS quarter sections may be a more efficient sampling unit than PLS sections; however, additional analyses are required that would consider travel time and cost/sample unit. The current stratified sampling design, with PLS sections as sampling units, should provide a reasonably accurate and precise population estimate for the sampling effort used in the pilot survey if classification errors can be minimized and plots in habitat classes 3 and 4 contain essentially no ring-necked duck pairs. Currently, we have begun reprocessing the MN-GAP data to reduce the habitat class misclassification rate. We will have more lead time with the data this year and intend to assess the classification error rates prior to the survey. However, we believe plots in habitat class 3 and 4 should be surveyed, at least for a few years because so much of the survey area is included in these habitat classes.

We intend to conduct the pilot survey for a second year in 2005, again sampling PLS sections in habitat classes 1 and 2 using a stratified random design. In addition, we will sample PLS sections in habitat classes 3 and 4 using a double sampling approach (Thompson 1992). We will draw a large initial simple random sample of PLS sections from all PLS sections falling in habitat classes 3 and 4. Aerial photos and National Wetland Inventory data for these sampled sections will then be inspected to determine sections that may have been misclassified. We will then survey a random subsample of 50 potentially misclassified sections in 2005, requiring approximately 20 additional hours of flight time.

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Table 1. Minnesota ring-necked duck breeding pair survey habitat classes, June 2004.

Habitat class	Definition[a]	%[b]
1	Survey plots that have ≥ the median amount (3.18 ha) of MN-GAP cover class 14 and/or 15 nesting cover that was within 250 m of and adjacent to open water (i.e., potentially high pair numbers).	15.3
2	Survey plots that have < the median amount (3.18 ha) of MN-GAP cover class 14 and/or 15 nesting cover that was within 250 m of and adjacent to open water (i.e., potentially low pair numbers).	15.3
3	Survey plots that have no MN-GAP cover class 14 or 15 nesting cover but that include open water that is <250 m from a shoreline (i.e., possibly some pairs).	25.2
4	Survey plots that have no MN-GAP cover class 14 or 15 nesting cover or that include only open water ≥250 m from a shoreline (i.e., no pairs).	44.2

[a]Survey plots are Public Land Survey sections. MN-GAP cover class 14 is described as wetlands with <10% tree crown cover that is dominated by emergent herbaceous vegetation such as fine-leaf sedges. MN-GAP cover class 15 is described as wetlands with <10% tree crown cover that is dominated by emergent herbaceous vegetation such as broad-leaf sedges and/or cattails.

[b]Percent of the survey area.

Table 2. Minnesota Ecological Classification System section sample plots (i.e., Public Land Survey sections) and sampling rates in the Minnesota ring-necked duck breeding survey, June 2004.

Ecological Classification System section	Area[a]	Sample plots	Sampling rate (%)
W & S Superior Uplands[b]	1,638	18	1.1
Northern Superior Uplands	1,810	13	0.7
N Minnesota & Ontario Peatlands	1,817	26	1.4
N Minnesota Drift & Lake Plains	5,048	78	1.5
Minnesota & NE Iowa Morainal	3,510	50	1.4
Lake Agassiz, Aspen Parklands	316	15	4.7

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

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

Table 3. Ring-necked ducks counted in each Ecological Classification System section in the Minnesota ring-necked duck breeding survey, June 2004.

Ecological Classification System section	Pairs	Lone males	Flocked males[a]	Lone females	Grouped birds[b]
W & S Superior Uplands[c]	2	1	0	0	0
Northern Superior Uplands	6	1	0	0	0
N Minnesota & Ontario Peatlands	6	2	4	0	7
N Minnesota Drift & Lake Plains	30	9	16	3	11
Minnesota & NE Iowa Morainal	26	6	8	0	0
Lake Agassiz, Aspen Parklands	23	11	11	1	11

[a]Males in a flock of <6.
[b]Mixed sex flocks that could not be separated into pairs or $\geq$6 males in a flock.
[c]Western and Southern Superior Uplands sections combined due to the small area of the Southern Superior Uplands occurring in the survey area

Table 4. Estimated number of indicated breeding ring-necked duck pairs in the Minnesota survey area, June 2004.

Estimator	Indicated pairs[a]	Upper 95% CL	Lower 95% CL	CV(%)
Stratified random	8,999	12,059	5,938	17.2
Ratio (plot size)	9,059	12,130	5,989	17.3
Ratio (nesting cover)	8,449	11,651	5,247	19.3

[a]Population estimates might be biased low because Public Land Survey sections classified as containing no nesting cover (classes 3 and 4) were not sampled.

Table 5. Estimated number of indicated breeding ring-necked duck pairs in the Ecological Classification System sections in the Minnesota survey area, June 2004.

	Indicated pairs[a]				
Ecological Classification System section	Density[b]	Estimate	Upper 95% CL	Lower 95% CL	CV (%)
W & S Superior Uplands[c]	0.1667	273	702	4	74.1
Northern Superior Uplands	0.3204	580	1,270	9	54.0
N Minnesota & Ontario Peatlands	0.4651	845	2,275	36	82.0
N Minnesota Drift & Lake Plains	0.7066	3,567	5,109	2,025	21.7
Minnesota & NE Iowa Morainal	0.7974	2,799	4,906	691	37.4
Lake Agassiz, Aspen Parklands	2.9589	935	1,582	288	32.0

[a]Population estimates were based on a stratified random sample of Public Land Survey (PLS) sections in habitat classes 1 and 2. Population estimates might be biased low because PLS sections classified as containing no presumed nesting cover (classes 3 and 4) were not sampled.
[b]Average density of indicated pairs (per PLS section-sized plot) in habitat class 1 and 2 plots.
[c]Western and Southern Superior Uplands sections combined due to the small area of the Southern Superior Uplands occurring in the survey area.

Table 6. Post-hoc habitat classification of Public Land Survey (PLS) quarter sections having indicated breeding pairs of ring-necked ducks, June 2004. These quarter sections were part of surveyed (habitat class 1 or 2) PLS sections, but would have been classified as having little or no nesting cover (habitat class 3 or 4) in the Minnesota survey if quarter section-sized plots had been used.

		Post-hoc classification (%)[a]		
Habitat class[b]	No. of quarter sections with indicated pairs	Class 1 or 2	Class 3	Class 4
3	13	84.6	15.4	0.0
4	6	50.0	50.0	0.0

[a]Based on aerial photos and National Wetland Inventory data.
[b]Based on MN-GAP data.

Table 7. Cross tabulation of 406 Public Land Survey (PLS) quarter sections in habitat classes 3 or 4. Quarter sections were cross tabulated by the number of adjoining PLS quarter sections in habitat classes 1 or 2 and the number of indicated ring-necked duck breeding pairs in the quarter section. Each PLS quarter section with a habitat class of 3 or 4 and its adjoining PLS quarter sections were part of a PLS section chosen as a survey plot in Minnesota, June 2004.

	Indicated pairs/quarter section				
No. of habitat class 1 or 2 quarter sections[a]	0	1	2	3	4
1	223 (94.5)[b]	9 (3.8)	1 (0.4)	2 (0.9)	1 (0.4)
2	133 (96.4)	3 (2.2)	0	1 (0.7)	1 (0.7)
3	31 (96.9)	1 (3.1)	0	0	0

[a]Classifications based on MN-GAP data.
[b]Number of quarter sections (row percent).

Table 8. Cross tabulation of 406 Public Land Survey (PLS) quarter sections in habitat classes 3 or 4. Quarter sections were cross tabulated by Ecological Classification System section and the number of indicated ring-necked duck breeding pairs in the quarter section. Each PLS quarter section was part of a PLS section chosen as a survey plot in Minnesota, June 2004.

	Indicated pairs/quarter section				
Ecological classification system section	0	1	2	3	4
W & S Superior Uplands[a]	41 (93.2)[b]	3 (6.8)	0	0	0
Northern Superior Uplands	27 (100.0)	0	0	0	0
N Minnesota & Ontario Peatlands	60 (98.4)	1 (1.6)	0	0	0
N Minnesota Drift & Lake Plains	148 (93.7)	6 (3.8)	1 (0.6)	3 (1.9)	0
Minnesota & NE Iowa Morainal	83 (97.7)	2 (2.4)	0	0	0
Lake Agassiz, Aspen Parklands	28 (90.3)	1 (3.2)	0	0	2 (6.5)

[a]Western and Southern Superior Uplands sections combined due to the small area of the Southern Superior Uplands occurring in the survey area.
[b]Number of quarter sections (row percent).

Table 9. Cross tabulation of 406 Public Land Survey (PLS) quarter sections in habitat classes 3 or 4. Quarter sections were cross tabulated by habitat class and the number of indicated ring-necked duck breeding pairs in the quarter section. Each PLS quarter section was part of a PLS section chosen as a survey plot in Minnesota, June 2004.

	Indicated pairs/quarter section				
Habitat class	0	1	2	3	4
3	218 (94.4)[a]	9 (3.9)	0	3 (1.3)	1 (0.4)
4	169 (96.6)	4 (2.3)	1 (0.6)	0	1 (0.6)

[a]Number of quarter sections (row percent).

Table 10. Post-hoc classification of 400 randomly selected Public Land Survey quarter section- and section-sized plots in habitat classes 3 or 4 in the Minnesota survey area.

			Post-hoc habitat class[a]		
Plot size	Habitat class[b]	*n*	1 or 2	3	4
Quarter section	3	100	30	44	26
Quarter section	4	100	8	25	67
Section	3	100	50	42	8
Section	4	100	17	23	60

[a]Based on aerial photos and National Wetland Inventory data.
[b]Based on MN-GAP data.

Table 11. Source of misclassifications in post-hoc classification of 400 randomly selected Public Land Survey quarter section- and section-sized plots in habitat classes 3 or 4 in the Minnesota survey area.

				Source of misclassifications			
Plot size	Habitat class[a]	*n*	Correctly classified[b]	MN-GAP limitations[c]	Incorrect GIS analysis	Minimum area problem[d]	Oversight[e]
Quarter section	3	100	44	20	22		14
Quarter section	4	100	67	17		3	13
Section	3	100	42	25	10	3	20
Section	4	100	60	20		2	18

[a]Based on MN-GAP data.
[b]Based on aerial photos and National Wetland Inventory data.
[c]Wetland and nesting cover features misclassified or too small to be delineated in MN-GAP data.
[d]Definition of minimum patch size for open water (0.6 ha) was too large.
[e]MN-GAP cover class 10 (lowland deciduous shrub) should have been combined with classes 14 (fine-leaf sedge) and 15 (broad-leaf sedge/cattail) to better describe presumed nesting cover, and cover class 13 (floating aquatic) should have been combined with class 12 (open water) to better describe the extent of a wetland basin.

Table 12. Estimated number of indicated ring-necked duck breeding pairs occurring in Public Land Survey sections in habitat classes 3 and 4 in the Minnesota survey, June 2004. These are estimates of the pairs that were uncounted.

Ecological classification system section	Pair density[a]	No. of quarter sections	Estimate	Upper 95% CL	Lower95 % CL
W & S Superior Uplands[b]	0.0682	15,342	1,046	2,202	0
Northern Superior Uplands	0.0000	23,578	0	0	0
N Minnesota & Ontario Peatlands	0.0164	26,109	428	1,267	0
N Minnesota Drift & Lake Plains	0.0886	32,757	2,903	5,011	780
Minnesota & NE Iowa Morainal	0.1047	13,274	1,389	2,942	0
Lake Agassiz, Aspen Parklands	0.1667	16,981	2,830	7,367	0

[a]Average density of indicated pairs (per Public Land Survey quarter section-sized plot).

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

Table 13. Monte Carlo simulations used to estimate the potential bias in the estimated ring-necked duck population resulting from not sampling PLS sections in habitat classes 3 and 4. Estimates are based on a stratified sampling design using quarter section sampling units. The difference between the 2 estimates represents the uncounted pairs in the survey area.

