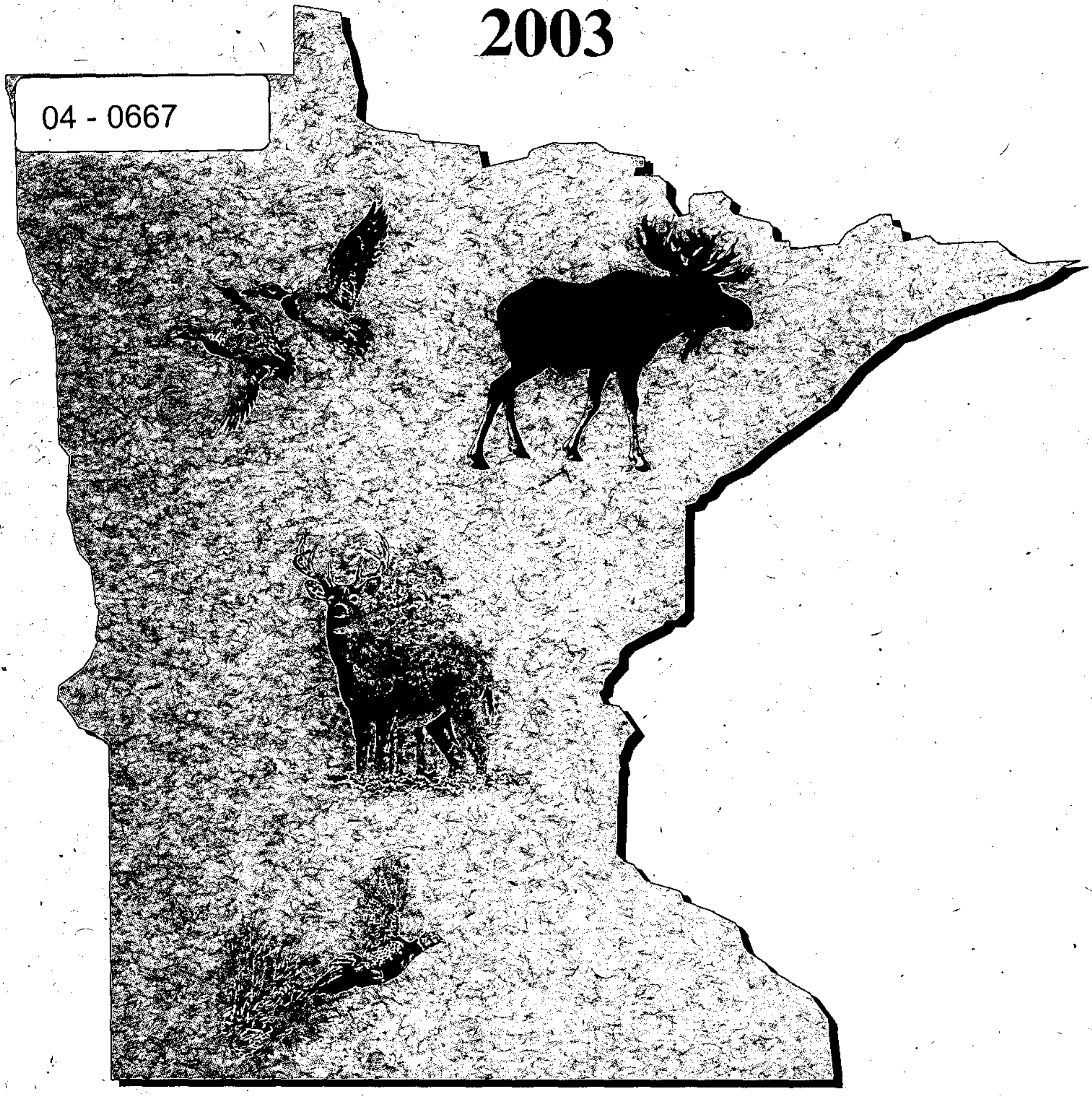
SUMMARIES OF WILDLIFE RESEARCH FINDINGS 2003



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

SUMMARIES OF WILDLIFE RESEARCH FINDINGS 2003

Edited by
Michael W. DonCarlos
Richard O. Kimmel
Jeffrey S. Lawrence
Mark S. Lenarz

Minnesota Department of Natural Resources
Division of Fish and Wildlife
Box 20, 500 Lafayette Road
St. Paul, MN 55155-4020
(651) 297-4964

For more Information contact:
DNR Information Center
500 Lafayette Road
St. Paul, MN 55155-4040
(651) 296-6157 (Metro Area)
1 (888) MINNDNR

TTY (651) 296-5484 (Metro Area)
1 (800) 657-3929
http://www.dnr.state.mn.us

This volume contains interim results of wildlife research projects. Some of the data and interpretations may change as a result of additional findings or future, more comprehensive analysis of the data. Authors should be contacted regarding use of any of their data.

Printed in accordance with Minn. Stat. Sec. 84.03

TABLE OF CONTENTS

Farmland Wildlife

Forest Wildlife

Forest Deer

Moose

River Otter

Wetland Wildlife

Farmland Wildlife Populations and Research Group
Madelia Research Center
35365 – 800th Avenue
Madelia, Minnesota 56062-9744
(507) 642-8478

ESTIMATING DEER POPULATIONS IN SOUTHEAST MINNESOTA

Robert Osborn, Christopher DePerno, and Brian Haroldson

INTRODUCTION

Managers in the Farmland Zone have expressed concern regarding the accuracy of density predictions from the Farmland Deer Model, which is used to predict the number of white-tailed deer (*Odocoileus virginianus*) present on each permit area in the Farmland Zon. Many managers in northwest Minnesota believe the model is underestimating density, while many managers in southeast Minnesota believe the model is overestimating density. Opinions of managers in the rest of the Farmland Zone are varied, inconsistent, and do not reveal any common pattern. Concerns over model results in much of the Farmland Zone can likely be addressed with minor adjustments to the model. The consistent nature of the concerns expressed in northwest and southeast Minnesota are, however, more indicative of potential biases in the model that need to be evaluated.

It is possible that both concerns of underestimation in the northwest and overestimation in the southeast are valid. The Farmland Model is an accounting-type model, which simply adds births and subtracts deaths from a predetermined population estimate. All inputs in the Farmland Model are basic biological data that can be evaluated with research. However, the Farmland Model has never been adequately calibrated and the Minnesota Department of Natural Resources currently conducts no deer population surveys to provide an independent estimate of population size.

Grund (2001) evaluated an accounting-type model that incorporated aspects of both the Forest Zone and Farmland Zone deer models. This study, conducted in the Mille Lacs Wildlife Management Area, used sensitivity analysis to identify the most influential model inputs and evaluated model accuracy by comparing its predictions against independent population estimations. Briefly, Grund (2001) concluded that female reproductive and survival rates most influenced population predictions. Moreover, because of their higher reproductive potential, adult females were more influential than juvenile females. Also, Grund (2001) stated the population predictions of accounting-type models are subject to drift and that the amount of error increased over time. Model drift was the result of errors accumulating through time and margins of error became unacceptably large within approximately 4 years (Grund 2001). This assumes, of course, the population estimate used to initiate the model was accurate. Consequently, Grund (2001) recommended that accounting-type models be recalibrated every 4-5 years.

Accuracy of deer population estimates generated by the Farmland Model are influenced by many variables; 2, however, are of immediate concern and will be addressed by this study. First, estimates of the starting populations used to initiate the model within each permit area were not derived independently, but rather were obtained using another model (the predecessor to the current model). Consequently, it is not known whether the initial population estimates were accurate. Second, the Farmland Model has not been checked for drift for 11 years, which is well beyond the critical period suggested by Grund's (2001) research.

All additional inputs of the Farmland Model are testable with research. Some, such as adult female and fawn mortality, are currently being evaluated (DePerno et al. 2003). Others will be tested as time, money, and other logistical constraints allow. However, the best biological data will not produce an accurate population estimate if the estimated starting population is incorrect. The goals of this project were to provide independent population estimates of white-tailed deer in 3 southeast Minnesota permit areas and to determine the most accurate and cost-effective sampling design for future deer surveys. Results also were used to evaluate the accuracy of the Farmland deer model.

METHODS

Permit Area Selection

Because of financial and logistical constraints, it was not possible to survey all permit areas within the Big Woods Southeast Deer Management Sub-unit (BWSE; Permit areas 341 – 349). Therefore we selected a subset of permit areas that were representative of the breadth of habitat conditions in the BWSE. We used level 2 MN-GAP data to identify cover classes (non-vegetated, crop/grasslands, shrublands, aquatic, upland coniferous forest, lowland coniferous forest, upland deciduous forest, lowland deciduous forest, upland coniferous/deciduous mix, lowland coniferous/deciduous mix) and ArcView (ESRI, Redlands, CA) to calculate land area by cover class within each permit area (Appendix A).

Cluster analysis (Johnson and Wichern 1992) was used to examine habitat characteristics of the BWSE permits areas and identify natural groupings. Lowland coniferous forests and lowland coniferous/deciduous mixed forests are not found in the BWSE, thus 8 cover classes were used for data analysis. Three groupings were identified (Figure 1) and 1 permit area from each grouping was selected. To minimize double counting of animals due to animal movement between permit areas during the survey, adjoining permit areas were not selected. Thus, based upon habitat characteristics and proximity, permit areas 341, 346, and 347 were selected for the survey.

Survey Methodology

Deer populations in each permit area were estimated using helicopter quadrat surveys. Quadrat surveys have been used successfully to estimate populations of caribou (*Rangifer tarandus*; Sniff and Skoog 1964), moose (*Alces alces*; Evans et al. 1966), and mule deer (*O. heimonus*; Bartmenn et al. 1986) in a variety of habitat types. Each permit area was divided into 1 square mile quadrats (sections from Land Survey data) and a subsample of 20-30% of these quadrats was surveyed. A systematic random sampling design using a square grid (Cressie 1993, D'Orazio 2003) was employed in each permit area. This design ensured entire coverage of the permit area and minimized potential problems associated with animal movement between quadrats (i.e., double counting) and animals being scared off a quadrat prior to the start of the survey. Systematic designs are also typically easier to implement and often result in estimates that are more precise than those obtained using simple or stratified random sampling designs (Cressie 1993, D'Orazio 2003). Surveys were conducted during winter when deciduous vegetation had dropped its leaves and when deep (approximately 6-8 inches) snow cover was anticipated to last for several days. This improved visibility and ensured that enough time was available to allow the survey to be completed. Quadrats were flown until observers were confident they had seen all deer within each quadrat. Density estimates were be calculated using the formulas of D'Orazio (2003).

RESULTS

Surveys were conducted in permit areas 341 and 346 between 13 February 2004 and 26 February 2004. At the start of the survey, snow conditions were acceptable (approximately 6 inches of snow). Because no new snow was received during the survey period, snow conditions became unacceptable before permit area 347 could be surveyed.

Between 16% and 17% of each permit area was surveyed. The number of deer seen per square mile (recall that each quadrat was 1 square mile in size) in permit area 341 ranged between 0 and 116 and averaged 7.9 deer (Table 1). The number of deer seen per square mile in permit area 346 ranged between 0 and 62 and averaged 24.2 deer (Table 2). Visibility bias was not measured. Consequently, these values represent a minimum estimate.

DISCUSSION & MANAGEMENT IMPLICATIONS

Data from this study provides, for the first time, independent estimates of population size that can be used to check the predictions of the Farmland Deer Model. Like all accounting style models, the Farmland Deer Model is sensitive to the starting population used to initialize the model. This study provides a benchmark which can be used to evaluate and if necessary recalibrate the Farmland Deer Model.

Prior to the survey, the Farmland Deer Model for permit area 341 was predicting approximately 12 deer per square mile (a 50% overestimate from unadjusted survey data), while the model for permit area 346 predicted approximately 20 deer per square mile (a 20% underestimate). These data suggest the issue of model accuracy is more complicated than simply over- or underestimation within a broad geographic area. It is possible, however, that biases in the predictions of the Farmland Deer Model are similar in permit areas with similar habitat composition. This hypothesis will be evaluated with additional surveys which are to be conducted in southeastern Minnesota during the winter of 2004-2005.

This study also provides a data set, which via *post-hoc* analysis, can be used to evaluate the effectiveness of survey protocols. For example, a plot of the number of deer seen on each quadrat verses the percentage of woody cover in that quadrat (Appendix B) shows that very few deer are observed on quadrats that have less than 10% woody cover. This information may provide useful criteria which can be used to classify each quadrat as either a low or a high deer density area. Under these circumstances, a stratified random experimental design would likely be more effective. This hypothesis also will be evaluated with additional surveys during the winter of 2004-2005.

Assuming data from future surveys continue to indicate it is possible to obtain accurate population estimates at reasonable cost, surveys will be conducted in other portions of the Farmland Zone. Using an approach as outlined above, it may be possible to divide the Farmland Zone into smaller regions and select a small number of representative permit areas in each region that could be surveyed on a 4-5 year rotation. With the assumption that permit areas with similar habitat composition have similar deer densities, this would allow for the Farmland Deer Model to be recalibrated on a 4-5 year basis as suggested by Grund (2001).

Finally, over the next few years, the Minnesota DNR will be examining deer population goals. The current population goals are several years old, may no longer reflect the biological or social carrying capacities of permit areas, and need to be re-evaluated. Data from this study and future surveys will provide valuable information regarding the performance of the Farmland Deer Model, will provide empirical data necessary to justify to the public the need to change population goals, and will allow the Minnesota DNR and the public to have greater confidence in the decisions made during the goal revision process.

LITERATURE CITED:

Bartmann, R. M., L. H. Carpenter, R. A. Garrott, and D. C. Bowden. 1986. Accuracy of helicopter counts of mule deer in pinyon-juniper woodland. Wildlife Society Bulletin 14:356-363.

Cressie, N. A. C. 1993. Statistics for spatial data. Second edition. Wiley, New York, New York, USA.

D'Orazio, M. 2003. Estimating the variance of the sample mean in two-dimensional systematic sampling. Journal of Agricultural, Biological, and Environmental Statistics 8:280-295.

DePerno, C. S., B. S. Haroldson, T. J. Brinkman, B. J. Bigalke, C. C. Swanson, I. L. Lajoie, J. A. Jenks, J. D. Erb, and R. G. Osborn. 2003. Survival and home ranges of white-tailed deer in southern Minnesota. Pages 35-54 in M. DonCarlos, R. Kimmel, J. Lawrence, and M. Lenarz, editors. Summaries of Wildlife Research Findings, 2002. Division of Wildlife, Minnesota Department of Natural Resources, St. Paul Minnesota, USA.

Evans, C. D., W. A. Troyer, and C. J. Lensink. 1966. Aerial census of moose by quadrat sampling units. Journal of Wildlife Management 30:767-776.

Grund, M. D. 2001. Options for monitoring white-tailed deer populations in Minnesota. Ph.D. Dissertation, Southern Illinois University, Carbondale, Illinois, USA.

Hayek, L. C. and M. A. Buzas. 1997. Surveying natural populations. Columbia University Press, New York, New York, USA.

Johnson, R. A. and D. W. Wichern. 1992. Applied multivariate statistical analysis. Third edition. Prentice Hall, Upper Saddle River, New Jersey, USA.

Sniff, D. B. and R. O Skoog. 1964. Aerial censusing of caribou using stratified random sampling. Journal of Wildlife Management 28: 391-401.

Table 1. Summary statistics for the deer survey conducted in permit area 341, Minnesota, February 2004.

Variable	Value
n	102
Minimum	0
Maximum	116
Mean	7.9
Standard Deviation	11.8
90% CI	6.0 - 9.9

Table 2. Summary statistics for the deer survey conducted in permit area 346, Minnesota, February 2004.

Variable	Value
n	50
Minimum	0
Maximum	62
Mean	24.2
Standard Deviation	14.4
90% CI	20.8 - 27.7

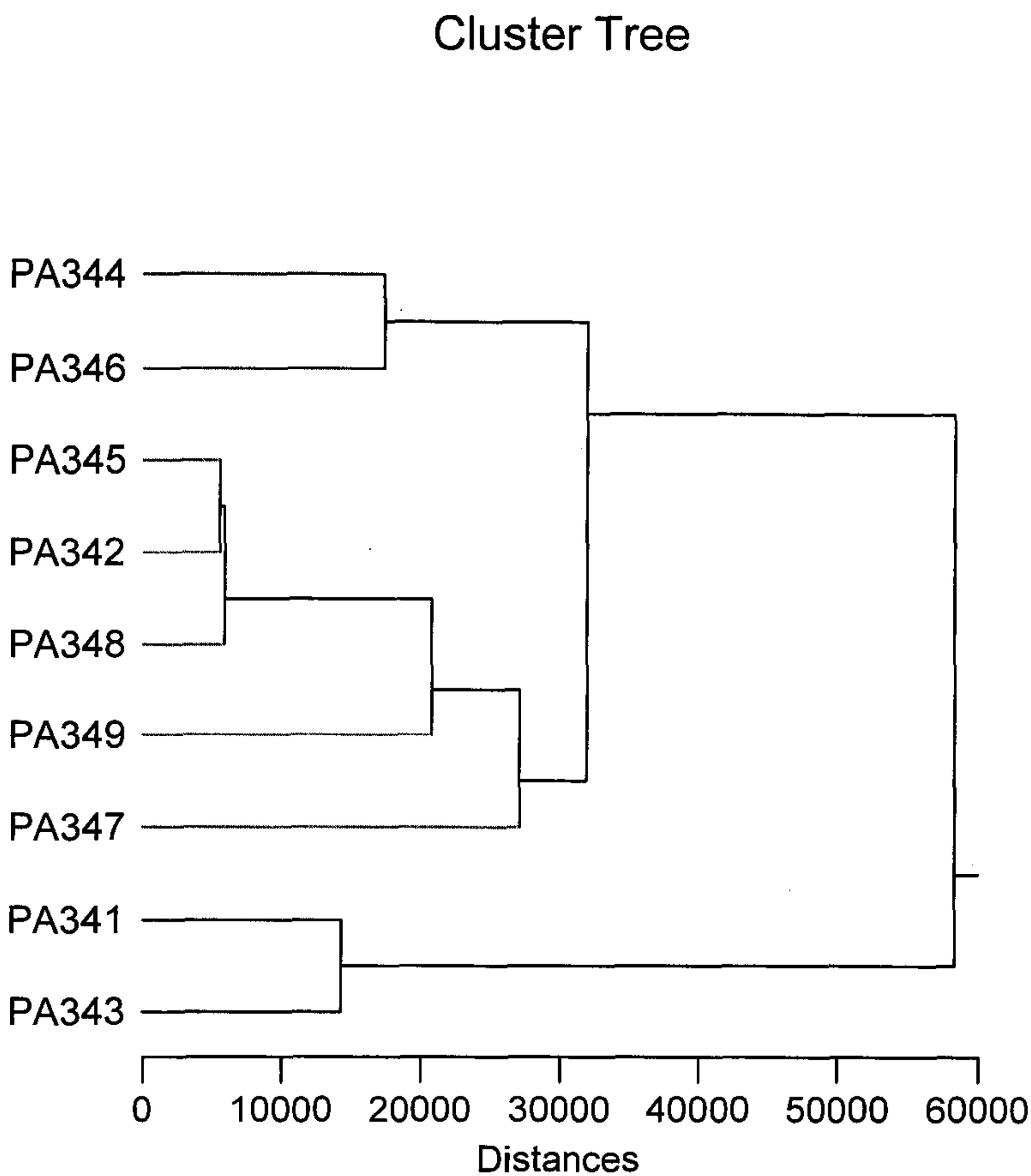


Figure 1. Cluster tree diagram of permit areas within the Big Woods Southeast Deer Management Sub-unit based upon habitat characteristics. Distances were calculated using the average linkage method.

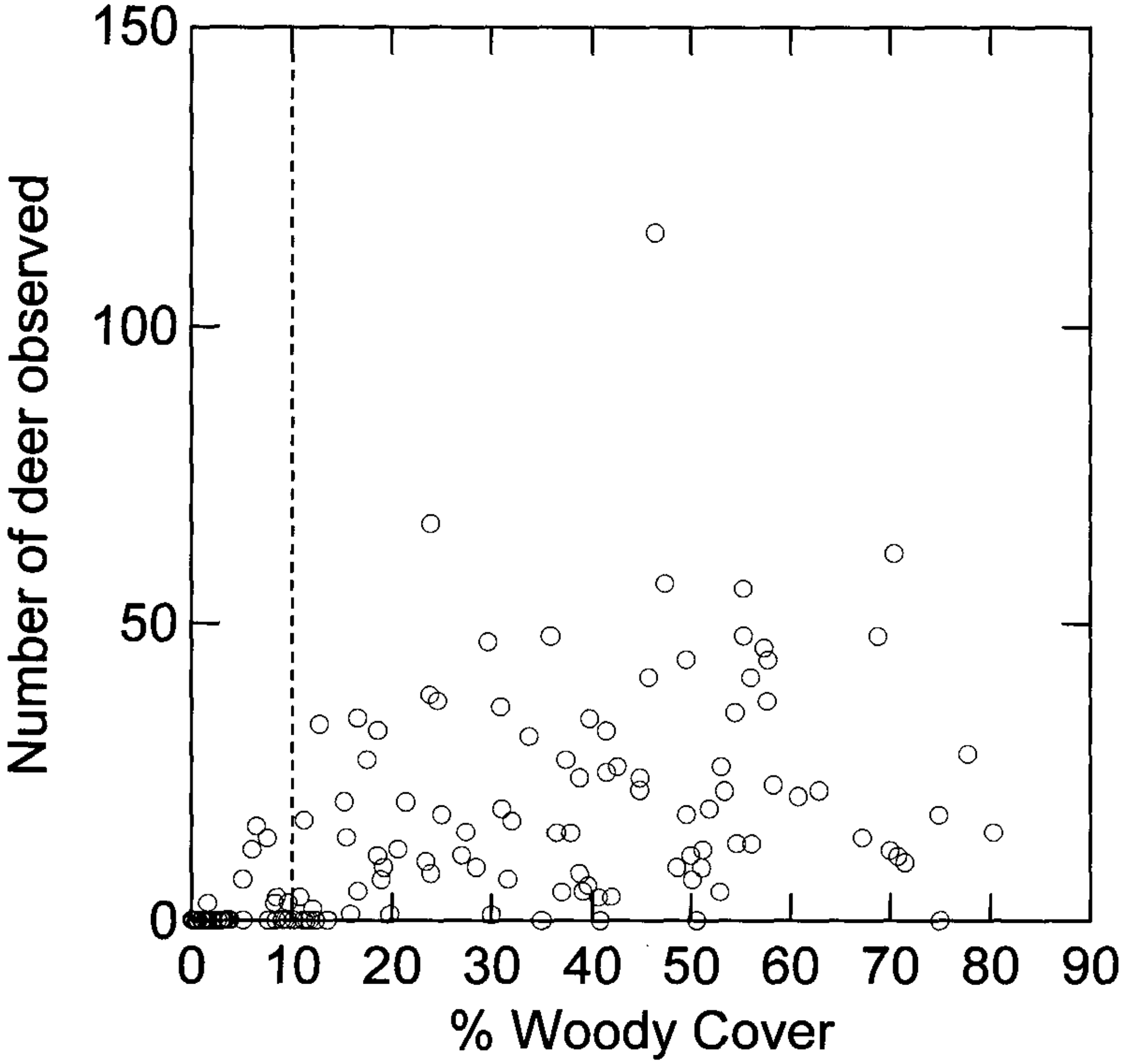
APPENDIX A

Appendix A. Acres of land in each cover class for permit areas in the Big Woods Southeast Deer Management Sub-unit.

Cover Class	Big Woods Southeast Deer Management Sub-unit Permit Area									Avg
	341	342	343	344	345	346	347	348	349	
Non-vegetated	10,759	4,531	24,704	1,254	6,881	6,649	3,782	3,135	3,972	7,296
Crop/grass	318,781	163,072	357,692	80,119	148,734	99,657	244,345	165,310	197,381	197,232
Shrubland	4,093	3,790	2,857	1,761	1,948	5,185	1,184	1,370	4,632	2,980
Aquatic	11,865	16,699	2,676	989	7,073	8,279	526	918	7,572	6,289
Upland Coniferous Forest	402	766	372	472	250	322	77	699	988	483
Lowland Coniferous Forest	0	0	0	0	0	0	0	0	0	0
Upland Deciduous Forest	43,644	40,127	28,196	30,294	39,952	80,668	24,935	33,487	89,987	45,699
Lowland Deciduous Forest	10,261	9,381	8,066	5,451	8,076	8,119	3,265	5,577	13,227	7,980
Upland Mixed Forest	360	377	120	613	359	774	174	1,962	1,468	690
Lowland Mixed Forest	0	0	0	0	0	0	0	0	0	0
Unclassified	292	34	0	0	2	37	41	3	26	48
Total	400,798	238,777	424,683	120,953	213,275	209,690	278,329	212,461	319,253	

APPENDIX B

Appendix B. Scatter plot of number of deer observed in each quadrat verses the percentage of woody cover on quadrats flown during the southeast Minnesota deer survey, February 2004.



FARMLAND WHITE-TAILED DEER RESEARCH NEEDS SURVEY – PRELIMINARY REPORT

Brian Haroldson

During 2003, the Farmland Wildlife Research Group of the Minnesota Department of Natural Resources (DNR) began a planning effort to identify and prioritize data needs for improving management of white-tailed deer (*Odocoileus virginianus*) in Minnesota's agricultural landscape. Natural resource managers from state (DNR Wildlife [n=72], Parks [n=40], Enforcement [n=4]) and federal (National Wildlife Refuge [n=6]) agencies with management responsibility in Minnesota's farmland were surveyed via e-mail on 15 March 2004 to assess their data needs. The survey consisted of 9 questions about deer population management, clientele management, deer population modeling, hunting season structure, alternative harvest strategies, management challenges, and primary data needs. This report is a summary of the survey responses of DNR wildlife staff only. After 2 mailings and a telephone reminder, 34 (47%) surveys were completed and returned. Responses were received from all but one DNR area wildlife office in the farmland zone.

To more effectively manage deer populations, managers reported a high priority need for information on deer abundance (91%), population models (59%), reproduction/recruitment (44%), and adult female survival (42%). In contrast, information on food habits (85%), predation (movements [82%], survival [71%], cause-specific mortality [68%], food habits [65%] of predators), habitat requirements (50%), and movements of deer (47-56%) were rated as low priority (Table 1).

For clientele management, respondents indicated that information on landowner (53%) and hunter (50%) attitudes/interests was a high priority need. Only 9% of respondents rated non-hunter attitudes/interests as high priority, while 41% rated them as low priority.

Nearly half (48%) of the managers indicated that the current hunting season structure (season length, bag limits, antlerless permit allocation, etc.) is effective at accomplishing antlerless deer harvest goals in their work area. Twenty-seven percent did not know and 24% reported that the season structure was not effective. Reasons given by this last group included failure of hunters to take advantage of opportunities to kill multiple antlerless deer (i.e., inadequate harvest of antlerless deer), inadequate number of hunters, and lack of access to land.

A majority of respondents agreed that alternative harvest strategies should be evaluated for effectiveness in increasing antlerless deer harvest in areas that needed additional harvest. These strategies included late-season antlerless hunts (70%), issuing buck licenses via lottery (64%), early-season antlerless hunts (58%), and earn-a-buck regulations (55%). However, there was little support (16%) for evaluating the impact of antler point restrictions on antlerless harvest.

Survey recipients were asked to identify challenges for deer management during the next decade. The most frequently cited challenges were special interest deer hunting regulations (e.g. Quality Deer Management, or QDM), maintaining deer populations within established goals, hunter access, rural and urban development, and depredation.

When asked to describe their primary data needs to improve deer management in the farmland zone, the 32 respondents identified 124 different items. The top data needs included independent deer population estimates/trend indicators, accurate and timely harvest data, evaluation of various season structures on deer harvest, deer population demographics, clientele surveys, and accurate data inputs for the population model (Table 2).

A comprehensive report summarizing responses of all survey recipients (DNR Wildlife, Parks, Enforcement, and National Wildlife Refuge Biologists) will be provided under separate cover. As a result of this survey, data needs will be prioritized to help establish future research direction.

Table 1. Ranking of population data needs of DNR wildlife staff to more effectively manage white-tailed deer populations in Minnesota's farmland (from deer research needs survey, Mar-May 2004).

Information needed	Priority
Animal abundance (# deer/permit area)	High
Population models	
Survival	
Reproduction/recruitment	
Habitat/population relationships	
Cause-specific mortality	Medium
Animal distribution	
Movement/home range	
Habitat requirements	
Predation (food habits, cause-specific mortality, survival, movement of predators)	
Food habits (diets)	Low

Table 2. Primary data needs of DNR wildlife staff to improve management of white-tailed deer in Minnesota's farmland (from deer research needs survey, Mar-May 2004).

Data need	n
Independent population estimate / trend indicator	22
Accurate and timely harvest data	12
• harvest trends	
• harvest rates on public vs. private land	
• antlerless harvest by non-permit holders	
• registration compliance	
Evaluation of season structure on harvest	9
• alternative harvest strategies	
• season timing (rut vs. non-rut)	
• Zone 3 evaluation	
Deer population demographics	8
• buck:doe ratio	
• age:sex ratio	
• antler quality data	
Hunter attitudes	8
Landowner attitudes	7
Accurate data inputs for population model	7
Reproduction / recruitment	7
Cause-specific mortality	6
Hunter demographics	6
• # hunters/permit area	
• # permit applicants/permit area	
• recruitment of hunters (archery, muzzleloader) via all-season license	
Movement / home range	5
Valid population goals / estimation of carrying capacity	4
Survival	3
Predation	3
Habitat / population relationships	3
Improved management of deer in refuge areas (parks, land closed to hunting)	3
Deer distribution	2
Depredation abatement strategies	2
Other	
• Attitudes of DNR policy makers towards deer management issues	
• Rate of suburban development and eventual level of change to the landscape	
• Ample data to support management decisions	
• Projected commodity crop make-up / prices / federal farm programs	
• Impact of vehicle kills on urban deer populations	
• Elk / deer interaction	
• Quantify land closed to hunting	

MINNESOTA DEPARTMENT OF NATURAL RESOURCES CWD SURVEILLANCE PROGRAM 2003

Jeannine Tardiff and Michael DonCarlos

INTRODUCTION

Chronic Wasting Disease (CWD) is a transmissible spongiform encephalopathy (TSE) that affects elk (*Cervus elaphus*), mule deer (*Odocoileus hemionus*), and white-tailed deer (*Odocoileus virginianus*) (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). An infectious protein or "prion" is believed to be the etiological agent of CWD. 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.

Chronic Wasting Disease was first recognized in 1967 by researchers studying captive mule deer and in 1978 in captive white-tailed deer and elk (Williams and Young 1980). CWD has been diagnosed in captive cervid populations from Nebraska, Oklahoma, Kansas, Montana, Colorado, Wyoming, South Dakota, Wisconsin, 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). However, recently CWD has been detected west of the continental divide in Colorado and within wild deer populations of Nebraska, Wisconsin, Illinois, South Dakota, Utah, and New Mexico (Chronic Wasting Disease Alliance, www.cwd-info.org, 2004). 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 four positive deer located at White Sands Missile Base, New Mexico (Chronic Wasting Disease Alliance, www.cwd-info.org, 2004).

Incubation time of the disease, from infection to clinical signs, is a few months to approximately 3 years; clinical signs may include a loss of body condition and weight, excessive salivation, ataxia, and behavioral changes; and there is no known cure for the disease (Williams and Young 1980, Spraker et al. 1997, Miller et al. 1998).

Public health officials and the Centers for Disease Control in Atlanta, Georgia, have not identified strong evidence for CWD transmission to humans or animals other than deer and elk (Belay et.al., 2004). Experimental and circumstantial evidence suggests that transmission of the disease in deer and elk is through direct contact with infected animals (Miller et al. 1998) or by contact with a contaminated environment (Miller et.al., 2004).

Wildlife disease control strategies must be based on an understanding of specific disease etiology and epidemiology, and most infectious diseases are extremely difficult to eliminate from wild populations once established. Because the epidemiological attributes of CWD remain nebulous, Minnesota Department of Natural Resources (MN DNR) is attempting to acquire all available information about CWD and effective control strategies primarily by assessing the progress of the disease in other states and observing the outcomes of selected management alternatives. Given the extended incubation period associated with CWD, the apparent capacity for horizontal and vertical transmission, and the possibility of environmental contamination, it is imperative that CWD be identified, isolated, and controlled as rapidly as possible following detection within a population.

In response to the discovery of CWD in wild Wisconsin deer and a Minnesota captive elk herd in 2002, MN DNR developed a comprehensive wild deer CWD monitoring program that includes 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.