Simulation	Replications	No. of plots	Indicated pairs	Upper 95% CL	Lower 95% CL
1[a]	250	200	7,024	10,242	3,806
2[b]	250	407	16,362	26,007	6,717

[a]Samples drawn randomly (with replacement) from PLS quarter-sections with a habitat class of 1 or 2.

[b]Samples drawn randomly (with replacement) from all PLS quarter-sections. Sample size was doubled to account for additional sampling effort required to sample plots in habitat classes 3 and 4.

Table 14. Approximate sample sizes required to estimate population size with a 25% bound for PLS section- and quarter section-sized plots. Sample size determination assumed that there were no indicated breeding pairs in plots in habitat classes 3 and 4, that plots were allocated proportionally among strata, and that quarter section sized plots were independent.

Plot size	Allocation	Strata	Desired bound (%)	Sample size	~Area (mi.^{2})
Sections	Proportional	12	25%	412	412
Quarter sections	Proportional	12	25%	786	197

Table 15. Estimated density of indicated ring-necked duck breeding pairs occurring in Public Land Survey section-sized plots in habitat classes 1 or 2 in the Minnesota survey, June 2004.

Strata		Indicated pairs/plot	
Ecological classification system section	Habitat class	Mean	Variance
W & S Superior Uplands[a]	1	0.13	0.13
	2	0.20	0.40
Northern Superior Uplands	1	0.75	0.92
	2	0.11	0.11
N Minnesota & Ontario Peatlands	1	1.00	8.18
	2	0.00	0.00
N Minnesota Drift & Lake Plains	1	1.02	3.00
	2	0.33	0.51
Minnesota & NE Iowa Morainal	1	1.04	7.07
	2	0.50	1.40
Lake Agassiz, Aspen Parklands	1	3.73	18.42
	2	1.00	4.00

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

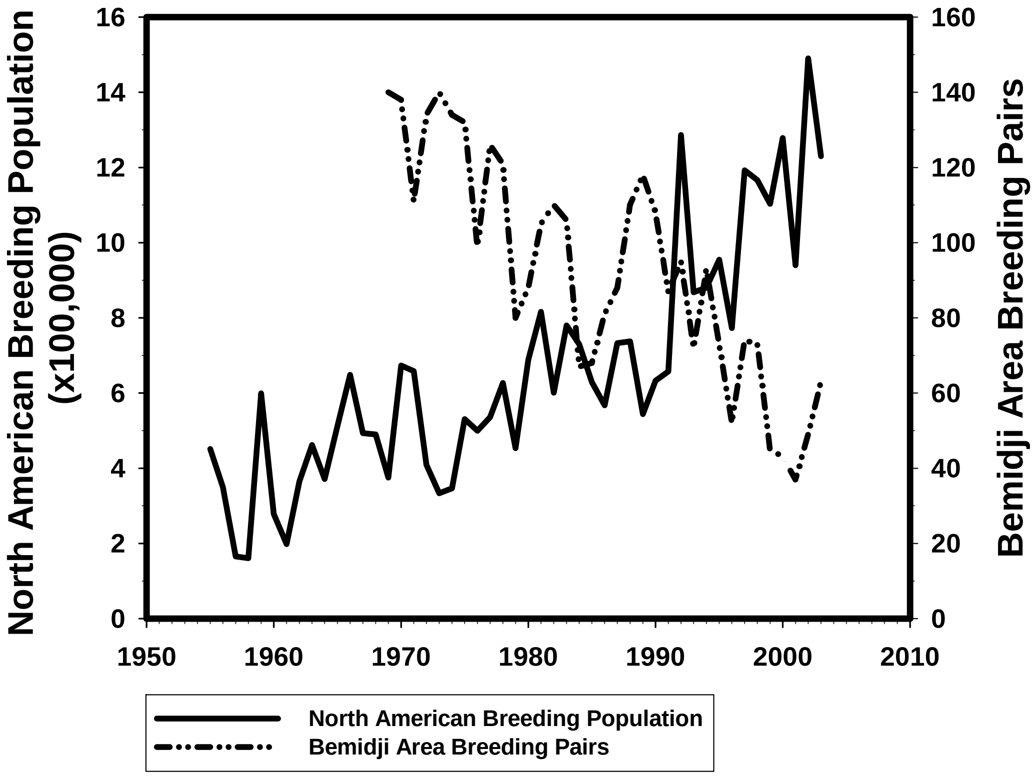

Figure 1. Ring-necked duck breeding population trends as reflected by the U.S. Fish and Wildlife Service *Breeding Pair Survey* and the Minnesota Department of Natural Resources' *Bemidji Area Ring-necked Duck Survey* (Zicus et al. 2004).

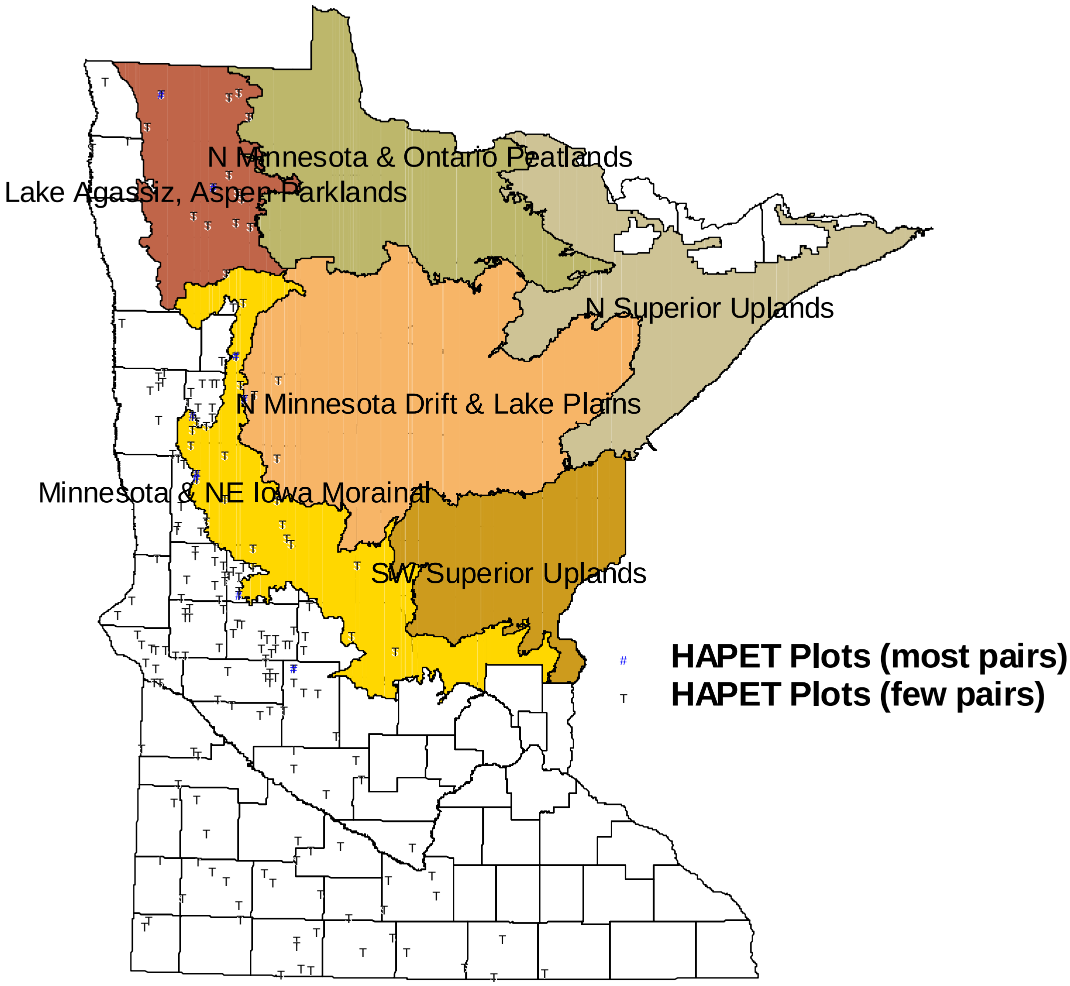

Figure 2. Minnesota Ecological Classification sections included in the pilot ring-necked duck breeding pair survey in 2004. Western and Southern Superior Uplands sections were combined due to the small area of the Southern Superior Uplands occurring in the survey area. Circles and triangles denote U.S. Fish and Wildlife Service 4-square mile survey plots used to define the primary ring-necked duck breeding range.

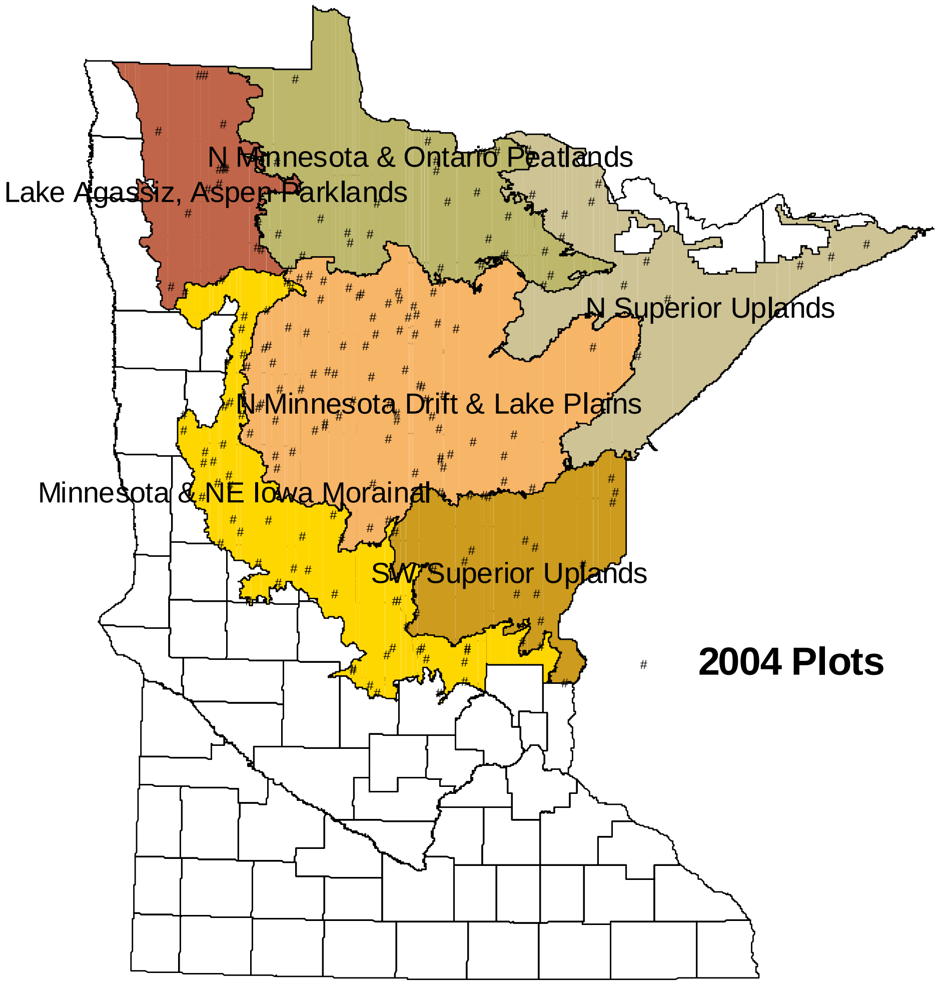

Figure 3. Survey plots included in the pilot ring-necked duck breeding pair survey, 2004. Western and Southern Superior Uplands sections combined due to the small area of the Southern Superior Uplands occurring in the survey area.