2003 HUNTER-KILLED DEER SURVEILLANCE METHODS

Sampling Areas

During the 2003 Minnesota deer hunting seasons, 37 sampling areas consisting of 59 Deer Management Areas (DMA) were selected for CWD monitoring of hunter-killed deer (Fig. 1). The plan was based upon a blocking protocol that enabled greater utilization of available personnel, and resulted in a more efficient approach to surveillance testing, as compared to the 2002 sampling plan. Due to the extended incubation period of CWD, deer $\geq$ 1.5 years of age were selected. To optimize the time spent collecting samples, collections occurred primarily during the Minnesota firearms deer season. All samples were voluntarily submitted by hunters.

Sample Size and Distribution

Using a power analysis, sample sizes for each sampling area were determined to ensure a $\geq$ 95% probability of detecting the disease, 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 to detect an infection rate of 1% with 95% confidence (Table 1). All sample locations were mapped.

Deer Head Collection

During the 2003 Minnesota firearms deer season, 132 registration stations within the selected DMAs were staffed for sample collection. Staff were trained and provided equipment to collect hunter data and to remove deer heads. Hunters were interviewed and data collected, including the DMA where the deer was harvested, the specific harvest location, hunter contact information, and MN DNR number. Additionally, the age of the deer was estimated. Deer heads were removed at the base of the skull using scalpels. All heads were given an ID number, individually bagged, and transported with datasheet to "extraction" sites.

Lymph Node Sample Extraction

Eleven "extraction" sites were established to collect medial retropharyngeal lymph node (MRPLN) samples from collected deer heads. Over 90 DNR wildlife research staff and veterinary/graduate students were trained to extract the MRPLNs. The process entailed cutting the soft tissue between the occipital condyles and the trachea, and identifying the lymph tissue. Once removed, the MRPLNs were stored in whirl-pak bags and frozen. All samples were labeled with the same ID number previously assigned to the deer head.

CWD Testing

All samples collected were transported to the Farmland Wildlife Population and Research Station in Madelia where they were inventoried, entered into a database, and shipped to the University of Minnesota Veterinary Diagnostic Laboratory for enzyme-linked immunosorbent assay (ELISA) testing of the lymph node tissue for the presence of the abnormal prion protein.

2003 RESULTS

CWD Surveillance

No positive results were detected in the 10054 usable samples collected from the selected sampling areas (Table 2). Females and males comprised 44% and 56% of the samples. Less than 3% (300) of the total samples (10354) were unusable. Assuming that the samples were randomly collected from each DMA, preliminary 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. Distribution maps for every sampling area were produced (Fig. 2).

2004 Surveillance

For 2004, the MN DNR plans to test the remaining DMA's across the state. (Fig. 3). The sampling areas, modeled deer population size, and the number of samples necessary to detect CWD at an infection rate of 1% with 95% confidence were determined similar to 2002 and 2003 (Table 3). A database will be maintained containing hunter contact information, DMA, location of harvest, and deer age. MPRLNs will be collected, cooled and/or frozen, and transported to the University of Minnesota Veterinary Diagnostic Laboratory in St. Paul, Minnesota for ELISA testing.

LITERATURE CITED

Belay, E.D., R.A. Maddox, E.S. Williams, M.W. Miller, P. Gambetti, and L.B. Schonberger. 2004. Chronic wasting disease and potential transmission to humans. Emerging Infectious Diseases Vol. 10, No. 6, June 2004. Available from www.cdc.gov/eid

Canadian Food Inspection Agency. 2002. Chronic wasting disease (CWD) of deer and elk. Canadian Food Inspection Agency Fact Sheet. http://www.cfia-acia.agr.ca/english/anima/heasan/disemala/cwdmdce.shtml [Accessed 27 August 2002.]

Chronic Wasting Disease Alliance. 2004. Learn about CWD: Map. www.cwd-info.org. [Accessed 6 July 2004.]

Cressie, N.A. 1993. Statistics for Spatial Data. Wiley Series, New York.

Miller, M.W., E.S. Williams, N.T. Hobbs, and L.L. Wolfe. 2004. Environmental sources of prion transmission in Mule Deer. Emerging Infectious Diseases Vol. 10, No. 6, June 2004. Avaialable from http://www.cdc.gov/ncidod/EID/vol10no6/04-0010.htm

Miller, M.W., E. S. Williams, C. W. McCarty, T. R. Spraker, T. J. Kreeger, C. T. Larsen, and E. T. Thorne. 2000. Epizootiology of chronic wasting disease in free-ranging cervids in Colorado and Wyoming. Journal of Wildlife Diseases 36:676-690.

Miller, M. W., M. A. Wild, and E. S. Williams. 1998. Epidemiology of chronic wasting disease in captive Rocky Mountain elk. Journal of Wildlife Diseases 34:532-538.

Spraker, T. R., M. W. Miller, E. S. Williams, D. M. Getzy, W. J. Adrian, G. G. Schoonveld, R. A. Spowart, K. I. O'Rourke, J. M. Miller, and P. A. Merz. 1997. Spongiform encephalopathy in free-ranging mule deer (*Odocoileus hemionus*), white-tailed deer (*Odocoileus virginianus*) and Rocky Mountain elk (*Cervus elaphus nelsoni*) in northcentral Colorado. Journal of Wildlife Diseases 33:1-6.

United States Animal Health Association. 2001. Report of the Committee on Wildlife Diseases. http://www.usaha.org/reports/reports01/r01wd.html [Accessed 27 August 2002.]

Williams, E. S., and S. Young. 1980. Chronic Wasting Disease of captive mule deer: a spongiform encephalopathy. Journal of Wildlife Diseases 16:89-98.

Williams, E.S., and S. Young. 1993. Neuropathology of chronic wasting disease of mule deer (*Odocoileus hemionus*) and elk (*Cervus elaphus nelsoni*). Veterinary Pathology 30:36-45.

Williams, E.S., M. W. Miller, and E.T. Thorne. 2002. Chronic wasting disease: implications and challenges for wildlife managers. Presented at the North American Wildlife and Natural Resources Conference, April 3-7, 2002, Dallas, Texas.

Table 1. 2003 sampling areas and sample size required to detect an infection rate of 1% with 95% confidence (98% confidence in combined DMAs)

Sampling Area (DMA)	Modeled Pre-Fawn Population Size	CWD Sample Size
Block 1		
115	29952	297
116/122/126/127	10440	382
178	13904	295
180	10590	294
Block 2		
201/204	5523	376
206	2920	283
202/203/208	4443	373
207/404	3726	370
209/210/285	5010	374
401/403	3666	369
405	2289	279
406	2478	281
Block 3		
248	4260	288
411	7960	293
412	9296	293
413	6955	292
414	6071	291
416	3264	285
422/423	3770	370
424/431	3751	370
433/446/447	7882	380
425/435	4382	372
Block 4		
223/224	5440	376
235/236	8718	381
418	5548	290
419/429	4083	371
426/427/428	7070	375
Block 5		
341	6354	291
342	5209	290
343/465	9461	381
344	3250	285
345	4238	288
346	5997	291
347/467	8501	380
348	5478	290
349	7773	292
462	3036	284

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

Sampling Area (DMU)	Total # of Samples Collected	Total # of Usable Samples	Negative	Positive	Unusable	Total # Usable Samples Female	% Usable Female	Total # Usable Samples Male	% Usable Male	Total % Unusable	Confidence Interval 1% Infection Rate
BLOCK 1											
115	363	349	349	0	14	137	39.26%	209	59.89%	3.86%	97.00
116/122/126/127	246	233	233	0	13	45	19.31%	184	78.97%	5.28%	90.50
178	437	424	424	0	13	165	38.92%	251	59.20%	2.97%	98.65
180	321	313	313	0	8	133	42.49%	180	57.51%	2.49%	95.90
BLOCK 2											
201/204	382	366	366	0	16	176	48.09%	190	51.91%	4.19%	97.78
206	285	278	278	0	7	146	52.52%	130	46.76%	2.46%	94.70
202/203/208	325	314	314	0	11	158	50.32%	154	49.04%	3.38%	96.20
207/404	362	350	350	0	12	166	47.43%	182	52.00%	3.31%	97.50
209/210/285	378	365	365	0	13	184	50.41%	181	49.59%	3.44%	97.77
401/403	285	270	270	0	15	117	43.33%	152	56.30%	5.26%	94.00
405	245	234	234	0	11	100	42.74%	134	57.26%	4.49%	91.60
406	194	185	185	0	9	85	45.95%	99	53.51%	4.64%	85.50
BLOCK 3											
248	272	266	266	0	6	147	55.26%	117	43.98%	2.21%	93.70
411	336	330	330	0	6	164	49.70%	162	49.09%	1.79%	96.60
412	219	213	213	0	6	95	44.60%	111	52.11%	2.74%	88.50
413	157	154	154	0	3	75	48.70%	79	51.30%	1.91%	79.00
414	421	415	415	0	6	216	52.05%	199	47.95%	1.43%	98.67
416	274	267	267	0	7	125	46.82%	136	50.94%	2.55%	93.90
422/423	224	219	219	0	5	92	42.01%	124	56.62%	2.23%	89.60
424/431	233	228	228	0	5	87	38.16%	141	61.84%	2.15%	90.60
433/446/447	392	378	378	0	14	126	33.33%	249	65.87%	3.57%	97.96
425/435	269	258	258	0	11	89	34.50%	169	65.50%	4.09%	93.10
BLOCK 4											
223/224	298	286	286	0	12	153	53.50%	133	46.50%	4.03%	94.78
235/236	305	299	299	0	6	108	36.12%	189	63.21%	1.97%	95.30
418	299	292	292	0	7	146	50.00%	144	49.32%	2.34%	95.10
419/429	250	241	241	0	9	102	42.32%	137	56.85%	3.60%	91.76
426/427/428	288	276	276	0	12	82	29.71%	189	68.48%	4.17%	94.09

Table 2 (continued). Summary of samples collected by sampling area. Sample numbers include hunter killed, car killed and opportunistic deer.

Sampling Area (DMU)	Total # of Samples Collected	Total # of Usable Samples	Negative	Positive	Unusable	Total # Usable Samples Female	% Usable Female	Total # Usable Samples Male	% Usable Male	Total % Unusable	Confidence Interval 1% Infection Rate
BLOCK 5											
341 (2002 &2003)	324	307	307	0	17	107	34.85%	198	64.50%	5.25%	95.77*
342 (2002 &2003)	349	336	336	0	13	175	52.08%	154	45.83%	3.72%	96.95*
343/465	310	302	302	0	8	127	42.05%	175	57.95%	2.58%	95.42
344	181	179	179	0	2	97	54.19%	82	45.81%	1.10%	84.30
345 (2002 & 2003)	328	318	318	0	10	138	43.40%	178	55.97%	3.05%	96.39*
346 (2002 &2003)	392	380	380	0	12	161	42.37%	218	57.37%	3.06%	98.06*
347/467	365	362	362	0	3	139	38.40%	223	61.60%	0.82%	97.57
348	296	289	289	0	7	139	48.10%	143	49.48%	2.36%	94.94
349	331	323	323	0	8	147	45.51%	172	53.25%	2.42%	96.37
462	237	235	235	0	2	93	39.57%	140	59.57%	0.84%	91.44
2003 Total	**10354**	**10054**	**10054**	**0**	**300**		**43.40%**		**55.70%**	**2.84%**	

Table 3. 2004 proposed sampling areas and sample size required to detect an infection rate of 1% with 95% confidence (98% confidence in combined DMUs)

Sampling Area (DMU)	Modeled Pre-Fawn Population Size	CWD Sample Size
Block 1		
104	14564	295
107	15160	295
110/283	5940	377
205/214	6000	377
211	10986	294
213(Red Lake)		300
Block 2		
167	10560	294
168	12308	294
170	17095	295
172	15785	295
197	9600	293
242	14003	295
243	9734	294
245	16907	295
246	19708	296
Block 3		
244/251/287	23594	386
297/298	9997	382
402	1939	276
407	2657	282
408	2519	281
409	3936	287
411	7960	293
420/421	3261	367
Block 4		
152/157	23287	386
156	13216	295
159	12496	295
174	10020	294
183	10605	294
222	5562	290
225	9656	294
249	9538	293
337/338/339	5442	376
228	3555	286

Table 3 (continued). 2004 proposed sampling areas and sample size required to detect an infection rate of 1% with 95% confidence (98% confidence in combined DMUs)

Sampling Area (DMU)	Modeled Pre-Fawn Population Size	CWD Sample Size
Block 5		
440	2515	281*
442	3143	284
443	1852	275*
448	3263	285*
449	4187	288*
450	1795	275*
451	2198	279*
452	1592	272*
453	2770	283*
454/455	4134	372
456	2349	280*
457	1964	275*
458	1644	273*
459	3701	286
461	2597	282*
463	1494	270*
464	1885	276*
466	2979	284*

* Over 2 year period

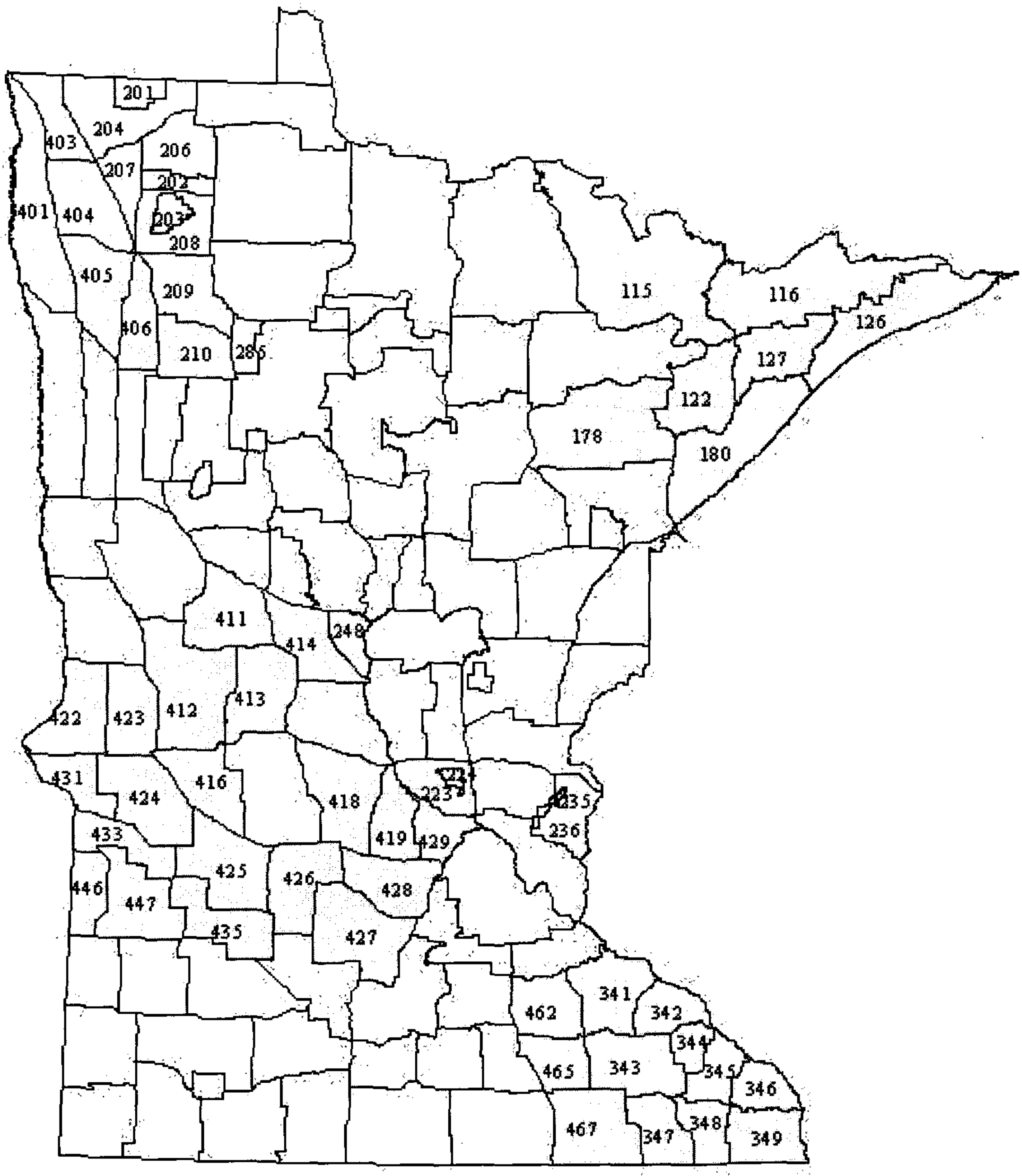


Figure 1. 2003 Chronic Wasting Disease sampling areas denoted by Deer Management Area.