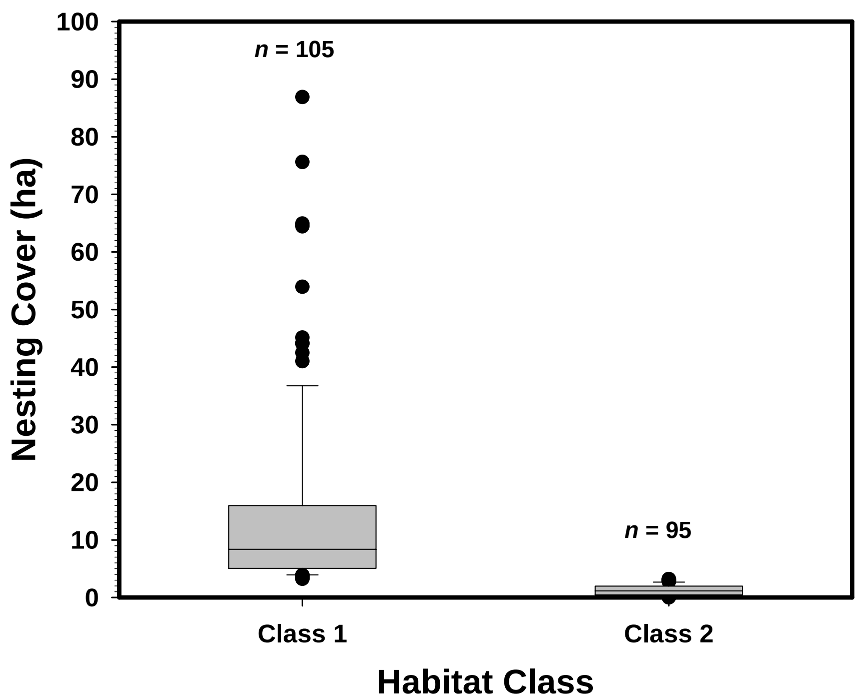

Figure 4. Box and whisker plots of the amount (ha) of nesting cover (sedge meadow and broadleaf sedge/cattail cover associated with open water) contained in habitat class 1 and 2 plots sampled in the ring-necked duck breeding pair pilot survey, 2004. Sedge meadow and broadleaf sedge/cattail cover was determined from MN-GAP data.

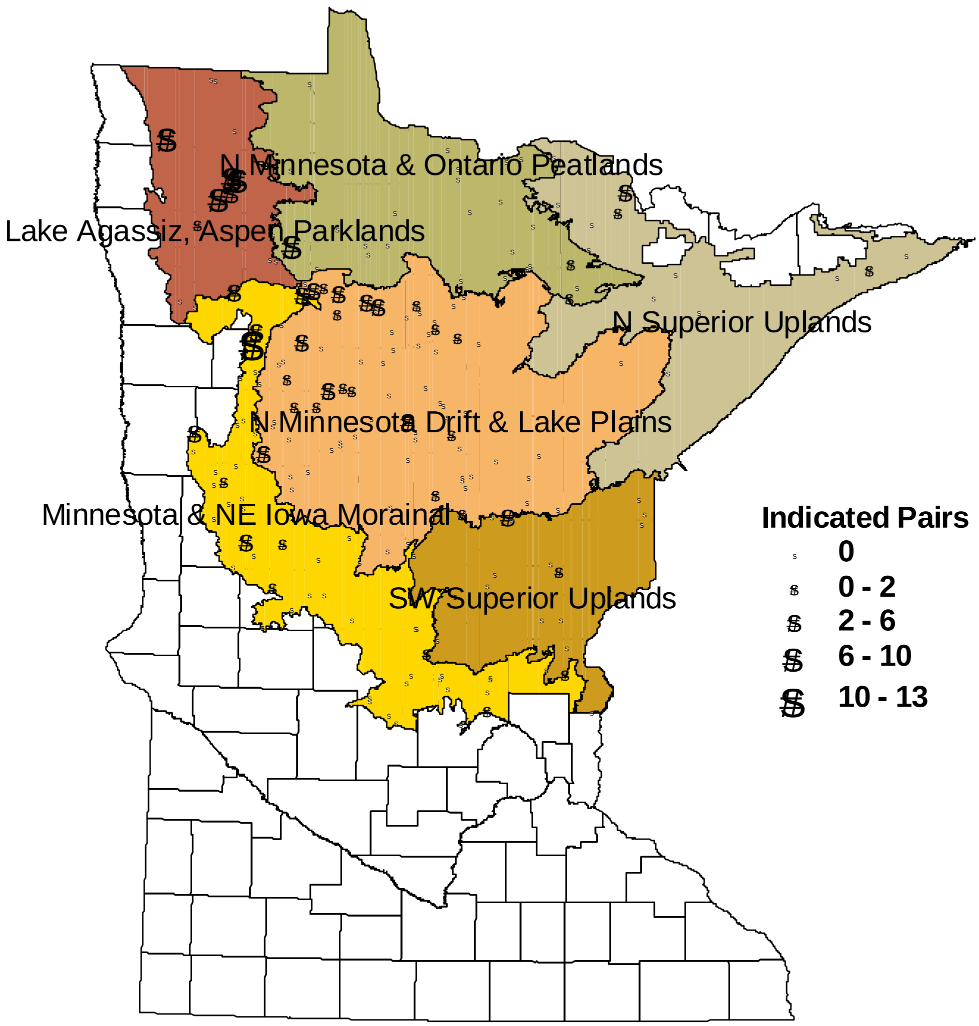

Figure 5. Number of indicated breeding pairs of ring-necked ducks observed on survey plots in the Minnesota survey area, June 2004. Western and Southern Superior Uplands sections combined due to the small area of the Southern Superior Uplands occurring in the survey area.

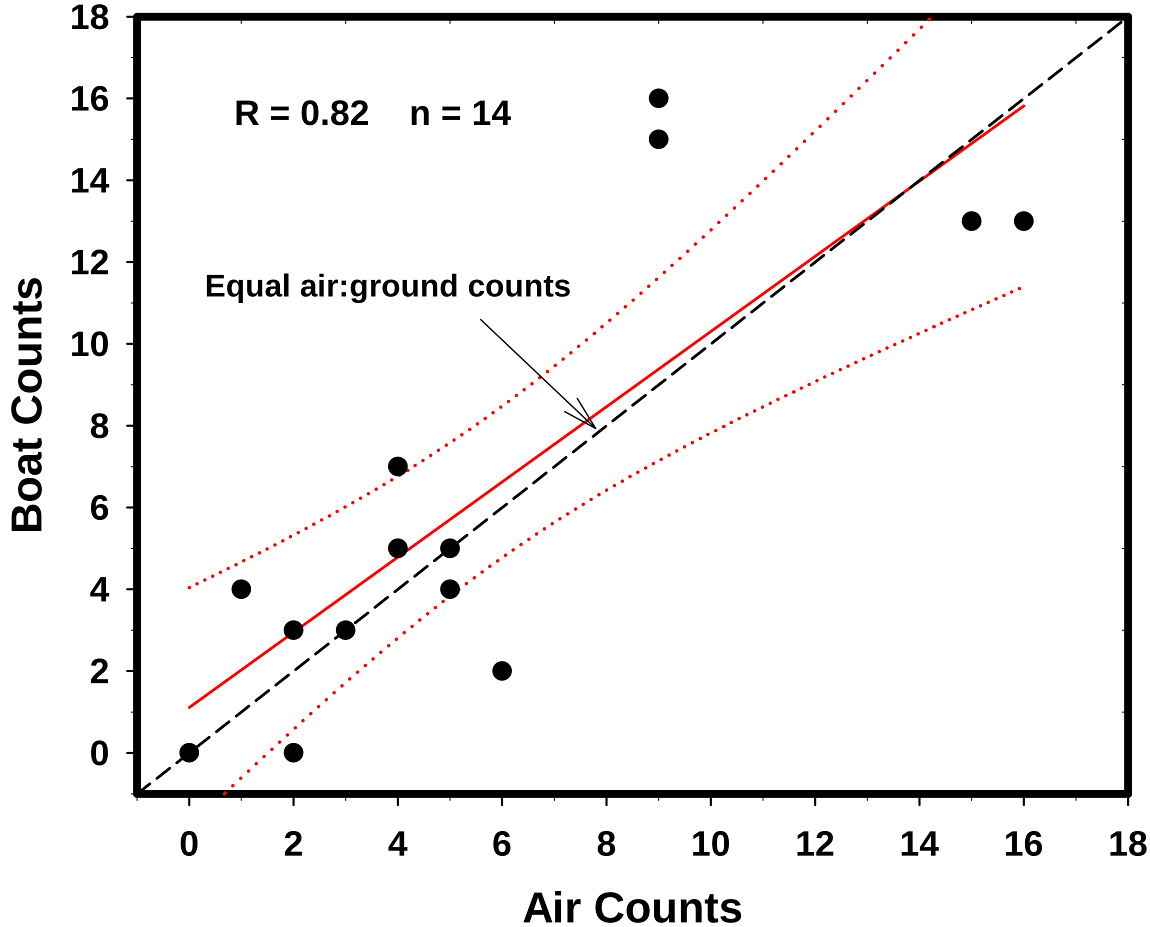

Figure 6. Regression line and 95% confidence interval comparing the numbers of indicated breeding pairs of ring-necked ducks counted from a boat and from the air on the same 14 lakes in the Bemidji vicinity, June 2004.

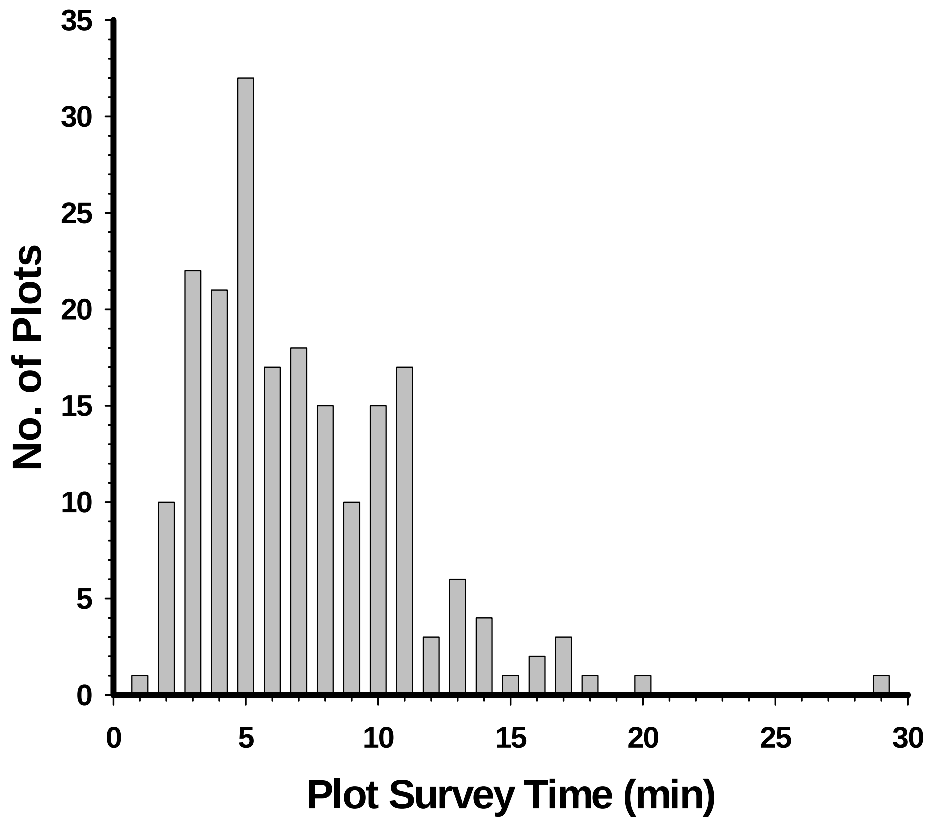

Figure 7. Time required for individual ring-necked duck breeding pair survey plots in the Minnesota survey area, June 2004.