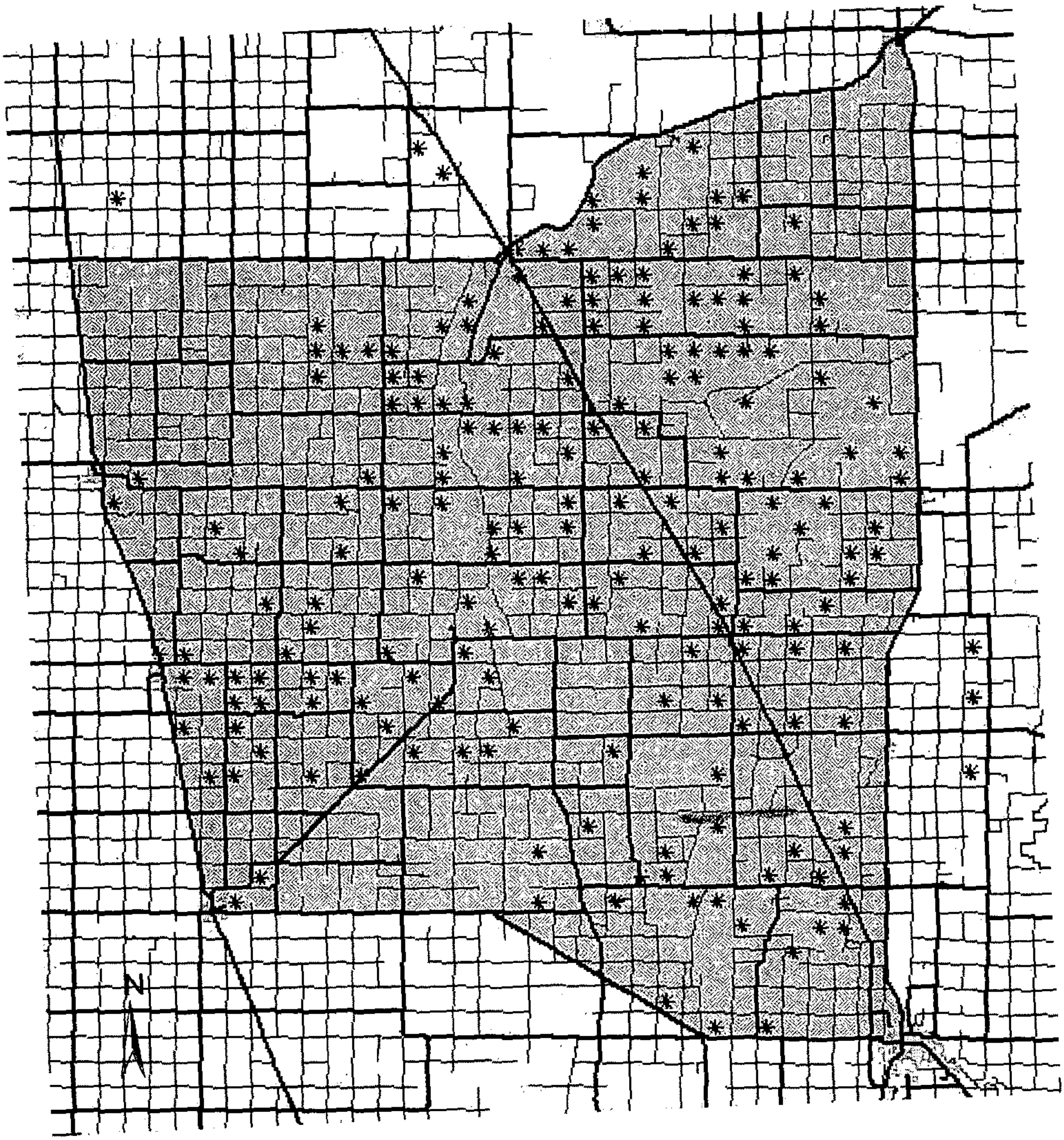


Figure 2. Sampling Area 404-207. Points denote harvest locations of deer tested for CWD.

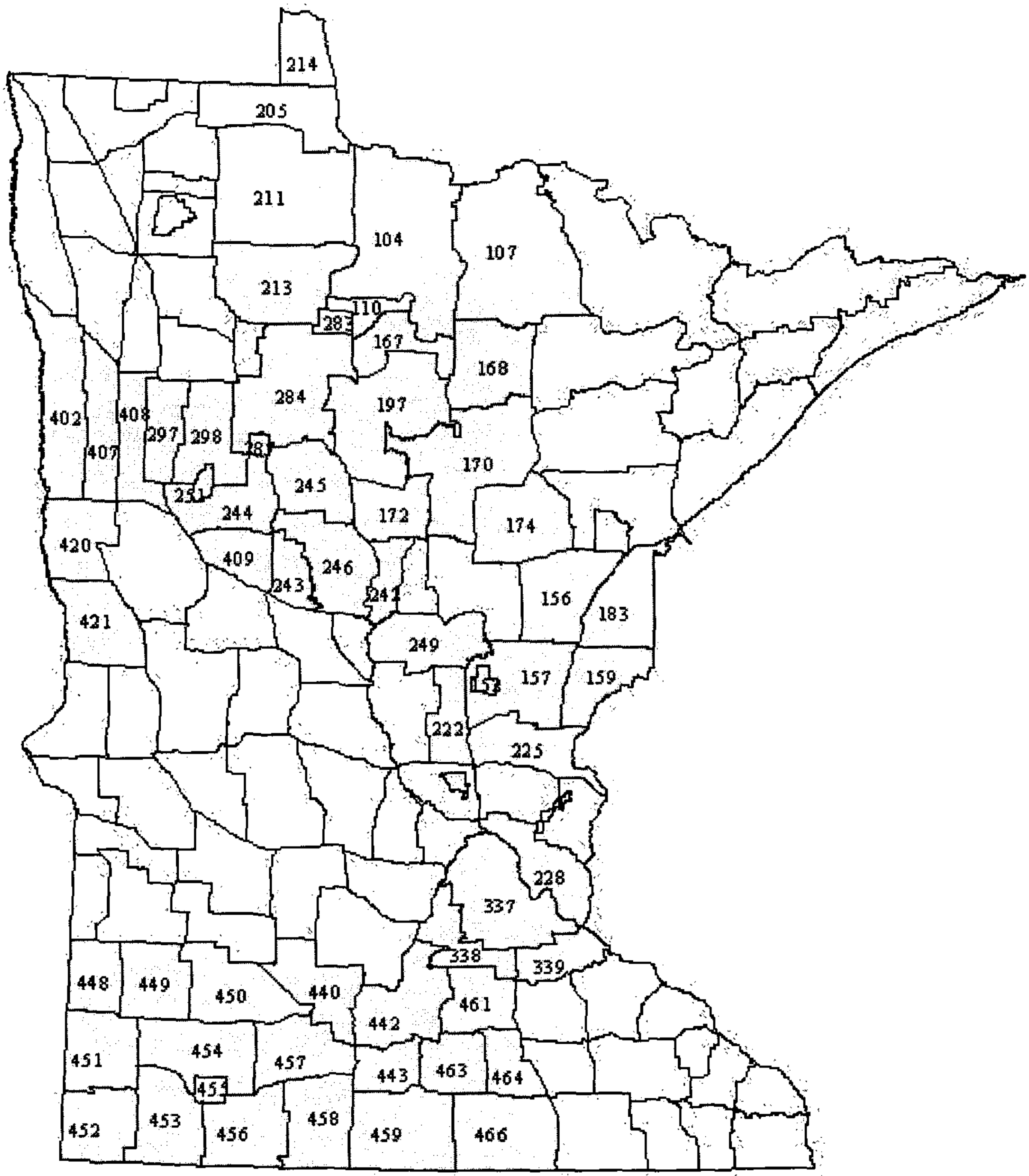


Figure 3. Proposed sampling areas for 2004. Areas are divided into blocks for collection purposes.

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

Kurt Haroldson, John Giudice, and Wendy Krueger

INTRODUCTION

Preferred winter habitat for ring-necked pheasants (*Phasianus colchicus*) in the Midwest includes grass, wetlands, woody cover, and a dependable source of food (primarily grain from harvested or standing crops) near cover (Gates and Hale 1974, Trautman 1982, Perkins et al. 1997, Gabbert et al. 1999). In the northern pheasant range, grain stubble and grass are often buried by snow and rendered unavailable for food and cover, respectively. Wetlands and high quality woody habitats that provide shelter at ground level have been extensively removed from agricultural landscapes. 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).

Although wetland restorations, woody habitats, and food plots are eligible cover practices in farm programs such as the Conservation Reserve Program (CRP), the vast majority of enrollments have been planted to grass habitats. Enrolling a more effective balance of cover practices has been hindered by the lack of information on how much winter habitat is needed to sustain pheasant populations within local landscapes. The purpose of this study is to quantify the relationship between amount of winter habitat and pheasant abundance over a range of winter conditions.

METHODS

During winter 2002-03, 9 study areas were selected in each of 4 regions located near Marshall, Windom, Glenwood, and Faribault (Fig.1). 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 sp.*) 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 standing corn 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.

We estimated the amount of winter cover, winter food, and reproductive cover on each study area by cover mapping to a geographic information system (GIS) from 2003 digital aerial photography. We used the Farm Service Agency's GIS coverage of farm fields (Common Land Units) as a base map and edited the field boundaries to meet 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 has not been completed for 2003, 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), which were conducted by high school teachers and students. 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 of pheasants observed, classified by sex and age. 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. Observers recorded weather conditions on individual study areas at the start and end of each survey. All study areas within a region were generally surveyed 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 units were used to record the time and position of observers throughout each survey as track logs, and to record the location of observed pheasants as 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 mean of 10 repeated surveys. 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 initial progress report.

SPRING SURVEY RESULTS

Observers completed all 360 surveys (10 repetitions on 36 study areas) during the spring 2003 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, 84% of the surveys were started with at least light dew present. Seventy-four percent of surveys were started under clear skies (<30% cloud cover) and 55% reported wind speeds <4 miles/hour. Only 1% of surveys were started on mornings with wind >10 miles/hour or with temperature <32°F. Among regions, Faribault experienced the most cloud cover (21% of surveys started with ≥60% clouds) and Glenwood experienced the most wind (63% of surveys started with wind speed ≥4 miles/hour).

Pheasants were counted on all 36 study areas during spring 2003. Abundance indices among study areas ranged from 18-306 pheasants observed per 100 miles (Table 1). Over all study areas, the mean pheasant index was 105 birds/100 miles. Pheasant indices were highest in the Windom region (162 birds/100 miles), followed by Glenwood (101 birds/100 miles), Marshall (87 birds/100 miles), and Faribault (70 birds/100 miles).

Hens were relatively abundant among study areas in 2003. The overall hen index averaged 56 hens/100 miles, and varied among study areas from 9-207 hens/100 miles (Table 1). Hen indices were highest in the Windom region (92 hens/100 miles), followed by Glenwood (52 hens/100 miles), Marshall (44 hens/100 miles), and Faribault (33 hens/100 miles). The observed hen:rooster ratio varied from 0.2 to 2.1 among study areas (Table 1). Fewer hens than roosters were observed on 1 study area in the Glenwood region, 2 areas in Windom, 3 areas in Marshall, and 6 areas in Faribault. The low hen:rooster ratios in the Marshall and Faribault study areas may reflect later survey dates (9-15 days later than other study areas), when a larger proportion of hens were nesting (and not visible to observers).

Pheasant indices were positively related to habitat abundance in all regions except Glenwood (Fig. 2). Cover index explained 55% of the variation in pheasant indices in the Marshall region, 43% in Faribault, 27% in Windom, but only 1% in Glenwood. At this early stage in our evaluation, we can't explain the poor pheasant-habitat relationship on the Glenwood study areas. However, habitat estimates (and possibly the model) will be improved when we complete the cover mapping of all study areas.

SUMMER SURVEY RESULTS

Observers completed all 360 surveys during the summer 2003 season. Weather conditions during the summer surveys ranged from excellent (calm, clear sky, heavy dew) to poor (light or no dew, overcast sky, or rain). Over all regions, 81% of the surveys were started with moderate-heavy dew present. Seventy-one percent were started under clear skies (<30% cloud cover), and 82% reported wind <4 miles/hour. In comparison, 91% of the statewide August Roadside Surveys were started under medium-heavy dew conditions, 89% under clear skies, and 79% 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 2003. Abundance indices among study areas ranged from 20-620 pheasants observed per 100 miles (Table 2). Over all study areas, the mean pheasant index was 183 birds/100 miles. Pheasant indices were higher in the Windom region (284 birds/100 miles), than in Faribault (165 birds/100 miles), Marshall (143 birds/100 miles), and Glenwood (140 birds/100 miles).

The overall hen index averaged 31 hens/100 miles, and varied among study areas from 6-106 hens/100 miles (Table 2). Hen indices were higher in the Windom region (51 hens/100 miles), than in Marshall (26 hens/100 miles), Glenwood (24 hens/100 miles), or Faribault (24 hens/100 miles). The observed hen:rooster ratio varied from 0.8 to 6.5 among study areas (Table 2) and averaged 2.4 overall. Fewer hens than roosters were observed on 1 study area each in the Windom, Marshall, and Faribault regions.

The brood index ranged from 3-77 broods/100 miles among study areas (Table 2) and averaged 26 broods/100 miles overall. The brood index was higher in the Windom region (36 broods/100 miles) than in the other regions (24 broods/100 miles in Faribault, 22 broods /100 miles in Marshall, and 20 broods /100 miles in Glenwood). Mean brood size averaged 5.1 chicks/brood and was relatively consistent among regions (5.5 in Faribault, 5.4 in Windom, 5.0 in Glenwood, and 4.6 in Marshall). On average, 0.6 broods were observed for every hen counted during spring surveys. The brood recruitment index was highest in Faribault (0.9 broods/spring hen), followed by Marshall (0.6 broods/spring hen), Windom (0.5 broods/spring hen), and Glenwood (0.4 broods/spring hen); this may be explained by the later spring surveys in Faribault and some Marshall study areas (when a higher proportion of hens had started nesting and were missed in spring surveys).

Pheasant indices were positively related to habitat abundance in all regions except Glenwood (Fig. 2). Cover index explained 50% of the variation in pheasant indices in the Marshall region, 29% in Windom, 10% in Faribault, and 0% in Glenwood. We will focus future investigation into why the pheasant-cover relationship was so weak in Glenwood, and to explain outliers in the Faribault and Windom plots (Fig. 2).

FUTURE PLANS

For the next reporting period, we plan to finish cover mapping the 36 study areas. This will permit a more accurate assessment of habitat abundance than the estimates used for this report. In addition, we will continue to survey pheasant populations during spring and summer. 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 of the study area boundaries.

LITERATURE CITED

Berner, A. H. 2001. Winterizing Minnesota's landscape for wildlife: providing food and cover for wintering pheasants and associated resident wildlife in Minnesota's farmlands. Minnesota Department of Natural Resources, St. Paul, Minnesota, USA.

Gabbert, A. E., A. P. Leif, J. R. Purvis, and L. D. Flake. 1999. Survival and habitat use by ring-necked pheasants during two disparate winters in South Dakota. Journal of Wildlife Management 63:711-722.

Gates, J. M., and J. B. Hale. 1974. Seasonal movement, winter habitat use, and population distribution of an east central Wisconsin pheasant population. Wisconsin Department of Natural Resources Technical Bulletin 76, Madison, Wisconsin, USA.

Haroldson, K. J., R. O. Kimmel, and A. H. Berner. 1998. Population responses of ring-necked pheasants, gray partridge, and meadowlarks to the Conservation Reserve Program. Pages 10-17 in B. Joselyn, editor. Summaries of wildlife research findings: 1998. Minnesota Department of Natural Resources, St. Paul, Minnesota, USA.

Perkins, A. L., W. R. Clark, T. Z. Riley, and P. A. Vohs. 1997. Effects of landscape and weather on winter survival of ring-necked pheasant hens. Journal of Wildlife Management 61:634-644.

Trautman, C. G. 1982. History, ecology and management of the ring-necked pheasant in South Dakota. South Dakota Department of Game, Fish and Parks Bulletin 7, Pierre, South Dakota, USA.

Table 1. Mean pheasant counts after 10 repeated surveys (N) on 36 study areas in Minnesota, spring 2003.