SEASONAL FOREST WETLANDS: CHARACTERISTICS AND INFLUENCES

Mark A. Hanson[1], Fred Ossman[2], and Shane Bowe[3]

SUMMARY OF FINDINGS

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

INTRODUCTION

In 1999, we initiated a study of 24 small, seasonally-flooded ($\leq$ 1.5 acres) wetlands in aspen-dominated landscapes of the Buena Vista and Paul Bunyan state forests in north central Minnesota. Study wetlands were assigned to one of three "age-class" levels of treatment, or identified as controls (Figure 1) based upon adjacent forest (stand) age-since-harvest using natural breaks identified with Arcview. We also blocked study sites on the basis of proximity to account for local influences of soils, landforms, or other geophysical features. We assigned study wetlands to clusters, each consisting of 4 adjacent wetlands (1 in each of 4 treatment groups) all located within the same general state forest area. Each state forest (hence subsection of the Ecological Classification System [ECS] Almedinger and Hanson 1998) contained three clusters comprised of four wetlands, including 1 control, 2 effect/recovery sites, and 1 clearcut treatment site (total of 12 sites per state forest). Control sites were those with no adjacent forest harvesting during the past 59+ years. Treatment sites included one 59+year area, which was harvested during the winter of 2000-2001 (clearcut treatment), and two effect/recovery sites consisting of wetlands in stands harvested 10-34 (young-age) and 35-58 (mid-age) years before present. Overall, our design included 6 replicate sites within these four age-class treatments, and two ECS subsection levels. Data gathering and analyses associated with this initial phase of the research are well underway. These analyses will assess wetland characteristics and potential changes observed during 2001-2005, the initial period following clear-cutting in adjacent uplands (winter 2000/2001). Here, we report on preliminary analyses of invertebrate-community responses to environmental gradients, including canopy closure, an attribute directly influenced by timber harvest.

OBJECTIVES

- To characterize aquatic invertebrate communities and site-level environmental characteristics (such as stand age-structure) contributing

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to variation in wetland habitats and invertebrates; and

- To comprehensively evaluate initial responses of aquatic invertebrate communities and other wetland features to clear-cut timber harvest.

METHODS

We sampled aquatic invertebrates using surface-associated activity traps (SAT; Hanson et al. 2000) deployed for 24 hr at random locations near the margin of each wetland. Five traps were used concurrently in each wetland. Aquatic macroinvertebrates were sampled during open-water periods, at approximately 3-week intervals during May, June, and July 2002. Water quality was also monitored during May, June, and July using 1-liter surface dip samples collected from the center of each wetland. Water samples were tested for chlorophyll *a*, total phosphorus (TP), and total Kjeldahl nitrogen (TKN) at the Minnesota Department of Agriculture laboratory in St. Paul, MN. We assessed turbidity, water temperature, total alkalinity, and specific conductance in each wetland at least twice during the open water period. Turbidity was measured using a LaMott portable nephelometer. Total alkalinity (TA) was determined by titration (Lind 1979). Specific conductance and dissolved oxygen (DO) were measured on site using YSI portable meters. Upland soil temperatures (Soil Temp) were obtained using a soil thermometer. We assessed extent of average percent canopy closure at 5 locations in each wetland using a Lemmon spherical densiometer (Lemmon 1957).

Resulting data were analyzed using direct gradient analysis. We used partial-redundancy analysis (pRDA), a linear form of direct gradient analysis, to identify relationships between invertebrate community characteristics and physical features, and to partition variance attributable to each significant environmental variable (ter Braak 1995, ter Braak and Smilauer 1998, Jongman et al. 1995). Results presented here are preliminary; interpretations are likely to change as additional data are collected and analyzed.

RESULTS AND DISCUSSION

Results of RDA indicated that invertebrate community structure during 1999-2002 was influenced by a suite of variables. These included duration of ponding (hydroperiod), state forest location, concentrations of dissolved constituents in the wetland water column (alkalinity and specific conductance), soil temperature, and canopy closure above the study wetland (Figure 2). As expected, date of sampling was also important because invertebrate abundance and community structure were dynamic and changed in predictable ways throughout the growing season.

Extent of canopy-closure over study wetlands was an important determinant of invertebrate community structure during all 4 years (1999-2002; Figure 2). This may reflect changing water temperature regimes, reduced litter inputs, or influences of other interactions among canopy characteristics, timber harvest, and wetland communities. It is interesting to note that the relative influence of canopy increased sharply during the first two years following timber harvest (Figure 2). This may reflect direct or indirect influences of clear-cut timber harvest which, obviously, reduced canopy closure over the 6 sites that were harvested during winter 2000-2001.

Hydroperiod showed significant, yet modest influences on invertebrate communities during the 2 years reported here (Figure 2). Batzer et al. (2004) also reported weak associations between wetland invertebrate communities and hydroperiod in small forest wetlands in north central Minnesota. Relationships between hydrology of small depressional wetlands and clear-cut timber harvest are poorly understood in forested landscapes. Some previous research indicates that tree removal has the potential to elevate water tables (Verry 1997, Roy et al. 2000) and modify local hydrology (Roy et al. 2000). Other unanticipated ecological responses to timber harvest are also possible. For example, extending hydroperiods of small forest wetlands may allow vertebrate and invertebrate predators to persist and disrupt natural community dynamics. Hence, other animals including amphibians and early arriving birds and waterfowl, may face added competition for food resources before larger water bodies become ice-free. We expect that subsequent data and analyses should provide better characterization of these wetlands and help clarify relationships between wetland communities and clearcut timber harvest.

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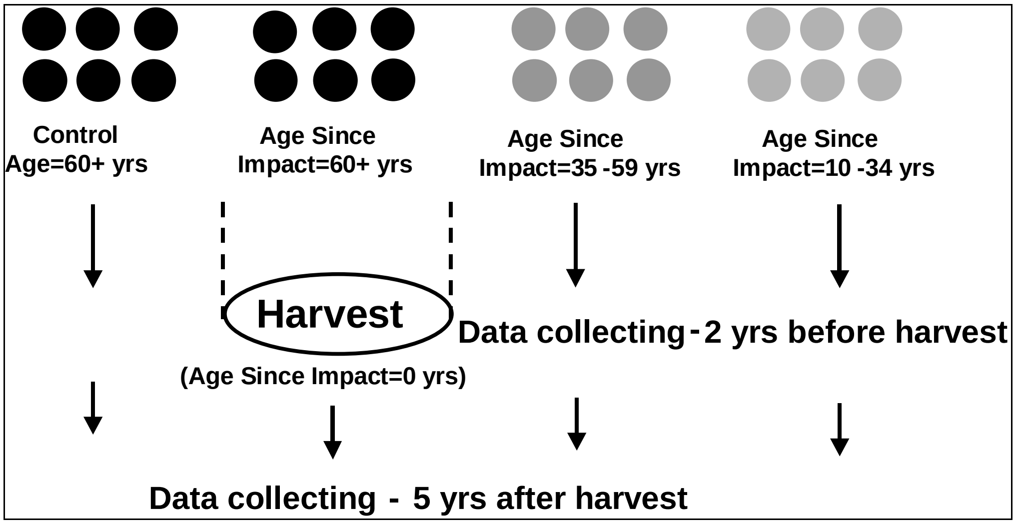

Figure 1. Wetland study design depicting treatment and effect/recovery groups. Phase I includes data collected from first two years of the study. Clear-cut treatment was conducted the winter between the second and third years. Phase II includes sampling efforts for additional three years post-treatment. Study was replicated in a second state forest to detect differences of subsection locality based on the Ecological Classification System (Almendinger and Hanson 1998). Note: The four groups represent the chronology of the adjacent landscape relative to years since last forest harvest.

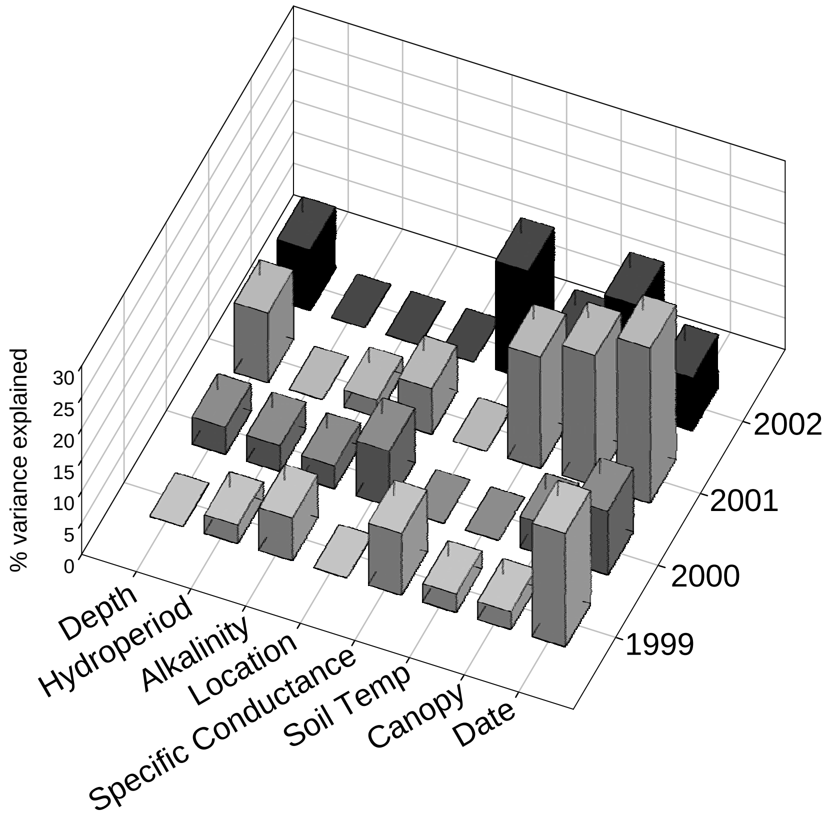

Figure 2. Bars depict percent variance in aquatic invertebrate communities which was explained by environmental variables we measured during 1999-2004. Eight environmental variables included here each explained more variance than expected by chance during at least 2 of these 4 study years.

TESTING THE EFFICACY OF HARVEST BUFFERS ON THE INVERTEBRATE COMMUNITIES IN SEASONAL FOREST WETLANDS

Mark A. Hanson[4], James O. Church[5], Anthony T. Miller[2,6], Brian J. Palik[7], and Malcolm G. Butler[2]

SUMMARY OF FINDINGS

We assessed community-level responses of aquatic invertebrates in small, seasonal forest wetlands to evaluate potential influences of timber harvest and harvest buffers in adjacent uplands. Data gathered during the first 4 years following clear-cut timber harvest (2001-2004) indicated that tree removal produced discernable shifts in aquatic invertebrate communities in adjacent seasonal wetlands. Retention of harvest buffers appeared to partially mitigate against these influences, but benefits of buffers may be limited by windthrow or other factors. Additional site-level research is needed to clarify relationships between physical and ecological characteristics of seasonal wetlands and adjacent silviculture activities, and to better document efficacy and longevity of harvest buffers.

INTRODUCTION

Seasonal wetlands (sensu Stewart and Kantrud 1971) are abundant in forested landscapes and support unique biological communities. Until recently, these sites were often overlooked by forest managers who were largely unaware of their ecological Significance, or potential consequences of silvicultural activities in adjacent uplands. Seasonal wetlands are common in some portions of Minnesota's Laurentian Mixed Forest (Almendinger and Hanson 1998, Palik et al. 2003). Although variable and unique, these wetlands share some distinguishing features. Seasonal wetlands typically occur in localized depressions, and are usually isolated from adjacent waters. In general, these seasonal wetlands fill during spring from snow-melt, and then dry due to evapotranspiration by early-midsummer. However, site-to-site variation in hydrology, soil characteristics, precipitation, wetland size, and other features result in extreme variability in timing and duration of annual flooding (hereafter hydroperiod). An individual wetland basin may remain dry during low-moisture years, yet be flooded year-round during periods when moisture is more abundant (Brooks 2004).