Study Area	REGION	N	TOTAL/ 100 MI	COCKS/ 100 MI	HENS/ 100 MI	HEN:COCK RATIO
1	MARSHALL	10	129.2	64.6	64.6	1.000
2	MARSHALL	10	93.3	62.5	30.8	0.493
3	MARSHALL	10	88.3	46.6	41.7	0.896
4	MARSHALL	10	195.0	75.0	120.0	1.600
5	MARSHALL	10	63.9	37.8	26.1	0.689
6	MARSHALL	10	60.4	26.4	34.0	1.286
7	MARSHALL	10	56.4	28.2	28.2	1.000
8	MARSHALL	10	39.2	17.1	22.1	1.294
9	MARSHALL	10	58.8	29.4	29.4	1.000
10	GLENWOOD	10	40.8	18.4	22.4	1.222
11	GLENWOOD	10	89.0	42.4	46.6	1.100
12	GLENWOOD	10	128.2	77.5	50.7	0.654
13	GLENWOOD	10	82.6	37.8	44.8	1.184
14	GLENWOOD	10	60.9	29.1	31.8	1.091
15	GLENWOOD	10	196.3	92.1	104.2	1.131
16	GLENWOOD	10	111.4	46.2	65.2	1.412
17	GLENWOOD	10	18.2	7.0	11.2	1.588
18	GLENWOOD	10	180.0	87.7	92.4	1.053
19	WINDOM	10	306.3	98.9	207.4	2.096
20	WINDOM	10	183.6	93.9	89.7	0.956
21	WINDOM	10	87.8	43.4	44.4	1.024
22	WINDOM	10	248.2	115.1	133.1	1.157
23	WINDOM	10	211.9	86.6	125.2	1.446
24	WINDOM	10	95.0	52.5	42.5	0.810
25	WINDOM	10	127.6	48.1	79.5	1.653
26	WINDOM	10	154.1	63.5	90.5	1.426
27	WINDOM	10	46.1	22.2	23.9	1.078
28	FARIBAULT	10	193.6	79.3	114.3	1.442
29	FARIBAULT	10	48.4	30.6	17.8	0.581
30	FARIBAULT	10	48.4	21.8	26.6	1.222
31	FARIBAULT	10	58.5	49.8	8.8	0.176
32	FARIBAULT	10	75.1	41.0	34.1	0.833
33	FARIBAULT	10	69.6	42.2	27.4	0.649
34	FARIBAULT	10	68.6	35.2	33.4	0.950
35	FARIBAULT	10	29.6	17.5	12.1	0.692
36	FARIBAULT	10	40.8	16.7	24.2	1.450

Table 2. Mean pheasant counts after 10 repeated surveys (N) on 36 study areas in Minnesota, summer 2003.

Study Area	REGION	N	TOTAL/ 100 MI	COCKS/ 100 MI	HENS/ 100 MI	HEN:COCK RATIO	CHICKS /100 MI	BROODS /100MI	BROOD SIZE	BROODS/ SUMMER HEN	BROODS/ SPRING HEN
1	MARSHALL	10	267.0	23.5	37.3	1.58824	206.3	35.9	5.744	0.963	0.556
2	MARSHALL	10	74.6	10.6	16.5	1.56000	47.5	11.9	4.000	0.718	0.385
3	MARSHALL	10	64.1	10.7	8.7	0.81818	44.7	11.7	3.833	1.333	0.279
4	MARSHALL	10	209.0	29.5	57.5	1.94915	122.0	35.0	3.486	0.609	0.292
5	MARSHALL	10	131.1	13.9	21.4	1.54545	95.8	19.3	4.957	0.902	0.742
6	MARSHALL	10	142.9	4.8	23.0	4.80000	115.1	20.1	5.714	0.875	0.593
7	MARSHALL	10	192.7	6.4	30.0	4.71429	156.4	32.7	4.778	1.091	1.161
8	MARSHALL	10	130.7	9.9	20.8	2.10000	100.0	20.8	4.810	1.000	0.939
9	MARSHALL	10	71.1	4.8	15.4	3.18182	50.9	13.2	3.867	0.857	0.448
10	GLENWOOD	10	149.5	6.1	20.2	3.33333	123.2	21.2	5.810	1.050	0.945
11	GLENWOOD	10	97.4	6.8	17.1	2.50000	73.5	17.1	4.300	1.000	0.367
12	GLENWOOD	10	141.0	8.6	29.5	3.44444	102.9	20.0	5.143	0.677	0.394
13	GLENWOOD	10	80.9	4.3	13.0	3.00000	63.5	11.3	5.615	0.867	0.252
14	GLENWOOD	10	211.4	6.6	31.1	4.73333	173.7	30.7	5.657	0.986	0.966
15	GLENWOOD	10	340.7	26.9	54.6	2.03448	259.3	43.5	5.957	0.797	0.418
16	GLENWOOD	10	91.4	10.5	16.2	1.54545	64.8	16.2	4.000	1.000	0.248
17	GLENWOOD	10	19.9	4.1	5.8	1.40000	10.0	2.5	4.000	0.429	0.223
18	GLENWOOD	10	125.0	9.3	24.1	2.60000	91.7	19.4	4.714	0.808	0.211
19	WINDOM	10	438.9	40.0	68.4	1.71053	330.5	53.7	6.157	0.785	0.259
20	WINDOM	10	619.7	30.3	106.4	3.50847	483.0	77.1	6.267	0.725	0.859
21	WINDOM	10	257.9	23.2	50.5	2.18182	184.2	33.7	5.469	0.667	0.758
22	WINDOM	10	254.5	22.6	50.5	2.24000	181.4	36.1	5.025	0.714	0.271
23	WINDOM	10	287.1	22.9	57.8	2.52174	206.4	37.9	5.447	0.655	0.302
24	WINDOM	10	302.0	32.0	48.0	1.50000	222.0	34.0	6.529	0.708	0.800
25	WINDOM	10	105.7	18.4	23.1	1.25641	64.2	15.1	4.250	0.653	0.190
26	WINDOM	10	193.9	23.2	37.3	1.60377	133.3	26.3	5.067	0.706	0.291
27	WINDOM	10	91.3	20.4	16.1	0.78723	54.8	12.2	4.500	0.757	0.509
28	FARIBAULT	10	381.0	16.2	51.4	3.17647	313.3	57.1	5.483	1.111	0.500
29	FARIBAULT	10	65.2	5.8	6.8	1.16667	52.6	6.8	7.714	1.000	0.384
30	FARIBAULT	10	66.1	3.2	11.3	3.50000	51.6	9.7	5.333	0.857	0.364
31	FARIBAULT	10	121.6	13.7	18.6	1.35714	89.2	15.7	5.688	0.842	1.786
32	FARIBAULT	10	168.6	12.7	27.1	2.13333	128.8	25.4	5.067	0.938	0.745
33	FARIBAULT	10	481.0	10.3	67.2	6.50000	403.4	66.4	6.078	0.987	2.421
34	FARIBAULT	10	76.3	7.5	14.5	1.94118	54.4	11.4	4.769	0.788	0.341
35	FARIBAULT	10	56.0	8.4	8.4	1.00000	39.1	8.0	4.889	0.947	0.661
36	FARIBAULT	10	65.8	7.5	6.7	0.88889	51.7	11.7	4.429	1.750	0.483

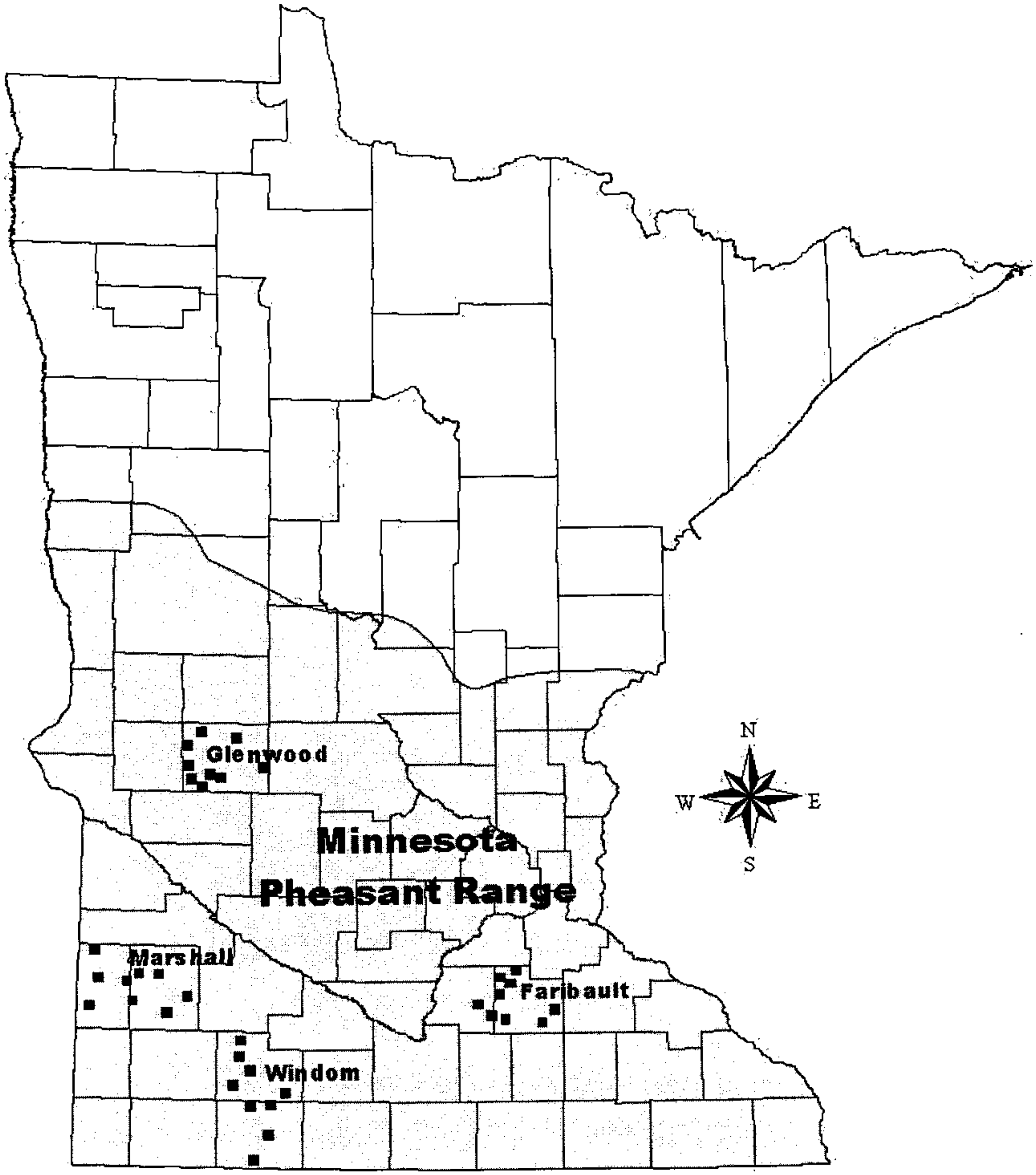


Figure 1. Locations of winter-habitat study areas within Minnesota's pheasant range, 2003.

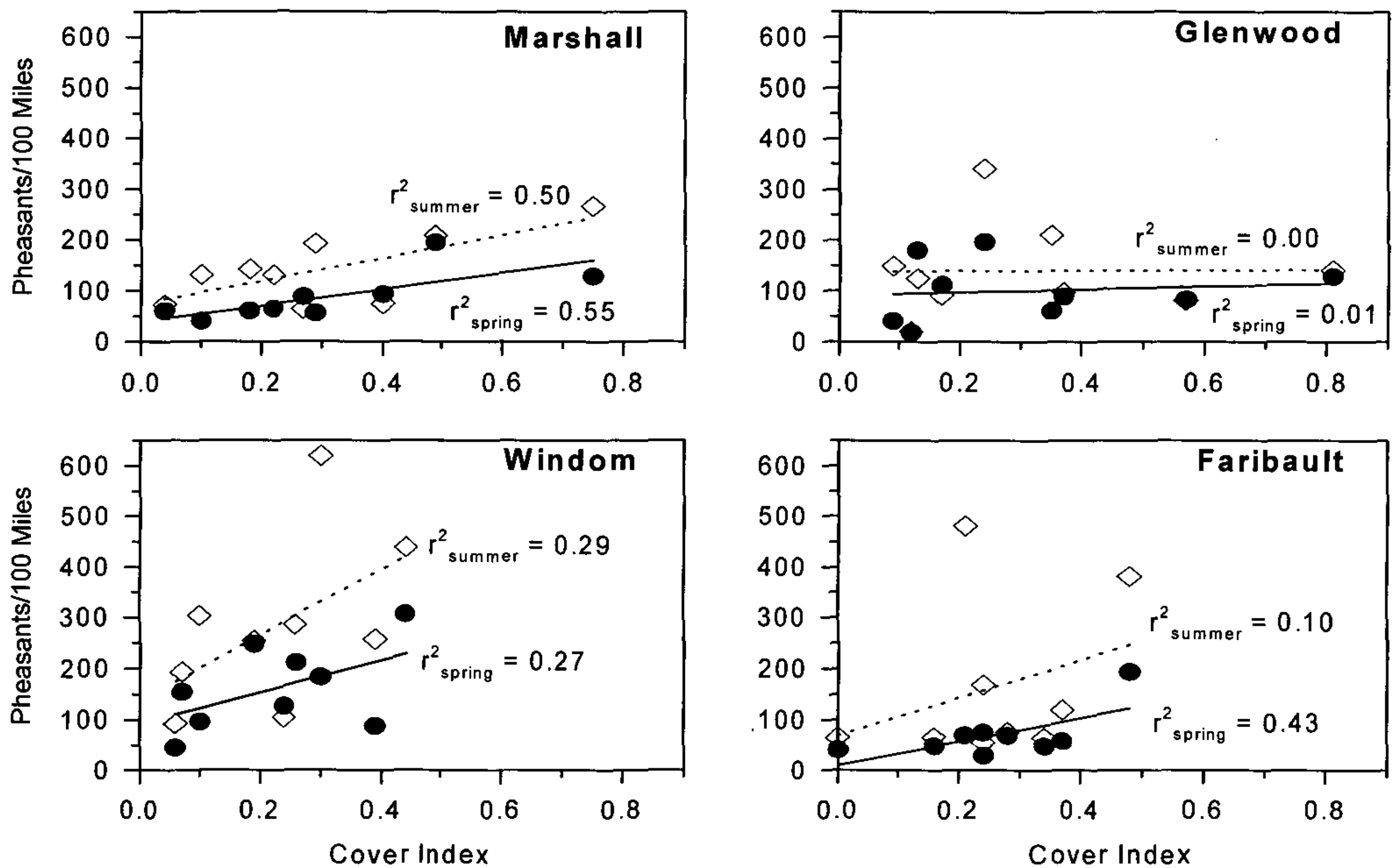


Figure 2. Relationship between relative pheasant abundance (pheasants counted/100 miles of survey) and habitat abundance (cover index) on 9 study areas in each of 4 regions in Minnesota during spring (circles, solid lines) and summer (diamonds, dashed lines) 2003.

2003 MINNESOTA PRAIRIE CHICKEN SEASON SUMMARY

Wendy Krueger, Bryan Spindler, and Erik Steenberg

The 2003 Minnesota prairie chicken (*Tympanuchus cupido pinnatus*) season was the first hunt of its kind since 1942. The 5-day hunt (October 18-22) was limited to 100 permits in 7 permit areas (PA) in northwestern Minnesota between Fergus Falls and Crookston (Fig. 1). Hunter quotas were set for each PA by estimating the minimum population as 2 x the number of males counted on leks within each PA and allowing a maximum harvest of 10% of the minimum population. Applicants were selected through a computerized lottery system. Legal shooting hours were ½ hour before sunrise until sunset. Each licensed hunter was allowed to harvest 2 prairie chickens and hunters were required to register their birds at designated electronic licensing system (ELS) stations.

There were 853 total applicants for the 100 available permits, and 93 permits were issued (Table 1). Landowners were allotted 20% of total permits available in each PA and they purchased 14% of total permits issued. ELS stations registered 115 birds for an average of 1.2 per licensee. Registration varied from 0.6 birds per hunter in PA 407B to 1.8 birds per hunter in PA 405A (Table 1). Overall, 68% of hunters harvested at least 1 bird (Table 2).

A post-hunt mail survey was conducted to evaluate the prairie chicken hunt and hunter satisfaction (Appendix A). The first mailing was sent to all permit holders on January 21, 2004 and a second mailing was sent to non-respondents on February 17, 2004. Ninety-two of the 93 hunters who bought licenses replied to the survey for a 99% return rate. Only 1 respondent (1%) stated that they did not hunt during the 2003 season.

Hunters averaged 2.2 days afield during the 5-day season (Fig. 2). Thirty-two hunters (35%) said that they hunted between 11 and 15 total hours; 18% hunted more than 15 hours during the season (Fig. 3). Fifty-five percent of respondents flushed between 6 and 50 prairie chickens while hunting, 15% flushed less than 6 birds, and 12% flushed over 100 (Fig. 4). An estimated 219 sharp-tailed grouse (Tympanuchus phasianellus campestris) were flushed during the prairie chicken season by 14 respondents.

The survey indicated that hunters killed and retrieved 129 prairie chickens, or 1.4 birds per hunter (n=91) (Table 1). This is slightly higher than ELS registration data. However, there were reports that some ELS stations registered only 1 of a 2-bird limit (both birds needed to be registered separately). A total of 25 prairie chickens were reported as being knocked down and not retrieved during the hunt.

Most respondents hunted public land to some extent (84%) (Fig. 5). Those that hunted private land overwhelmingly reported that access to private lands was easy (98%) (Fig. 6). The most popular hunting technique was to walk with dogs to flush birds (79%), but 38% of hunters used multiple techniques (Fig. 7). Fourteen hunters (15%) reported interference by other hunters at least once.

The vast majority of hunters were satisfied with the hunt based on an average overall satisfaction rating of 8.7 on a scale of 1 (poor) to 10 (excellent) with 60% of hunters rating the hunt excellent (Fig. 8). Ninety-five percent of respondents indicated that they would apply for future Minnesota prairie chicken hunts.

Acknowledgments: R. Kimmel and J. Giudice helped develop the hunter survey. We thank T. Wolfe, B. Winter, R. Hier, D. Hedtke, E. Johnson and K. Haroldson for survey review. T. Rogers provided ArcView support, T. Klinkner assisted with survey mailings and report formatting and D. Hedtke, R. Kimmel, and K. Haroldson edited the report.

Table 1. Applicants, permits available, permits issued and harvest by permit area for the Minnesota prairie chicken season, October 2003.

Permit Area	# Applicants (Landowner)	# Permits Available (Landowner)	#Permits Issued[a] (Landowner)	# Prairie Chickens Registered via ELS	# Prairie Chickens Reported Harvested via Survey	Average # birds per hunter ELS	Average # birds per hunter Survey
405A	120 (2)	12 (3)	13 (1)	23	25	1.8	1.9
407A	109 (3)	13 (3)	14 (2)	10	12	0.7	0.9
407B	97 (0)	15 (3)	13 (0)	8	8	0.6	0.6
407C	80 (2)	13 (3)	14 (2)	20	20	1.4	1.4
420A	116 (6)	14 (3)	10 (3)	14	15	1.4	1.5
420B	203 (2)	18 (4)	17 (2)	24	31	1.4	1.8
421A	128 (3)	15 (3)	12 (3)	16	18	1.3	1.5
Total	853 (18)	100 (22)	93 (13)	115	129	1.2	1.4

[a] Permits issued exceeded permits available when the final hunter selected in the lottery was a member of a hunting party that had applied together.

Table 2. Hunter success by permit area for the Minnesota prairie chicken season, October 2003.

Permit Area	Hunters that Harvested 0 Birds	Hunters that Harvested 1 Bird	Hunters that Harvested 2 Birds	Hunters that Harvested at least 1 bird (%)
405A	1	1	11	92.3
407A	8	2	4	42.9
407B	8	2	3	38.5
407C	3	2	9	78.6
420A	3	0	7	70.0
420B	4	2	11	76.5
421A	3	2	7	75.0
Total	30	11	52	67.7

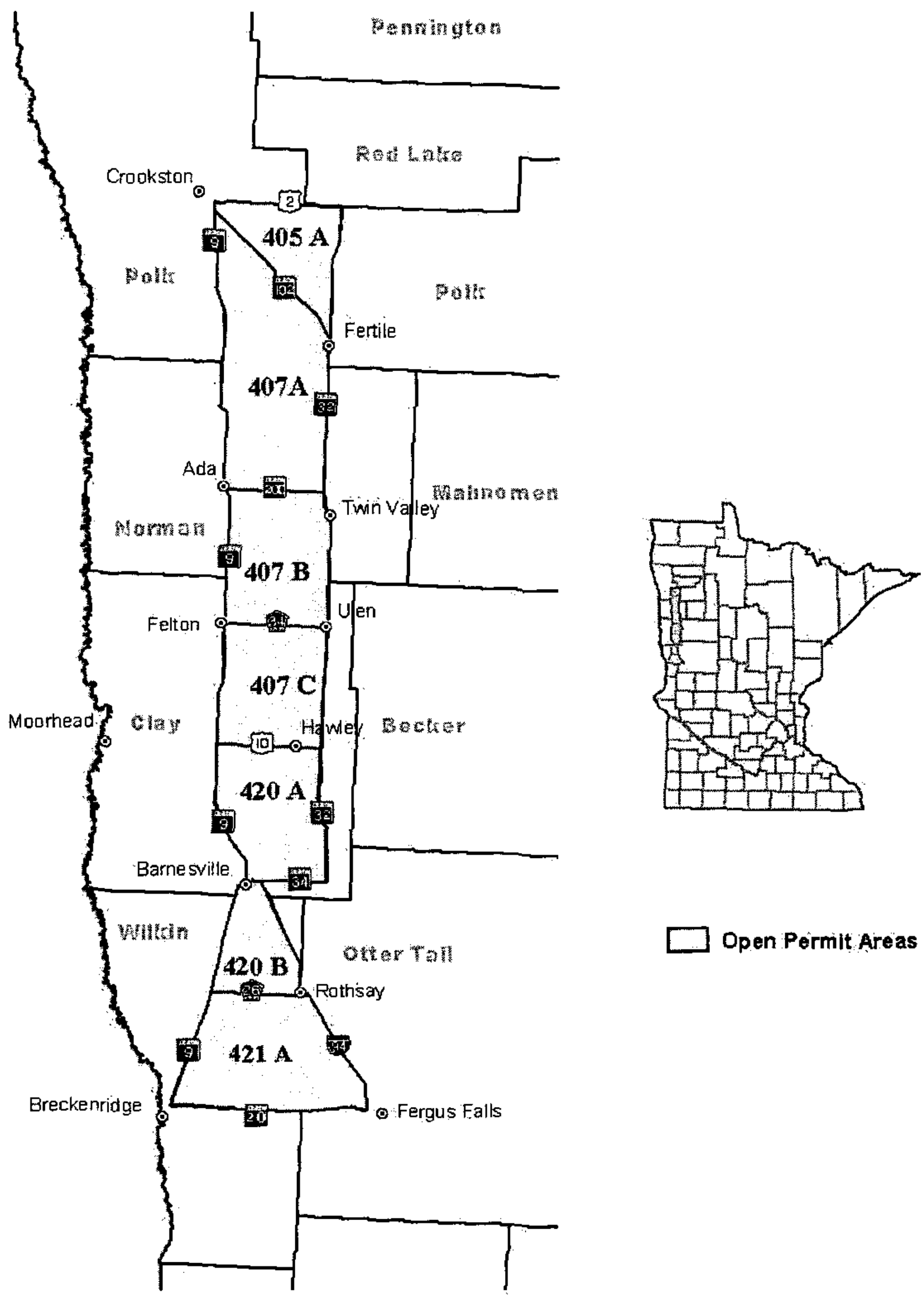


Figure 1. Location of hunting permit areas for the Minnesota prairie chicken season, October 2003

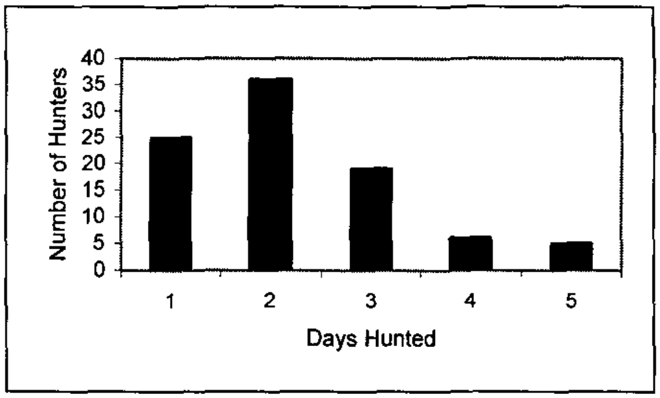


Figure 2. Number of days hunted per hunter during the Minnesota prairie chicken season, October 2003.

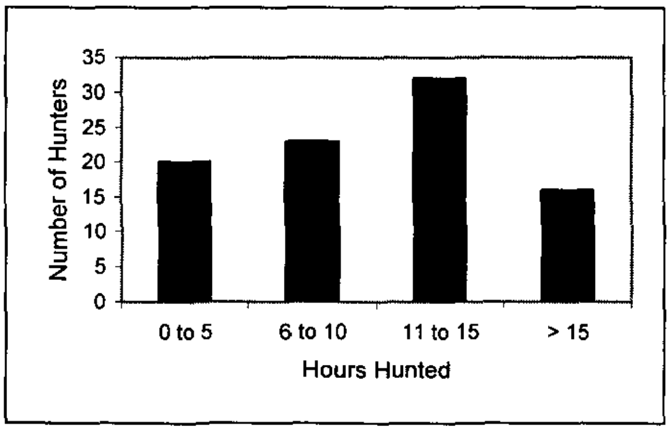


Figure 3. Number of hours hunted per hunter during the Minnesota prairie chicken season, October 2003.

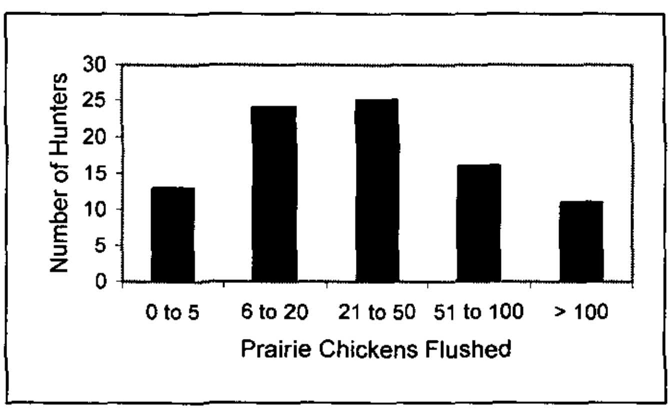


Figure 4. Number of prairie chickens flushed per hunter during the Minnesota prairie chicken season, October 2003.

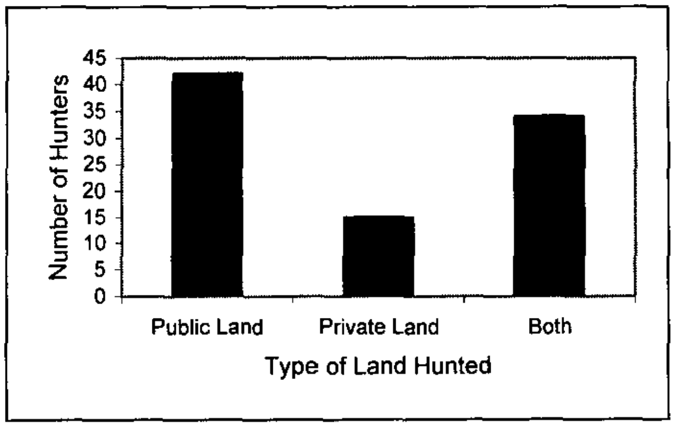


Figure 5. Land ownership class hunted during the Minnesota prairie chicken season, October 2003.

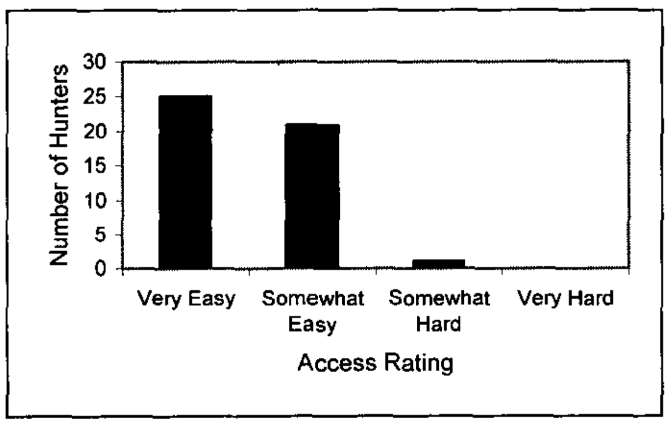


Figure 6. Land access difficulty by hunters using private land during the Minnesota prairie chicken season, October 2003.

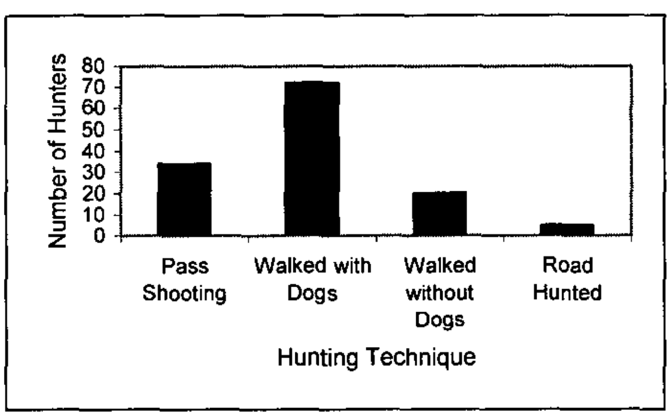


Figure 7. Hunting techniques used during the Minnesota prairie chicken season, October 2003.

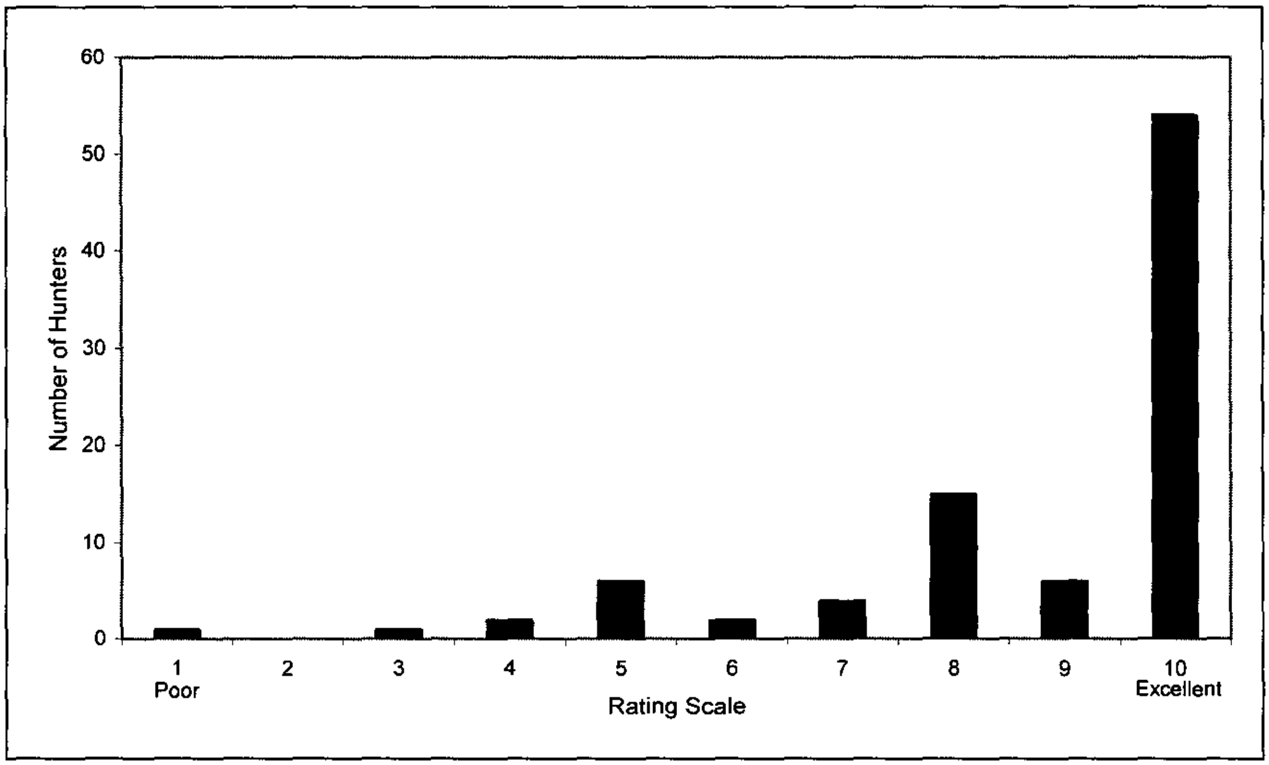


Figure 8. Overall hunt satisfaction with the Minnesota prairie chicken season, October 2003.

Appendix A. Survey form for the 2003 Minnesota prairie chicken hunter survey.

2003 Minnesota Prairie Chicken Hunter Survey
Place an X to mark your answers or fill in the blank.

1) Did you hunt Prairie Chickens in Minnesota during the 2003 season?
YES_____ **NO***_____*If you answered NO, please skip to question 12.