Palik et al. (2001) suggested that processes and organisms in small seasonal wetlands exhibit strong functional linkages to adjacent forested uplands. This is well illustrated by the fact that seasonal wetlands are thought to gain most of their energy from litter originating in adjacent uplands (Oertli 1993). Annual leaf fall is widely considered to be the major energy source for resident organisms. Endogenous primary production from algae growing within seasonal wetlands may also be important, but the magnitude and fluctuation of this contribution to overall productivity is poorly understood.

Seasonal wetlands are also influenced by presence of an adjacent forest canopy. In addition to functioning as a source of organic matter, this canopy mediates light availability at the wetland surface. Canopy closure is a major influence on vegetation dominance in small wetlands, although relationships between light availability, primary production, and major vegetation forms

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are not yet well known. Removal of canopy via timber harvest has potential to influence biological communities in adjacent wetlands owing to increased sunlight, higher water temperatures, and reduced inputs of coarse woody debris and leaf litter.

Aquatic invertebrates are often the most abundant fauna in seasonal wetlands (Brooks 2000), and serve as important links between primary production and vertebrate consumers (Murkin and Batt 1987). Various species of birds, amphibians, and small mammals are known to forage on aquatic invertebrates in seasonal wetlands. Aquatic invertebrate communities in these habitats exhibit life cycles constrained by needs to 1) minimize harmful effects of desiccation, 2) reproduce rapidly, and 3) avoid being eaten by numerous vertebrate and invertebrate predators (Wiggins et al. 1980, Wellborn et al. 1996). In general, invertebrate species richness probably increases with hydroperiod length (Brooks 2004), but this is mitigated somewhat by complex influences of predation (Wellborn et al. 1996, Hanson et al. In Review). More broadly, aquatic invertebrate communities integrate abiotic and biotic features of wetland environments, thus these populations have potential to serve as indicators of wetland characteristics, including changes in functional relationships with adjacent uplands (Adamus 1996, Resh and Jackson 1993). However, invertebrate-based bioassessment techniques applied to wetlands over short time periods may have limited usefulness (Tangen et al. 2003).

Voluntary site-level guidelines have been formulated for timber harvesting adjacent to aquatic habitats (Minnesota Forest Resources Council 1999). These guidelines recommend retention of forested strips or "buffers" adjacent to riparian areas following clear-cut timber harvest near streams, lakes and open-water wetlands, but do not make a similar recommendation for small, seasonally-flooded wetlands. This may be unfortunate given the strength of functional linkages between small wetlands and adjacent upland landscapes, at least at local spatial scales (Palik et al. 2001, Colburn 2001). However, guidelines encourage retention of 5% cover in patches following clear-cut timber harvest, and suggest that these "five percent patches" may be focused adjacent to seasonal wetlands (Minnesota Forest Resources Council 1999). Whether these "five percent buffers" persist (and resist windthrow), or function as expected to preserve ecological integrity of seasonal wetlands is unclear. Finally, some evidence supports the notion that timber harvest modifies natural hydroperiods, at least of some wetland types (Dube and Plamondon 1995, Roy et al. 1997). If this is the case with seasonal wetlands, we expect consequences for resident invertebrate communities whose life-cycle strategies often exhibit narrow tolerances to influences of flooding, desiccation, and predation.

Research reported here was performed in collaboration with investigators from U.S. Forest Service North Central Research Station (NCRS, Grand Rapids, MN), the Natural Resources Research Institute (Duluth, MN), and the University of Minnesota (St. Paul, MN). Collectively, this group has been assessing efficacy of harvest buffer strips on various physical and ecological aspects of seasonal wetlands using study wetlands near Remer, Minnesota. Previously, we reported on pre-harvest variability (2000), sources of variance among aquatic invertebrate communities, and preliminary analyses assessing extent to which harvest buffers mitigate against invertebrate-community change (2001-2003; Hanson et al. 2003). Here, we summarize additional post-harvest results, and evaluate invertebrate community responses to timber harvest (and harvest buffers) based on data gathered during 2002-2004. Our specific component of this larger project has several objectives as indicated below. This is a partial summary. General findings and interpretation may change as a result of additional analyses and interpretation.

OBJECTIVES

- To assess natural variability of resident invertebrate communities;
- To identify and measure sources of variability in major taxa of aquatic invertebrates during the first three years following timber harvest;
- To examine potential responses of wetland invertebrate communities to timber harvest among the four treatment groups by assessing the efficacy of harvest buffers.

METHODS

Study Area

We assessed responses of aquatic invertebrate communities within 16 seasonally-flooded wetlands adjacent to aspen-dominated landscapes in north central MN (near Remer). Study wetlands were located on lands owned and managed by Potlatch, Inc. and Cass County, Minnesota. Wetland study sites were apportioned among four treatments as determined by forest-harvest configurations in adjacent uplands. Each of the four study area blocks included one wetland adjacent to clear-cut, one wetland adjacent to a partial buffer, one wetland adjacent to a full buffer, and one control (unharvested) site (Figure 1). Clear-cut treatments were defined as sites where all trees were harvested to the approximate wetland margin. Wetlands within the partial and full buffer treatments were each surrounded by 50-foot zone. Partial buffers were thinned to approximately 50 percent original basal area, and full buffers remained intact (no harvesting within buffers). No timber harvesting occurred in landscapes adjacent to control wetlands. Each treatment block was replicated four times (Figure 1).

Field and Laboratory methods

Aquatic invertebrate communities were sampled using surface-associated activity traps (SAT's) (Hanson et al. 2000). Samples were collected every two weeks beginning in late-April to early May for three sampling periods, or until the initial wetland drying (sites sometimes flood again during late summer or fall). Five SAT's were randomly deployed in each wetland for approximately 24-hours. Contents of each trap were condensed by passage through funnels fitted with 330-:m mesh, and preserved in 75 percent ethanol. Samples were processed in the laboratory. Invertebrates were identified to the lowest feasible taxonomic level, typically order, family, or genus using keys of Pennak (1989), Thorpe and Covich (1991), and Merritt and Cummins (1996).

Statistical analysis

Study wetlands were considered the units of observation for all our analyses. For each wetland, we summed numbers of invertebrates captured in five SATs to produce site totals of major taxa collected during each biweekly sampling effort. These totals were averaged annually, resulting in estimated mean numbers of organisms sampled per wetland during each study year. Thus, in general, our analyses were based on wetland-year combinations (16 wetlands sampled during 3 years) of major invertebrate taxa. We used indirect (principle components analysis, PCA) gradient analyses to assess community-level variability in aquatic invertebrates of wetland sites, and to relate these observed patterns to gradients induced by buffers and/or timber harvest. All invertebrate data were natural-log transformed (ln+1) prior to gradient analysis to limit influence of extreme values. PCA was performed using PC-ORD version 4.25 (McCune and Mefford 1999).

We used indicator species analysis (ISA; Dufrene and Legendre 1997) to identify relationships between individual invertebrate taxa and silvicultural treatments. ISA is a randomization technique that generates indicator values reflecting both relative abundance and relative frequency of taxa occurring among user-defined treatment groups. Calculated indicator values range from 0-100, and reflect percent agreement of taxa and treatment levels. For example, an indicator value of 100 for species A in treatment I would indicate that species A always occurred in treatment I, but was not found elsewhere. Untransformed invertebrate data were used in our ISA. ISA randomization procedures were based on 5000 permutations and were performed using PC-ORD version 4.25 (McCune and Mefford 1999).

RESULTS

2002 We sampled all 16 study wetlands during weeks of 29 April, 13 May, and 27 May 2002. Many study wetlands dried shortly after we completed gathering the third set of samples.

PCA identified four significant axes, and these accounted for 74.6 % of the variance in aquatic invertebrate communities in our study sites. These four axes respectively explained 29.6, 18.8, 15.0, and 11.2 % of invertebrate community variance. PCA showed modest separation between control and clear-cut treatments along principle component axes one and three (Figures 2 and 3), but not along axis two (Figure 2). Control wetlands tended towards negative (left) scores along axis 1, and wetlands adjacent to clear-cut sites located generally along the positive side of this axis. PCA scores from wetlands adjacent to harvest buffers showed extreme variability, but tended to fall closer to control than to clear-cut treatments (Figures 1 and 2).

Hempitera (true bugs) was the only invertebrate taxon that was significantly associated with any wetland treatment. ISA values for this taxon were 68, 11, 18, and 3 in the clear-cut, partial buffer, full buffer, and control treatments, respectively (Table 1). This group consisted mostly of Corixidae (water boatman) that tended to be more abundant in sites adjacent to clear-cuts. Based on our ISA, no other invertebrate taxa occurred more frequently than expected by chance in any wetland treatment group.

2003 Fifteen of 16 study wetlands were sampled during weeks of 28 April, 11 May, and 27 May 2003. One site (DL4) flooded much later than other study wetlands during 2003, thus data collected there were not used in these analyses.

PCA identified four significant axes, and these respectively explained 28.5, 18.1, 16.1, and 12.8 % of variance in invertebrate communities (total = 75.5 %). Invertebrate community scores again showed modest trends among treatments, with most clear-cut sites falling along the positive (right) side of PCA axis 1, somewhat opposite most control wetlands, which tended toward negative values (left side, Figure 4). Axis two reflected no distinguishable pattern. However, along Axis three, clear-cut wetlands were positively associated, whereas control wetlands tended toward negative values (Figure 5). Again, buffer treatment scores were highly variable, but tended to cluster away from clear-cut sites (Figures 4 and 5).

ISA during 2003 identified fairy shrimp (*Eubranchipus* spp.), leeches (Hirudinea, aquatic bugs (Hempitera), and seed shrimp (Ostracoda) as significant indicators of harvest treatment (Table 2). *Eubranchipus* spp. ISA values were highest in the control treatment sites and declined in full buffer sites, with lowest values from partial buffer and clear-cut wetlands. Hemiptera and Ostracoda reflected an opposite trend, with highest indicator values in clear-cut treatments, and declining ISA scores through the partial buffer, full buffer, and control treatments (Table 2).

2004 As during previous years, three sets of biweekly invertebrate samples were gathered from study wetlands. Again during 2004, one site (DL4) flooded considerably later than others, thus was not considered in this analysis.

PCA identified three significant axes, explaining 40.6, 22.0, and 12.5 % of invertebrate community variance, respectively (total = 75.1 %). These ordinations indicated variability in control sites along axis one (left to right), but reflected considerable separation, thus treatment effects, between control and clear-cut sites along both axes two and three (Figures 6 and 7). As in previous study years, partial- and full-buffer site scores show similarity with other treatments, but it is interesting to note that 3 of 4 full buffer sites clustered near controls (Figure 7). Viewed more broadly, these ordinations appear to reflect consistent ecological differences between wetland sites adjacent to control and clear-cut uplands and also may indicate similarity between full buffer and control sites.

ISA indicated significant associations between several invertebrate taxa and timber harvest treatments. Dragonfly larvae (Odonata), clam shrimp (Conchostraca), and fingernail clams (Sphaeriidae) were captured more frequently in partial-buffer sites than would be expected by chance. Spring tails (Collembola) were significantly more common in samples from control (unharvested) wetland sites (Table 3). Water mites (Hydracarina) were significantly more common and abundant in clear-cut wetlands (Table 3).

DISCUSSION

Invertebrate communities in our study wetlands were highly variable, and were dominated by a modest number of aquatic taxa relative to reports from other regional wetland studies (reviewed by Euliss et al. 1999). Natural dynamics in these populations was such that seasonal fluctuations of invertebrates within individual wetlands sometimes exceeded spatial differences among similar sites on a given date (Hanson et al. 2003).