2) How many days did you hunt Prairie Chickens in Minnesota during the 2003 season?
1_____ **2**_____ **3**_____ **4**_____ **5**_____

3) Estimate the total number of hours you spent hunting Prairie Chickens in Minnesota during the 2003 season. **0-5**_____ **6-10**_____ **11-15**_____ **>15**_____

4) Estimate the number of Prairie Chickens you personally flushed while hunting in Minnesota during the 2003 season.
0-5_____ **6-20**_____ **21-50**_____ **51-100**_____ **>100**_____

5) Estimate the number of Sharp-Tailed grouse you personally flushed while hunting Prairie Chickens in Minnesota during the 2003 season? _______

6) How many Prairie Chickens did you personally shoot and retrieve in Minnesota during the 2003 season? _______

7) How many Prairie Chickens did you personally knock down but not retrieve in Minnesota during the 2003 season? _______

8) On which land ownership class did you hunt during the 2003 Minnesota Prairie Chicken season? **Public**______ **Private*** _______ **Both**______
*If you hunted private land, how difficult was it to gain access?
Very Easy____ **Somewhat Easy**____ **Somewhat Hard**____ **Very Hard**____

9) How did you hunt during the 2003 Minnesota Prairie Chicken season?
Pass Shooting_____**Walked With Dog(s)**_____**Walked Without Dogs**_____
Road Hunted_____
If Other Please Explain:____

10) How many times did hunters, other than members of your own party, interfere with your hunting during the 2003 Prairie Chicken season in Minnesota? _____

11) Rate your overall satisfaction with the 2003 Minnesota Prairie Chicken hunt.
POOR **AVERAGE** **EXCELLENT**
1____ **2**_____ **3**_____ **4**_____ **5**_____ **6**_____ **7**_____ **8**_____ **9**_____ **10**_____

12) Will you apply for future Prairie Chicken hunts in Minnesota?
YES_____ **NO**_____

13) Please provide any additional comments on the back.

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

Cory Kassube, Marco Restani, Wendy Krueger, and Richard Kimmel

Wildlife managers have succeeded in establishing wild turkey (*Meleagris gallopavo*) populations north of the ancestral range reported by Leopold (1931). The question of how far north this range can be extended in Minnesota remains unanswered, and little information is available on the survival and productivity of transplanted hens. It appears that deep snow, not low temperatures, lowers winter survival of hens by reducing food availability (Porter et al. 1983, Haroldson et al. 1998). Moreover, poor physiological condition of hens surviving a severe winter is a major factor reducing productivity. In Minnesota, supplemental food is provided in some areas in an attempt to improve the success of turkey transplants.

Despite the practice of planting food plots, only 2 research studies in Minnesota have evaluated supplemental food on winter survival of wild turkeys. Porter et al. (1980) found corn is an important food resource that can improve winter survival of turkeys. Kane (2003) also found higher winter survival of transplanted turkey hens on study areas with supplemental food plots. However, his conclusions regarding the value of supplemental food are equivocal because both food study areas were located at lower latitudes than the 2 control study areas, which confounded availability of supplemental food with latitude.

Our study is a continuation of Kane (2003). We will test 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 predation on supplemental food areas.

First, in 2004 we added 2 new study areas to improve study design and strengthen inferences regarding the relationship between supplemental food and survival (both winter and annual). The Morrison study area (supplemental food) is located at approximately the same latitude as the 2 control study areas (Bradbury and Snake River). The Sherburne study area serves as a control and is located at approximately the same latitude as the 2 supplemental food study areas (Foreston and Bock). Second, we are also examining annual survival and reproduction because Kane (2003) found annual survival between supplemental food and control study areas was similar despite a difference in winter survival across study areas (Fig. 1). Hens on supplemental food study areas had higher mortality during the nesting and brood rearing periods than on control areas, but factors responsible for this pattern remain unknown. Mortality of wild turkey hens is highest during the nesting period primarily because of predation (Vander Haegen et al. 1988, Palmer et al. 1993). Lower survival in Minnesota may have been caused by higher predator abundance on supplemental food study areas. Ultimately, for transplants to succeed and for the continued use of supplemental food to be justified at the population level, higher winter survival must translate into reproduction.

We began intensively monitoring turkey hens in January 2004. In winter 2004, 24 hens from Kane (2003) remained alive. We released an additional 62 hens from 9 January to 13 March. As of 10 May 2004, a total of 61 radio-tagged hens were present on the 6 study areas (Figs. 2 and 3). Turkeys on supplemental food study areas appeared to have higher winter survival than those on control sites, but releases at Bock, Foreston, and Bradbury took place very late in the winter - on 13 March 2004 – thus these birds did not face any severe winter conditions. Causes of mortality from 1 January 2004 to 31 March 2004 included avian predation by great horned owls (*Bubo virginianus*) and bald eagles (*Haliaeetus leucocephalus*), mammalian predation by coyotes (*Canis latrans*) and bobcats (*Lynx rufus*), and starvation.

We will determine productivity and nest success by monitoring hens for movement (Vander Haegen et al. 1988). After nests are located, we will determine clutch size (number of unhatched eggs and egg caps), initial brood size (number of hatched eggs), hatch success (proportion of hatched eggs/clutch), and cause of nest failure, if any. We will use flush-counts at 4 weeks post-hatch to determine productivity (Roberts et al. 1995, Hubbard et al. 1999).

We will determine the relative abundance of mammalian predators during the nesting season because the majority of mortality on the supplemental food study areas occurred at this time (Kane 2003). We will conduct systematic track surveys for mammalian predators to obtain relative abundance on control and food plot study areas (Sovada et al. 1995, 2000). We will locate naturally occurring sites suitable for track identification and randomly select 10 sites/study area to be used in this analysis. We will visit these sites once/month from May – July and identify and count the tracks present at each site.

Acknowledgements

We thank the National Wild Turkey Federation, Minnesota Department of Natural Resources, and Mille Lacs Band of the Ojibwe for sponsoring this project. St. Cloud State University provided administrative support.

LITERATURE CITED

Haroldson, K. J., M. L. Svihel, R. O. Kimmel, and R. R. Riggs. 1998. Effect of winter temperature on wild turkey metabolism. Journal of Wildlife Management 62:229-305.

Hubbard, M. W., D. L. Garner, and E. E. Klaas. 1999. Wild turkey poult survival in southcentral Iowa. J. Wildl. Manage. 63:199-203.

Kane, D. F. 2003. Winter survival of eastern wild turkeys translocated north of their ancestral range in Minnesota. Master's Thesis, St. Cloud State University, St. Cloud, Minnesota, USA.

Leopold, A. S. 1931. Game survey of the north central states. American Game Association. Washington, D.C., USA

Palmer, W. E., G. A. Hurst, J. E. Stys, D. R. Smith, and J. D. Burk. 1993. Survival rates of wild turkey hens in lobolly pine plantations in Mississippi. J. Wildl. Manage. 57:783-789.

Porter, W. F., R. D. Tangen, G. C. Nelson, and D. A. Hamilton. 1980. Effects of corn food plots on wild turkeys in the upper Mississippi Valley. J. Wildl. Manage. 44:456-462.

_____, G. C. Nelson, and K. Mattson. 1983. Effects of winter conditions on reproduction in a northern wild turkey population. J. Wildl. Manage. 47:281-290.

Roberts, S. D., J. M. Coffey, and W. F. Porter. 1995. Survival and reproduction of female wild turkeys in New York. J. Wildl. Manage. 59:437-447.

Sovada, M. A., A. B. Sargeant, and J. W. Grier. 1995. Differential effects of coyotes and red foxes on duck nest success. J. Wildl. Manage. 59:1-9.

_____, M. A., M. C. Zicus, R. J. Greenwood, D. P. Rave, W. E. Newton, R. O. Woodward, and J. A. Beiser. 2000. Relationships of habitat patch size to predator community and survival of duck nests. J. Wildl. Manage. 64:820-831.

Vander Haegen, W. M., W. E. Dodge, and M. W. Sayre. 1988. Factors affecting productivity in a northern wild turkey population. J. Wildl. Manage. 52:127-133.

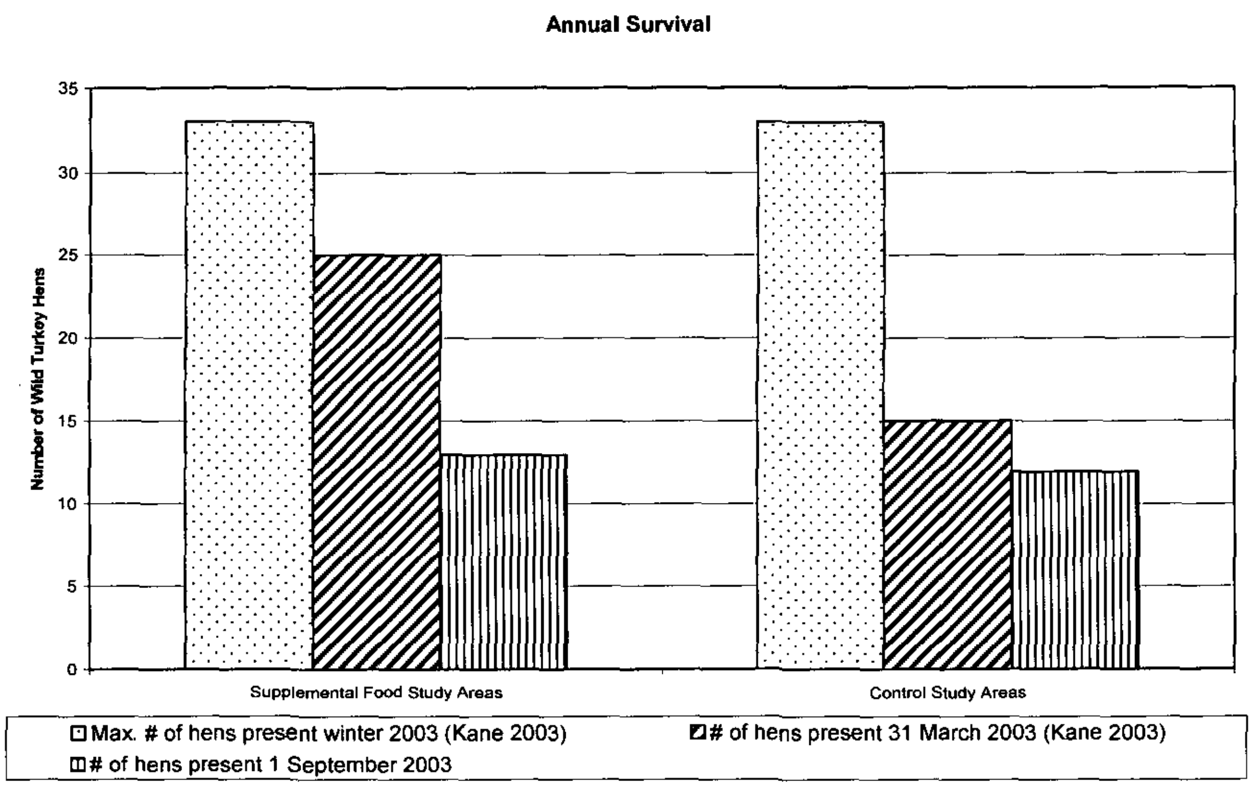


Figure 1. Annual survival of wild turkey hens in central Minnesota, 2003.

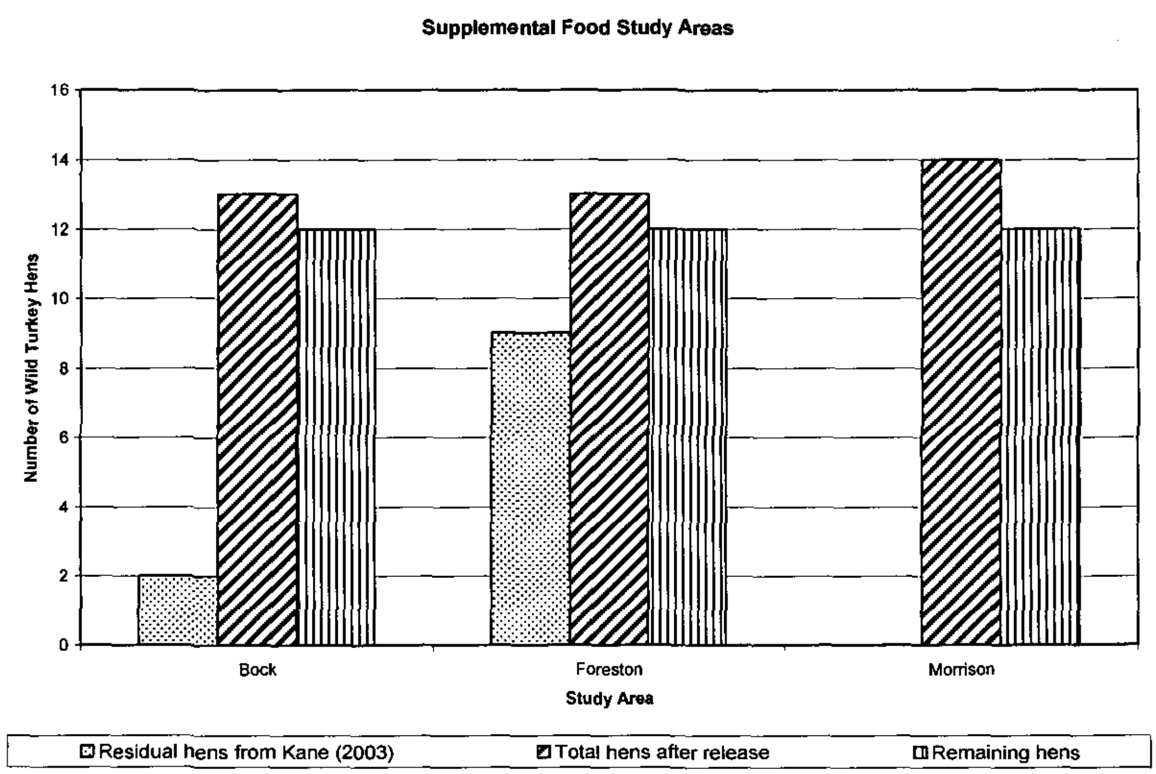


Figure 2. Residual hens from Kane (2003), total hens after release, and remaining hens as of 10 May 2004 on Supplemental Food Study Areas in Central Minnesota.

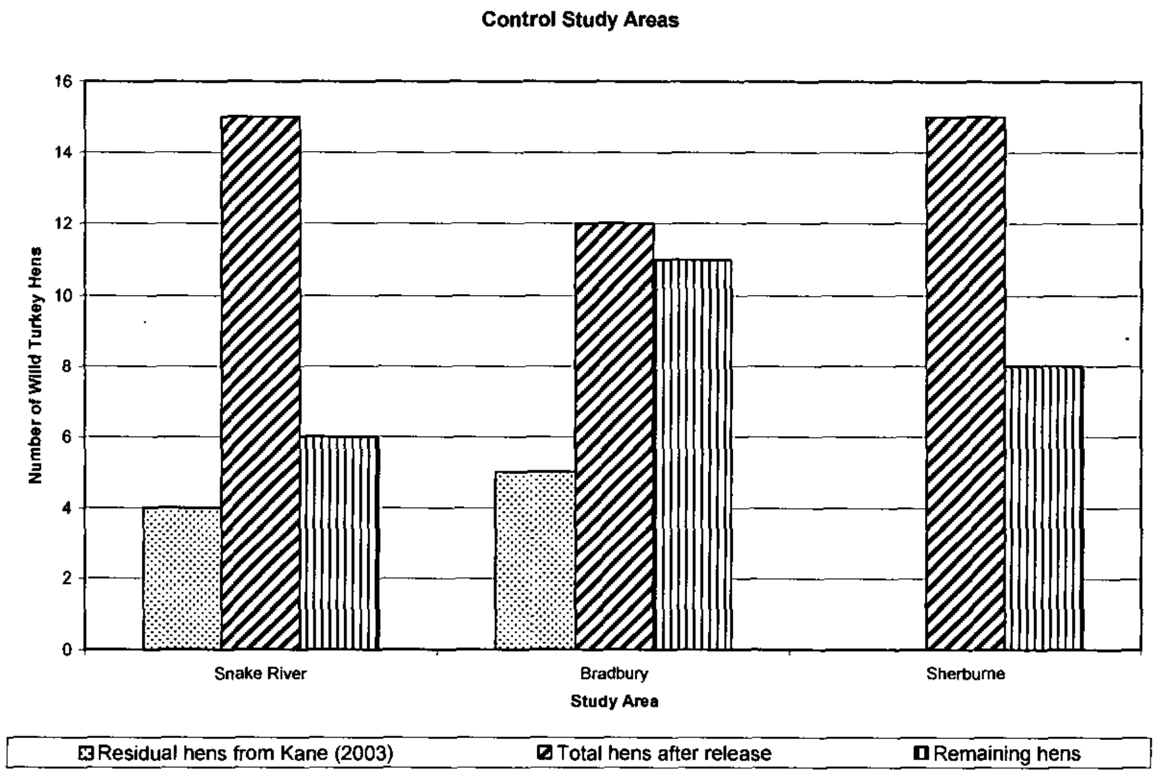


Figure 3. Residual hens from Kane (2003), total hens after release, and remaining hens as of 10 May 2004 on Control Study Areas in Central Minnesota.