Our results indicated that clear-cut timber harvest resulted in distinguishable, community-level responses of aquatic invertebrates in adjacent study wetlands during 2002-2004. Only two invertebrate taxa (Hemiptera and *Eubranchipus* spp.) showed consistent associations with specific harvest/buffer treatments. Thus, data patterns we observed may reflect subtle associations among harvest status and buffers among a suite of invertebrates rather than sharp increases or decreases in abundance of a few taxa. Although preliminary, these data may also indicate that harvest buffers have modest potential to conserve integrity of invertebrate communities in adjacent wetlands. We are aware of no other research specifically addressing efficacy of harvest buffers in Minnesota. However, these results support the notion that focusing residual trees (such as the recommended 5% leave trees) adjacent to wetlands following clear-cut timber harvest (Minnesota Forest Resources Council 1999) may help sustain ecological continuity of forest-wetland matrix in the Laurentian Mixed Forest.

In our previous project summary (Hanson et al. 2003), we reported weak overall correspondence between invertebrate communities and environmental variables. Here, we show modest associations between harvest treatments and invertebrate community characteristics from 2002-2004. Lack of stronger associations between invertebrate communities and silviculture activities in adjacent uplands may be due to the fact that these invertebrates show broader environmental tolerances than were measured in our study. This seems especially likely given that many invertebrates in freshwater wetlands are known to be well adapted to survival in ephemeral habitats where severe environmental conditions such as freezing, dessication, etc. are normal (Batzer et al. 2004, Euliss et al. 1999, Wiggins et al. 1980). Our previous analyses also indicated that a large proportion of variance in these invertebrate communities remains unaccounted for by environmental characteristics of wetlands measured in our study (Hanson et al. 2003). The latter may reflect the fact that key environmental variables simply were not included in our analyses.

Presently, we do not understand the ecological basis for observed invertebrate-community associations with buffers and timber harvest. Following timber harvest, we expected seasonal water temperature increases, altered vegetation communities, and reduced leaf litter inputs to our study wetlands. We also expected that these changes might influence invertebrate communities via physical and food-web mediated processes. For example, we noted that loss of wetland ice cover occurred earlier adjacent to clear-cut treatments during spring 2001, the only year in which these observations were gathered. We would expect that earlier ice-out and subsequent warming would modify chronology of some invertebrates, especially taxa with rigid life-cycle requirements such as *Eubranchipus spp*. However, data useful for clarifying these and other influences were not available for our analysis.

Preliminary results of this study support the suggestion of Palik et al. 1999 that seasonal wetlands are functionally linked to the adjacent forest. Our data are also consistent with findings of Batzer et al. (2004) who reported that macroinvertebrates in similar forest wetlands showed little statistical association with environmental variables, including those we measured. Invertebrate communities we studied were highly variable, yet showed modest responses to timber harvest, and perhaps harvest buffers, in the adjacent landscape. Forested buffers appeared to mitigate somewhat against influences of timber harvest, thus we suggest that retention of harvest buffers may be useful for maintaining ecological integrity of seasonal wetlands in the forested landscape. Future research is needed to confirm results reported here and to assess causal mechanisms. Managers would also benefit from future research leading to a better understanding of retention potential of harvest buffers in moist soils (to what extent, and for how long do harvest buffers resist windthrow), and duration of wetland responses induced by adjacent timber harvest.

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Table 1. Indicator and p-values for each of the 16 taxa analyzed in 2002. Indicator values indicate percent perfect indication of treatment based upon the relative abundance and relative frequency. Hemiptera was the only taxon that was found to be significant (p<0.10).

Taxon	Clear-cut	Partial buffer	Full buffer	Control	p-value
Diptera	17	9	61	14	0.76
Odonata	21	11	18	33	0.58
Trichoptera	16	17	7	40	0.45
Hydracarina	28	23	25	25	0.93
Collembola	30	44	6	20	0.46
Eubranchipus spp.	13	4	33	49	0.21
Conchostraca	0	29	25	9	0.81
Hirudinea	6	7	51	36	0.53
Oligochaeta	29	45	18	9	0.30
Coleoptera	27	24	27	22	0.85
Hemiptera	68	11	18	3	0.01
Ostracoda	24	35	17	24	0.73
Cladocera	56	7	18	19	0.50
Copepoda	31	28	33	8	0.62
Gastropoda	65	20	9	5	0.37
Sphaeriidae	6	24	44	24	0.77

Table 2. Indicator and p-values for each of the 16 taxa analyzed in 2003. Indicator values indicate percent perfect indication of treatment based upon the relative abundance and relative frequency. *Eubranchipus* sp., Hirudinea, Hemiptera, Ostracoda were significant (p<0.10) indicator taxa.

Taxon	Clear-cut	Partial buffer	Full buffer	Control	p-value
Diptera	9	10	58	24	0.66
Odonata	20	32	14	23	0.95
Trichoptera	13	4	43	19	0.47
Hydracarina	26	19	29	27	0.95
Collembola	14	11	11	65	0.22
Eubranchipus spp.	7	5	23	61	0.10
Conchostraca	0	60	39	1	0.38
Hirudinea	0	2	90	2	0.04
Oligochaeta	5	42	26	14	0.25
Coleoptera	25	25	27	23	0.80
Hemiptera	51	22	17	9	0.07
Ostracoda	52	24	14	10	0.01
Cladocera	17	3	42	38	0.49
Copepoda	27	44	20	9	0.30
Gastropoda	20	16	36	29	0.89
Sphaeriidae	4	55	11	30	0.41

Table 3. Indicator and p-values for each of the 16 taxa analyzed in 2004. Indicator values indicate percent perfect indication of treatment based upon the relative abundance and relative frequency. Odonata, Hydracarina, Collembola, Conchostraca, and Sphaeriidae were significant ($p<0.10$). *Eubranchipus* spp. and Hemiptera, have p-values of 0.1252 and 0.1092, respectively.

Taxon	Clear-cut	Partial buffer	Full buffer	Control	p-value
Diptera	21	7	17	55	0.28
Odonata	1	60	12	0	0.08
Trichoptera	10	9	21	32	0.74
Hydracarina	52	14	18	16	0.03
Collembola	22	15	15	48	0.10
Eubranchipus spp.	21	0	14	57	0.13
Conchostraca	0	78	20	1	0.07
Hirudinea	5	15	14	8	0.99
Oligochaeta	20	22	34	11	0.54
Coleoptera	27	27	24	22	0.83
Hemiptera	47	26	22	5	0.11
Ostracoda	24	40	23	13	0.39
Cladocera	27	6	17	50	0.14
Copepoda	44	32	19	5	0.26
Gastropoda	32	28	17	12	0.85
Sphaeriidae	1	61	20	17	0.09

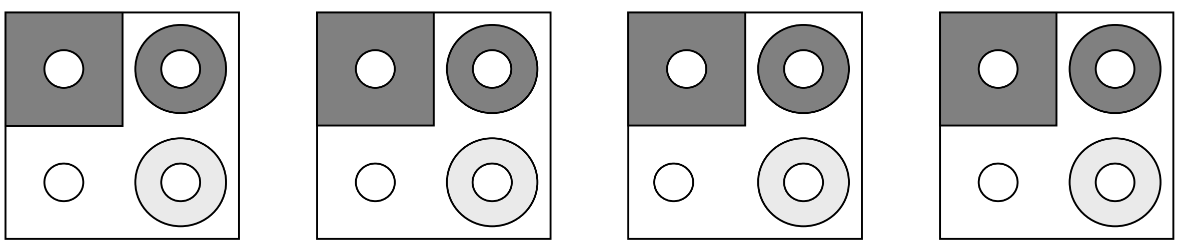

Figure 1. Drawings depict four experimental harvest/buffer configurations. Clockwise from upper left, these were control (no harvest), full buffer (no harvest within 50 feet of study wetlands, thinned buffer (50 percent thinning within buffer), and no buffer (clear-cut to wetland margins). Each group of four "treatments" was replicated in fourlandscape blocks.

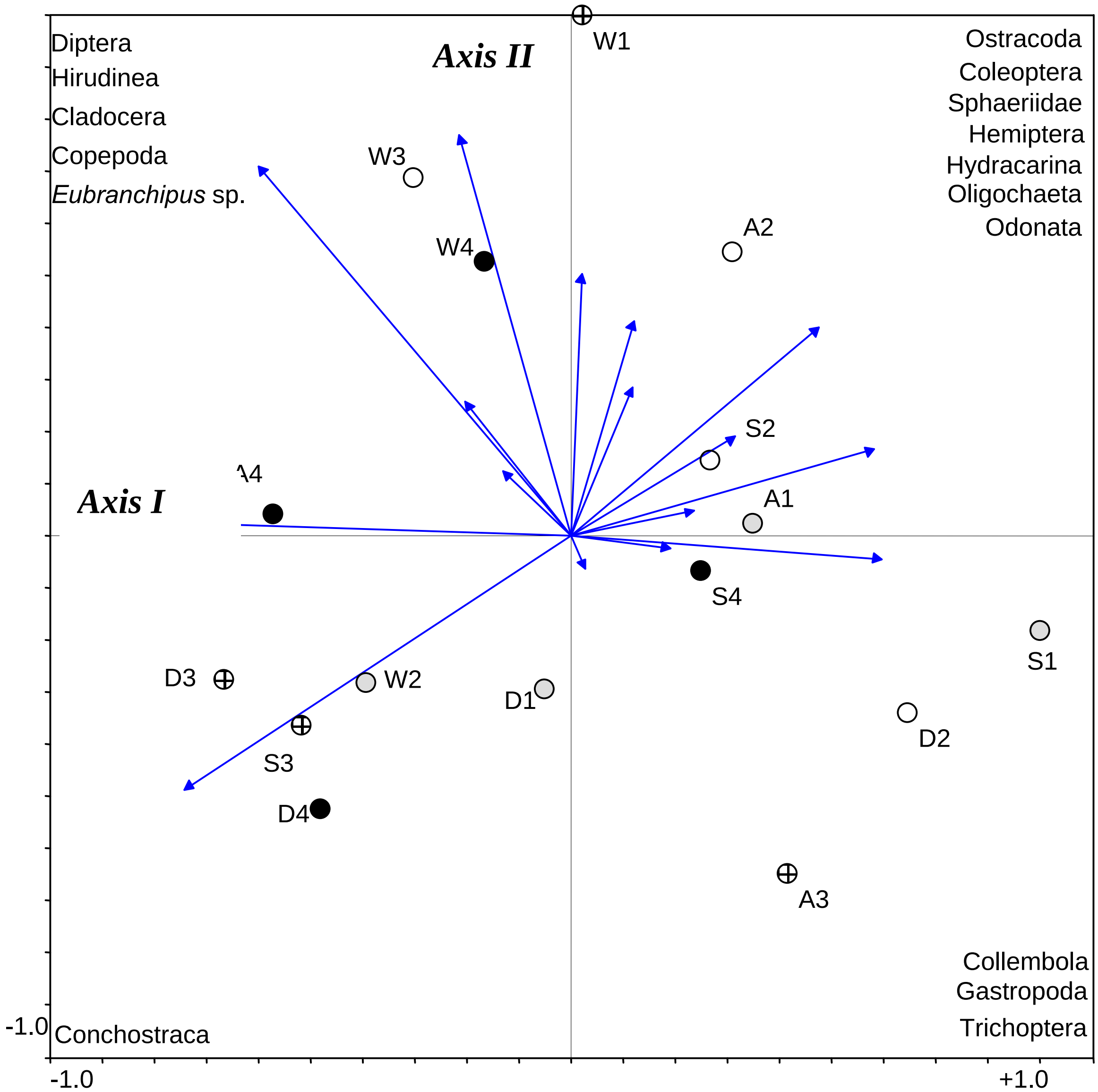

Figure 2. 2002 PCA ordination of sites (circles) and taxa (arrows) on principle component axes one and two. Axes one and two represent 29.6 and 18.8% of the variance, respectively. Black circles indicate control, grid circles indicate full buffer, gray circles indicate partial buffer, and white circles indicate clear-cut treatment. Arrows indicate taxa associations in quadrant in order shown.

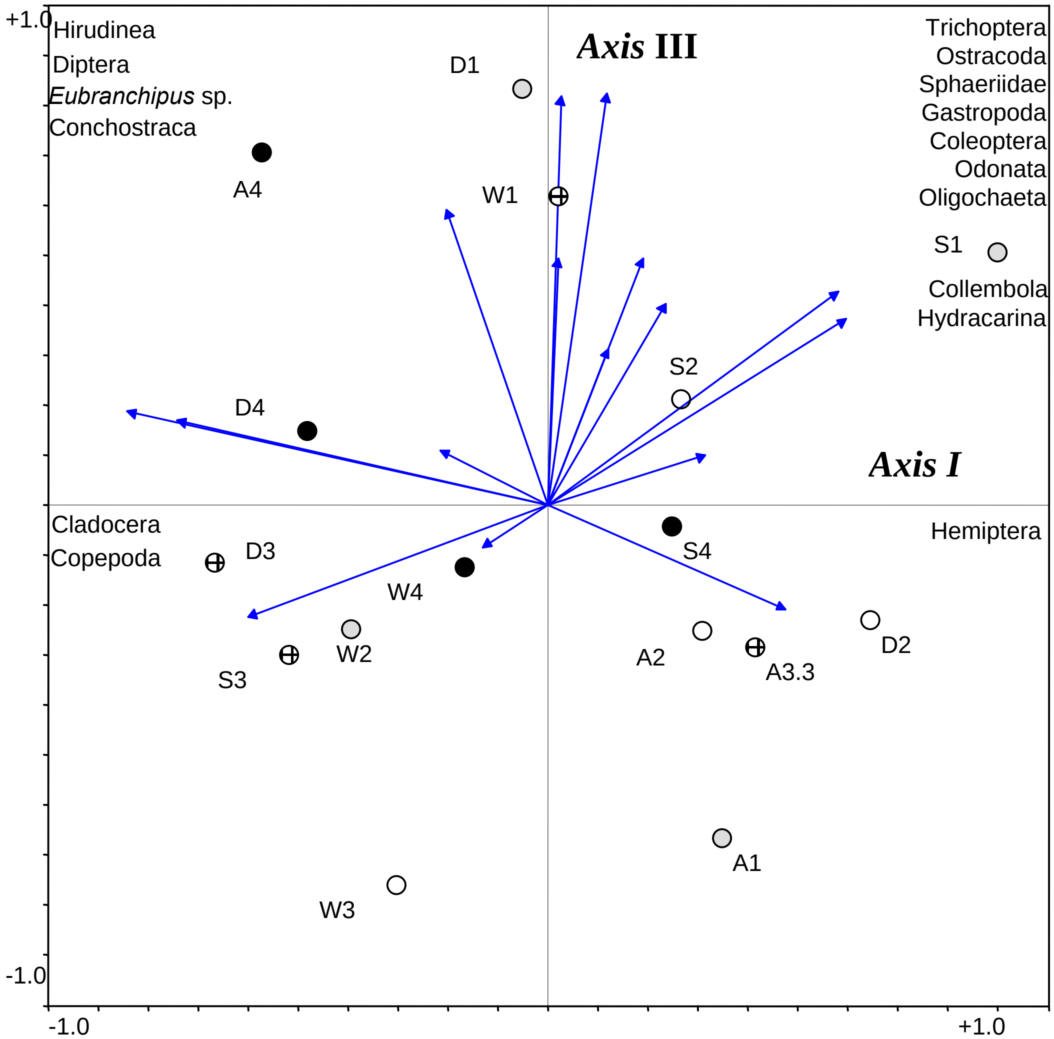

Figure 3. 2002 PCA ordination of sites (circles) and taxa (arrows) on principle component axes one and three. Axes one and three represent 29.6 and 15.0 % of the variance, respectively. Black circles indicate control, grid circles indicate full buffer, gray circles indicate partial buffer, and white circles indicate clear-cut treatment. Arrows indicate taxa associations in quadrant in order shown.

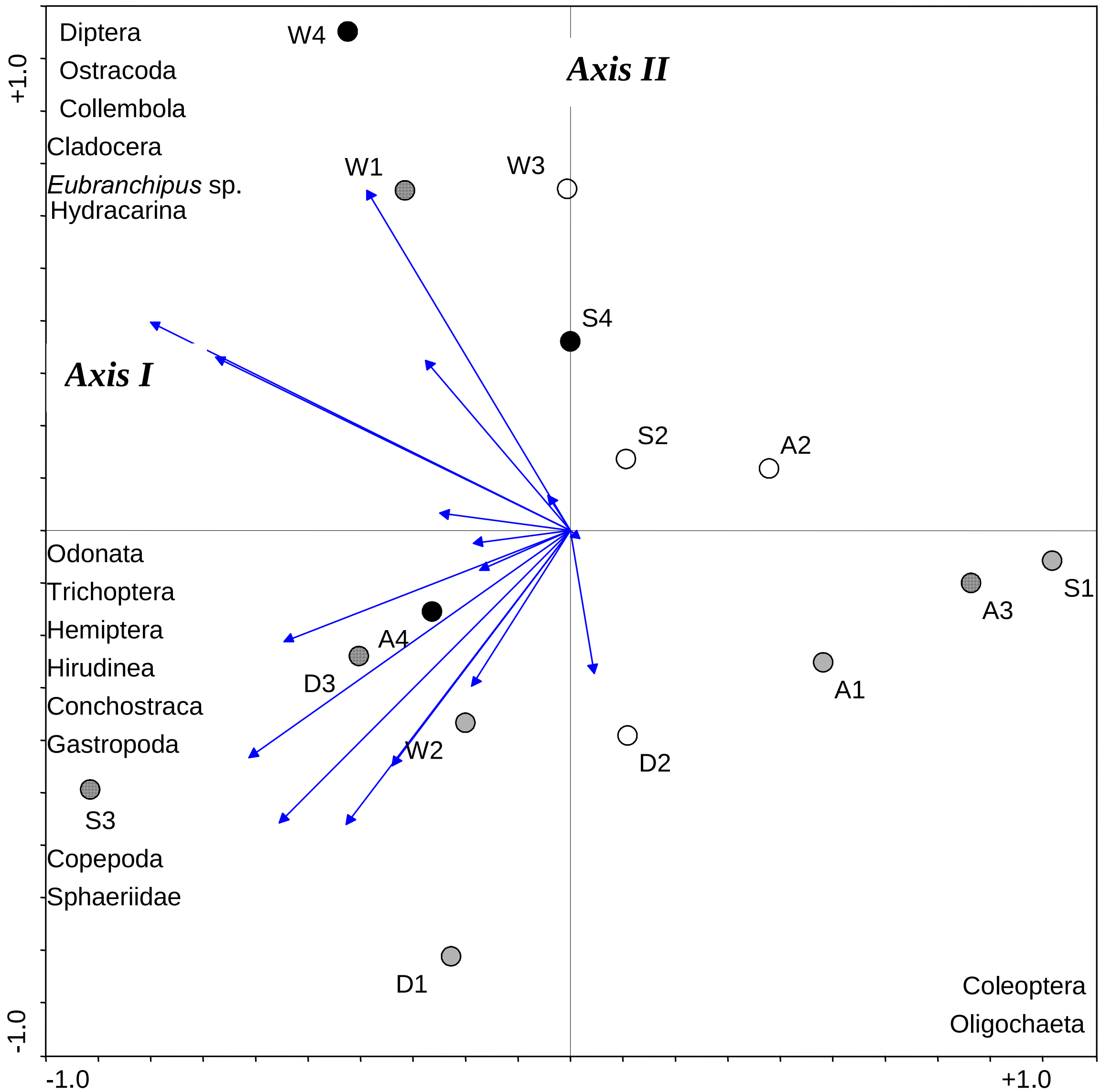

Figure 4. 2003 PCA ordination of sites (circles) and taxa (arrows) on principal component axes one and two. Axes one and two represent 28.5 and 18.1 % of the variance, respectively. Black circles indicate control, grid circles indicate full buffer, gray circles indicate partial buffer, and white circles indicate clear-cut treatment. Arrows indicate taxa associations in quadrant in order shown.

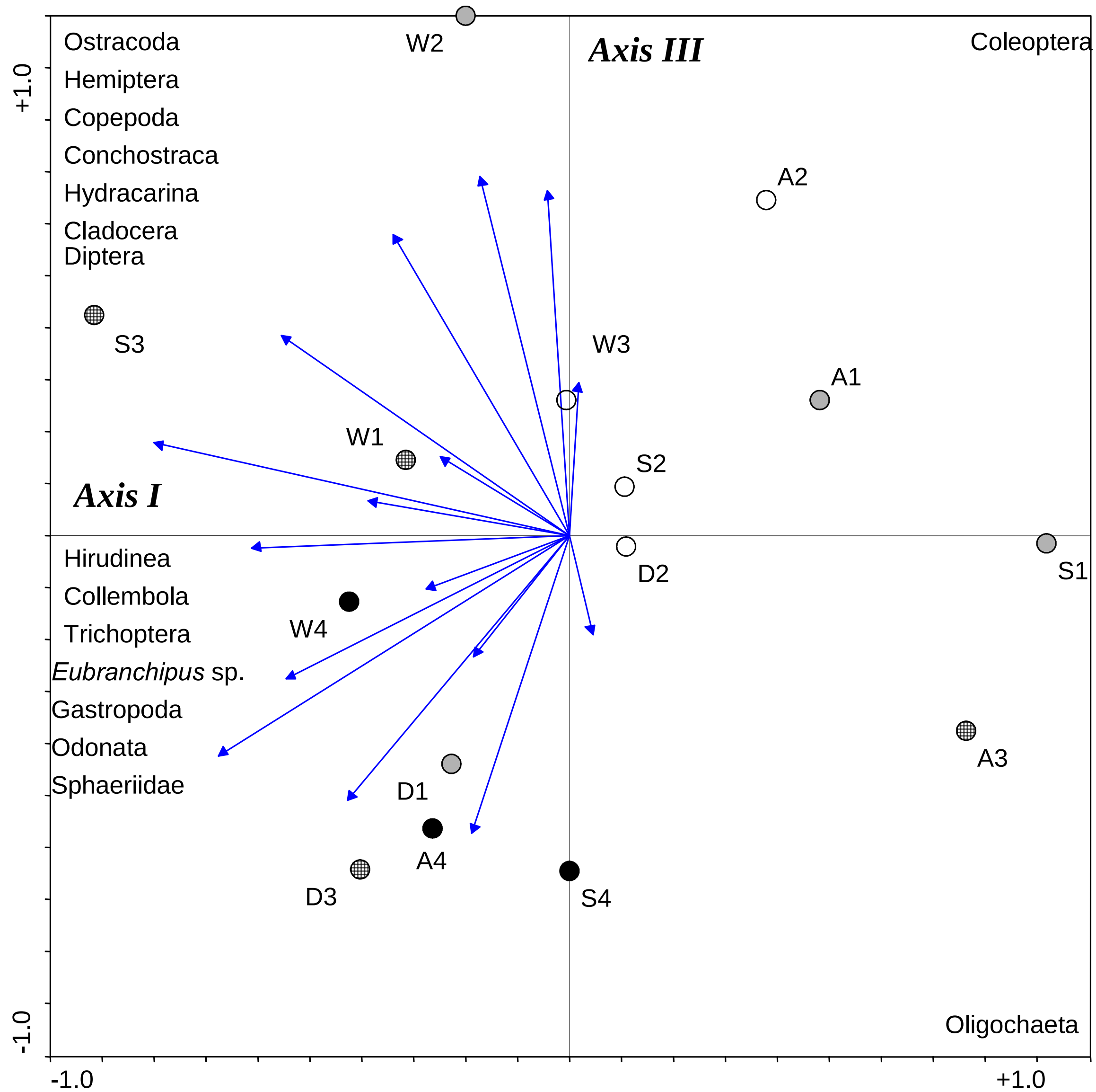

Figure 5. 2003 PCA ordination of sites (circles) and taxa (arrows) on principal component axes one and three. Axes one and three represent 28.5 and 16.1 % of the variance, respectively. Black circles indicate control, grid circles indicate full buffer, gray circles indicate partial buffer, and white circles indicate clear-cut treatment. Arrows indicate taxa associations in quadrant in order shown.

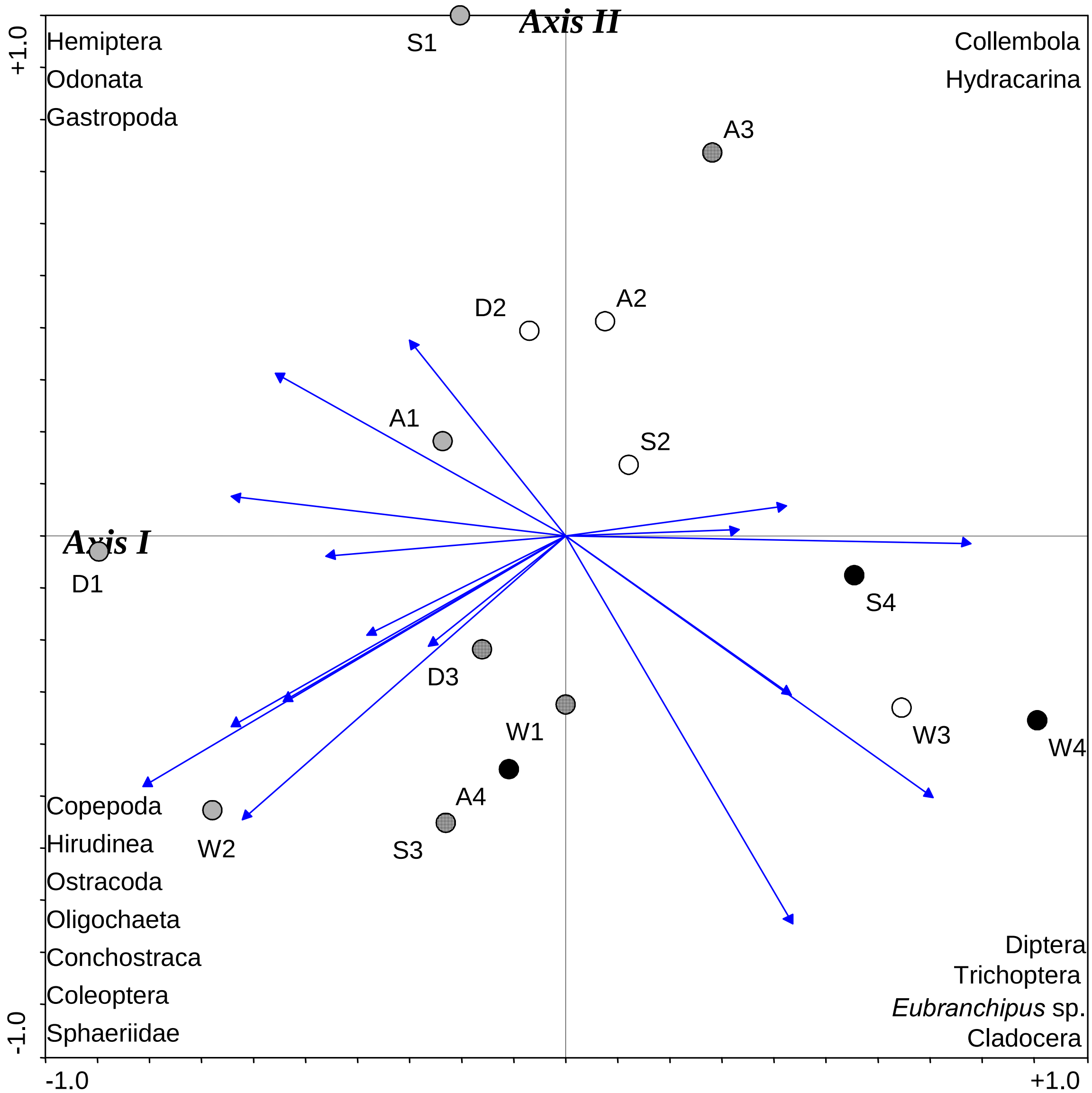

Figure 6. 2004 PCA ordination of sites (circles) and taxa (arrows) on principal component axes one and two. Axes one and two explain 40.6 and 22 % of the variance, respectively. Black circles indicate control, grid circles indicate full buffer, gray circles indicate partial buffer, and white circles indicate clear-cut. Arrows indicate taxa associations in quadrant in order shown.

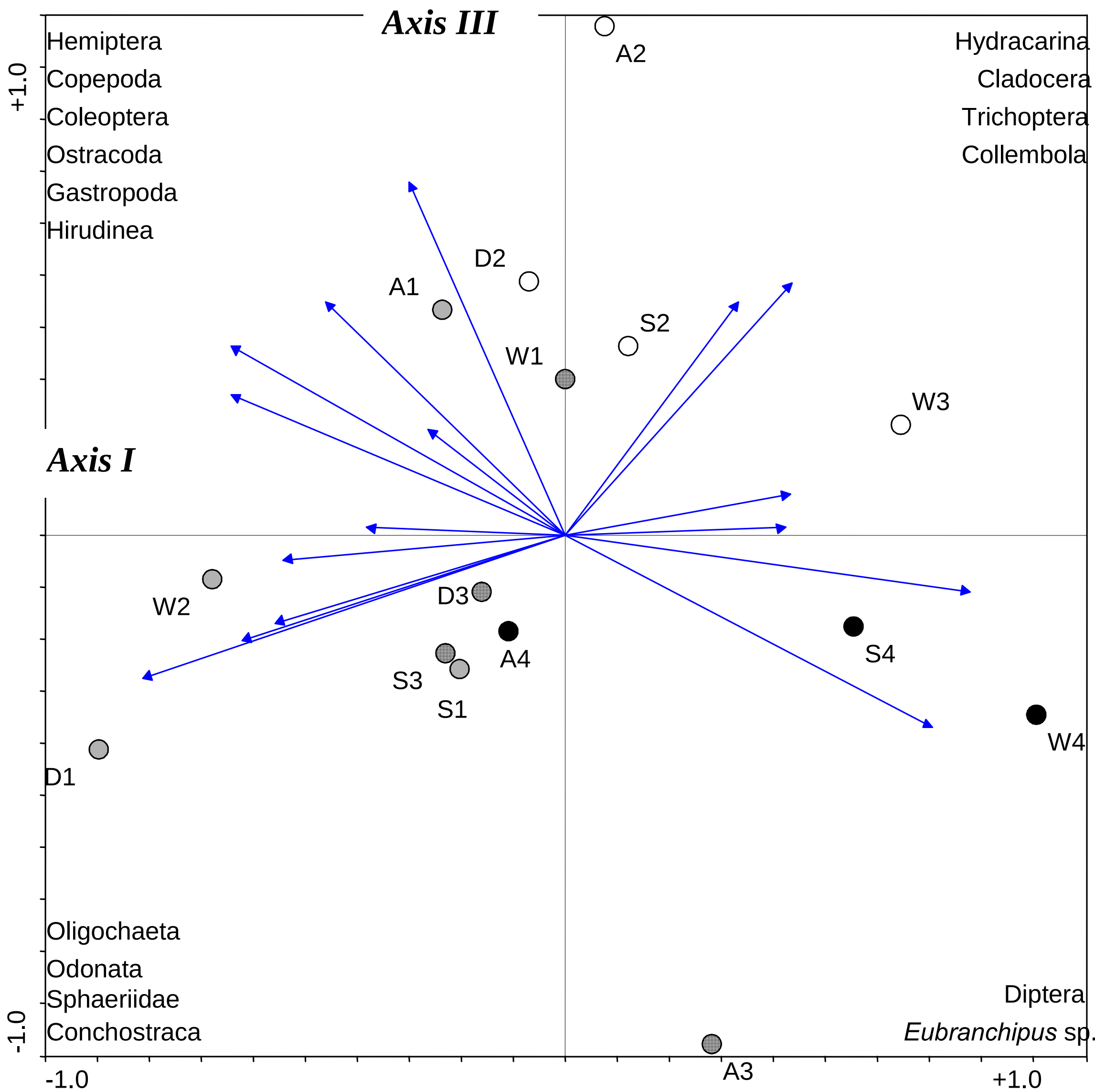

Figure 7. 2004 PCA ordination of sites (circles) and taxa (arrows) on principal component axes one and three. Axes one and three explain 40.6 and 12.5 % of the variance, respectively. Black circles indicate control, grid circles indicate full buffer, gray circles indicate partial buffer, and white circles indicate clear-cut. Arrows indicate taxa associations in quadrant in order shown.

HARVEST PARAMETERS OF URBAN AND RURAL MOURNING DOVES IN OHIO

David P. Scott[8], James B. Berdeen, David L. Otis[9], and R. Lyle Fendrick[10]

***Abstract*:** Few if any studies have examined the influence of a recently implemented hunting season on harvest characteristics of mourning doves (Zenaida macroura). We conducted a reward banding study in Ohio, USA, during 1996–1998 to compare harvest rates in urban and rural areas and to estimate overall harvest rate and band reporting rate. Estimates from band recovery models provided strong evidence for site- and year-specific variation in harvest rates of doves captured at urban and rural sites. Annual harvest rate estimates ranged from 0.006 (95% CI: 0.001 to 0.012) to 0.013 (95% CI: 0.005 to 0.017) for birds captured at urban sites, and from 0.027 (95% CI: 0.016 to 0.038) to 0.056 (95% CI: 0.041 to 0.071) for birds captured at rural sites. The estimated reporting rate of 0.173 (95% CI: 0.108 to 0.239) was less than previously published estimates, probably because of a lack of familiarity of hunters with dove bands. Before hunting was legalized in Ohio, almost 80% of the harvest of banded birds from Ohio occurred in 5 southern states. In our study, > 80% of the harvest of banded birds occurred in Ohio and only 10% occurred in the same southern states. Increased understanding of the role of urban landscapes as potential refuges from hunting pressure will improve our ability to manage dove harvests. Large-scale banding studies are needed to obtain contemporary estimates of harvest parameters, which are necessary for more informed harvest management of mourning doves.

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PUBLICATIONS

Below are lists of scientific reports and other publications by personnel in the Wildlife Populations and Research Unit for the approximate period March 2004 through February 2005. Some titles by Unit personnel pertain to work done while employed by the DNR, but other reports are on work done elsewhere (e.g., as a graduate student, employed by another agency, while on leave of absence, etc.) An (*) before an author's name indicates that the report was listed as in press in previous summaries of research project findings. Included under scientific reports are those which have been published and those accepted for publication (in press). Names in bold indicate a DNR employee.

PUBLICATIONS

Berdeen, J. B., and D. L. Otis**.** The Influence of Subcutaneous Transmitter Implants on Weight Change and Survival of Mourning Doves. *Wildlife Society Bulletin.* In Review.

Brinkman, T.J., C.S. Deperno, J. A. Jenks, **B. S. Haroldson**, and **R. G. Osborn**. 2005. Movement of female white-tailed deer: effects of climate and intensive row-crop agriculture. Journal of Wildlife Management. In press.

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***Carstensen Powell, M., and G. D. DelGiudice**. 2005 Birth, morphological, and blood characteristics of free-ranging white-tailed deer neonates. Journal of Wildlife Diseases. 41:171-183.

Carstensen Powell, M., G. D. DelGiudice, and B. A. Sampson. 2005. Risk of marking-induced abandonment may be minimal for free-ranging white-tailed deer neonates. Wildlife Society Bulletin**.** In press.

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DelGiudice, G. D., **J. Fieberg**, M. R. Riggs, and **M. Carstensen Powell**. 2004. Considering age structure in survival analyses – case study of female white-tailed deer. Proceedings of The Wildlife Society Eleventh Annual Conference, September 18-22, 2004, Calgary, Alberta, Canada.