Forest Wildlife Populations and Research Group
1201 East Highway 2
Grand Rapids, Minnesota 55744
(281) 327-4432

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

Glenn D. DelGiudice

BACKGROUND

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 dramatically 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.

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.

This study employs a replicated manipulative design, 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 5-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; the Shingle Mill Lake and Dirty Nose Lake sites were included beginning in winter 1992-93.

The objective of the experimental treatment (impact) was to reduce moderate (≥40-69% canopy closure) and optimum conifer thermal cover/snow shelter (≥70% canopy closure) to what is considered a poor cover class (< 40% canopy closure). We just completed our 14th winter of data collection and the 5th 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.

PROGRESS AND SUMMARY OF RESULTS

Capture and Handling of Study Deer

During this study, we have had 1,117 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 have been eartagged and released. As of 31 March 2004, a total of 417 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 and DelGiudice (this Research Summary).

During 5 February-26 March 2004, we had 74 captures, which included 45 initial captures (24 adult and 6 fawn females, 4 adult and 11 fawn males), and 29 (16 adult and 1 fawn females, 6 adult and 6 fawn males) recaptures. Although this winter had the lowest winter severity index (WSI = 38) of the study, actual snow depth was greater than in several other mild winters of the study when trapping success was far lower. In those winters, temperature-days (daily ambient temperatures ≤ -17.8°C), rather than snow-days (daily snow depth ≥38 cm), contributed more points to the their higher WSIs, and snow depths were persistently more shallow throughout the season. Further, during winter 2003-04, 15 female deer were captured by net-gun deployed from helicopter, the success of which is less dependent on snow conditions than Clover trapping and rocket-netting.

The fawn:doe capture ratio remained relatively high (71 fawns:100 does) compared to previous years. The highest fawn:doe capture ratio (105:100) occurred during winter 2000-01, which was moderately severe (WSI = 153), but followed an unprecedented 3 consecutive mild winters (WSI range = 45-57) (P. Bouley, State Climate Office, personal communication). Pregnancy rates of captured adult does during winters 1998-2000 were 100%, and presumably, each of these mild winters was subsequently associated with a high reproductive success in does. Ostensibly, the severity of winter 2000-01 did not have a markedly negative effect on the reproductive success of does in spring 2001, as the fawn:doe ratio of captured deer remained relatively high (81:100) during winter 2001-02. *Actual* fawn:doe capture ratios of 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 primarily to the preceding historically severe winter (1995-96, WSI = 183), during which the highest mortality rate (29.3%)

of radiocollared does was recorded. 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 74 deer captured during winter 2003-04, 29 new females (6 fawns, 23 adults) were recruited into the radiocollared study cohort. Including does already radiocollared when this winter began, 72 females have been monitored during December 2003-May 2004.

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 with a dop-tone or visual ultrasound; pregnant does were not fitted with vaginal radio-transmitter implants (Advanced Telemetry Systems, Inc., Isanti, MN) during this winter, as in several past winters. As in previous winters of the study, female fawns and does were fitted with VHF radiocollars (Telonics, Inc., Mesa, Az) for monitoring their movements and survival; however, 10 (1 killed by wolves 35 days after capture/release) does also were fitted with global positioning system (GPS) radiocollars (Advanced Telemetry Systems, Inc ., Isanti, MN). Body composition of deer was determined *in vivo* by the deuterium-dilution technique during winters 1999-00, 2000-01, 2001-02, which ranged from mild to moderately severe. Additional details are provided in Carstensen and DelGiudice (this Research Summary). Body composition determination was not part of handling during winter 2003-04. Upon completion of handling, all deer immobilizations were reversed with an intravenous injection of yohimbine HCl.

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 experimental treatment phase (1996-1999), and thus far during the post-treatment phase. 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 ($\pm$ 0.4 [SE], $n = 74$) in 1997 to 7.1 ($\pm$ 0.6, $n = 62$) years old (Fig. 1). During 2003, mean age was 6.2 ($\pm$ 0.4) years old, compared to 6.0 ($\pm$ 0.1) years old during the remainder of the study overall.

According to progesterone concentrations ($\geq$1.6 ng/ml, Wood et al. 1986; DelGiudice, unpublished data), the pregnancy rate of captured adult ($\geq$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% (Fig. 2). Mean serum progesterone concentrations differed ($P < 0.05$) between pregnant (3.8 $\pm$ 0.09, range = 1.6-8.9 ng/ml, $n = 218$) and non-pregnant (0.7 $\pm$ 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 ($P \leq 0.05$) in mean body mass at capture for pregnant (63.0 $\pm$ 0.7, range = 45.7-82.5 kg, n = 171) versus non-pregnant (54.6 kg $\pm$ 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 populations and their numbers. Nutrition is intricately related to all aspects of a deer's ecology, and it acts as a mechanistic thread between environmental variability and the variability of deer populations. 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 with an ambient temperature $\leq$ -17.8° C (0° F) and an additional point for each day with a snow depth $\geq$ 38.1 cm (15"). The WSI for our study sites has now ranged from 38 (winters 2003-04) to 185 (winter 1995-96) during the past 14 winters. Not only was the WSI of winter 2003-04 the study's lowest, most of it was attributable to points for temperature-days. However, snow cover was not shallower than during all other winters (Fig. 4). 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). Figure 4 depicts mean daily minimum ambient temperatures (monthly) and mean weekly (julian) snow depths throughout winters 1990-91 to 2003-04. Thus far, the study has captured a wide range of weather conditions, which will enhance the value of all interpretations of data 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 most valuable.

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 average 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 (Fig. 5). At or below this temperature threshold, heat losses may exceed energy expenditure for standard metabolism and activity, and additional heat is 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. As shown by Figure 5, the potential physiological challenge of ambient temperatures in January 2004 might have been rather typical, but certainly it was minimal during February and March 2004. Similarly, I used a snow depth threshold of $\geq$41 cm (16.1"), about two-thirds chest height of adult female deer, because energetically expensive bounding often becomes necessary at this depth, and overall movements become markedly restricted (Kelsall 1969, Kelsall and Prescott 1971, Moen 1976). Clearly, there has been a pronounced variability of days during the study's 14 winters when it is biologically reasonable to expect that there were potentially serious energetic implications associated with ambient temperature or snow depth (Fig. 5). It is noteworthy that extensive statistical analyses of age-specific survival and weather data from the first 6 years of our study (DelGiudice et al. 2002) have shown that snow conditions (depth and density) impose a far greater challenge to survival than ambient temperature. However, our data also indicate that 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.

Status and Cause-Specific Mortality of Study Deer

The status/fate of study deer through 31 December 2003 is shown in Figure 6. The "crude mortality rate" of our study deer was calculated by dividing the number of radiocollared deer that died during a reference period (e.g., winter defined as Dec-May) by the total number of deer that were radiocollared 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 2003, annual mortality rates of radiocollared females ranged from 9.1 to 47.6% (Fig. 7). The mortality rate of 2003 was rather typical at 28.8%. 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 old females, severe weather conditions, and wolf (*Canis lupus*) predation contributed to the higher mortality rates (Fig. 7). The number of antlerless permits issued varied considerably between 1991-2003. As reflected by the hunter-caused mortality rate 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 of the radiocollared deer, and this increased further to the study's second highest level during 2000 (19.4%, Fig. 7). During 2003, antlerless permits were unlimited, and hunter-caused mortality ranked third (17.0% similar to 17.1% in 1995). Although hunter harvest mortality is generally 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 ($n = 42$) than in 2003 ($n = 66$). Wolf-caused mortality of study does was low in 2003. 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 wolf predation rate occurring during 2001. Typically, wolf predation has had its greatest impact on the older segment of the study cohort of does. Mean age of female deer killed by wolves during 10 of the first 13 winters of the study was 6.0 ($\pm$ 1.8, $n = 9$)-11.7 ($\pm$ 1.7, $n = 8$) years old. Mean age of deer killed by wolves during winter 2002-03 was 9.2 ($\pm$ 2.0, $n = 4$) versus 7.8 ($\pm$ 0.6, $n = 59$) during the previous 12 years.

Most of the annual non-hunting mortality of the study deer occurs during winter, and typically, winter mortality of the collared adult female deer has been low (2.0-12.5%, Fig. 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, Fig. 8). Mortality during winter 2003-04 was among the lowest of the study (4.5%). With the inclusion of data from winter 2003-04, the relationship between WSI and percent winter mortality of adult female deer continued to be reasonably strong ($r^2 = 0.52$) and significant ($P = 0.004$, Fig. 9). Predation, and wolf predation specifically, were responsible for a mean 78.64% (± 7.4, range = 0.00-100%, n = 14) and 70.41% (± 7.8, range = 0-100%, n = 14), respectively, of the winter (Dec-May) mortality of collared fawn and adult females throughout the 14-year study period.

Monitoring Wolf Activity

As the study has progressed over the past 14 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, and just 5-6 years prior to initiation of the study had re-entered and became re-established in the area. When the study began in winter 1990-91, the area of the study sites was on the leading edge of wolf range expansion in Minnesota. Since spring 1993, we have captured and radiocollared 45 (25 females, 20 males) wolves from 7-9 packs which range over the 4 study sites (Table 1). Fates of these collared wolves include being killed by members of neighboring packs, shot and killed by humans, killed by cars, natural causes, radio failure, and dispersal out of the vicinity of the study sites.

Our most recent survival analysis showed that 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 been responsible for more wolf deaths than was attributable to natural forces (Fig. 10).

Based on aerial observations, pack sizes have ranged from 2 to 6 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 telemetry location data of the collared wolves are being used to more closely monitor their activity and distribution relative to the distribution and movements of the radiocollared deer. We will capture and radiocollar additional wolves this summer. As mentioned above, year-round monitoring and examination of mortalities of radiocollared 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) have been completed. Forest stand types are classified by dominant tree species, height class, and canopy closure class. Open habitat types, water sources, and roads have also been delineated. We are updating the coverage to include the final experimental cuts that were conducted on the treatment sites (Inguadona Lake, Shingle Mill Lake) and to account for any changes in type classification associated with succession during the last 13 years. Additionally, we are examining a subsampling technique to account for telemetry triangulation error in analyzing vegetation and habitat use by deer (Samuel and Kenow 1992, Kenow et al. 2001). The experimental treatment (i.e., conifer harvest) impacted 157 and 83 hectares (388 and 206 acres) of conifer canopy closure classes A (< 40%), B (40-69%), and C (≥70%) on the Inguadona Lake and Shingle Mill Lake study sites.

Acknowledgments and Project Cooperators

I gratefully acknowledge the excellent diverse skills and contributions of Barry Sampson, Dave Kuehn, and Ken Kerr to all aspects of this project. I am very appreciative of volunteers Richard Nelles and Rod Schloesser, who have contributed significant amounts of time and effort to the winter and spring field seasons for the past 4-7 years. Approximately 130 enthusiastic, competent, and dedicated interns have made collection of winter field data possible, and I am appreciative of their contributions as well. I also thank Mark Lenarz, Group Leader for the Forest Wildlife Populations and Research Group, for his continued support. Ongoing contributions by Don Pierce, Gary Anderson, John Tornes, and Dan Hertle (MNDNR); Larry Olson, Jerry Lamon, Ellisa Bredenburg, and Amy Rand (Cass County Land Department); Kelly Barrett and John Casson (U. S. Forest Service); John Hanson (Blandin Paper Co.); Carl Larson and Michael Houser (Potlatch Corp.) have been essential to the success of this study.

LITERATURE CITED

DelGiudice, G. D., K. E. Kunkel, L. D. Mech, and U. S. Seal. 1990*a*. Minimizing capture related stress on white-tailed deer with a capture collar. J. Wildl. Manage. 54:299-303.

DelGiudice, G. D., L. D. Mech, and U. S. Seal. 1990*b*. Effects of winter undernutrition on body composition and physiological profiles of white-tailed deer. J. Wildl. Manage. 54:539-550.

DelGiudice, G. D., L. D. Mech, U. S. Seal, and P. D. Karns. 1987*a*. Effects of winter fasting and refeeding on white-tailed deer blood profiles. J. Wildl. Manage. 51:865-873.

DelGiudice, G. D., L. D. Mech, U. S. Seal, and P. D. Karns. 1987*b*. Winter fasting and refeeding effects on urine characteristics in white-tailed deer. J. Wildl. Manage. 51:860-864.

DelGiudice, G. D., L. D. Mech, and U. S. Seal. 1994. Winter undernutrition and serum and urinary urea nitrogen of white-tailed deer. J. Wildl. Manage. 58:430-436.

DelGiudice, G. D., and M. R. Riggs. 1996. Long-term research on the white-tailed deer conifer thermal cover relationship: aligning expectations with reality. Trans. of North Am. Wildl. and Nat. Resour. Conf. 61:416-428.

DelGiudice, G. D., M. R. Riggs, P. Joly, and W. Pan. 2002. Winter severity, survival and cause-specific mortality of female white-tailed deer in north central Minnesota. J. Wildl. Manage. 66:698-717.

Gilbert, F. F. 1966. Aging white-tailed deer by annuli in the cementum of the first incisor. J. Wildl. Manage. 30:200-202.

Kelsall, J. P. 1969. Structural adaptations of moose and deer for snow. J. Mammal. 50:302-310.

Kelsall, J. P., and W. Prescott. 1971. Moose and deer behavior in snow. Can. Wildl. Serv. Rep. Ser. 15:1-27.

Kenow, K. P., R. G. Wright, M. D. Samuel, and P. W. Rasmussen. 2001. Integrating SAS and GIS software to improve habitat use estimates from radiotelemetry data. Wildlife Society Bulletin 29:1006-1009.

McDonald, P., R. A. Edwards, and J. F. D. Greenhalgh. 1973. Animal nutrition. Second ed. Hafner Publ. Co., New York. 479pp.

Moen, A. N. 1976. Energy conservation by white-tailed deer in the winter. Ecology 56:192-198.

Samuel, M. D., and K. P. Kenow. 1992. Evaluating habitat selection with radiotelemetry triangulation error. Journal of Wildlife Management 56:725-734.

Stewart-Oaten, A., W. W. Murdoch, and K. R. Parker. 1986. Environmental impact assessment: "pseudoreplication" in time? Ecology 67:929-940.

Warren, R. J., R. L. Kirkpatrick, A. Oelschlaeger, P. F. Scanlon, and F. C. Gwazdauskas. 1981. Dietary and seasonal influences on nutritional indices. J. Wildl. Manage.45:926-936.

Warren, R. J., R. L. Kirkpatrick, A. Oelschlaeger, P. F. Scanlon, K. E. Webb, Jr., and J. B. Whelan. 1982. Energy, protein, and seasonal influences on white-tailed deer fawn nutritional indices. J. Wildl. Manage. 46:302-312.

Wood, A. K., R. E. Short, A. Darling, G. L. Dusek, R. G. Sasser, and C. A. Ruder. 1986. Serum assays for determining pregnancy in mule and white-tailed deer. J. Wildl. Manage. 50:684-687.

Table 1. History of radiocollared gray wolves, north central Minnesota, 1993-2004. (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-NATURAL CAUSES, EMACIATED, MANGEY	AUG 2, 1998
2062	SOUTH INGY	AUG 1997	F	AD	SHOT	FEB 1998
2089	SHINGLE MILL	MAY 1993	F	AD	KILLED BY WOLVES	SEP 1994
2050	SHINGLE MILL	MAY 1993	M	AD	COLLAR CHEWED OFF	AUG 1993
2095	SHINGLE MILL	MAY 1995	F	AD	LOST SIGNAL	NOV 1995
2064	SHINGLE MILL	AUG 1996 MAY 2004	F	JUV	ON THE AIR	MAY 2004
2060	SHINGLE MILL	AUG 1996	F	JUV	LOST SIGNAL	FEB 1, 2000
2059	SHINGLE MILL	AUG 1996 JUL 1998 - RECAPTURED	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

Table 1 (continued). History of radiocollared gray wolves, north central Minnesota, 1993-2004. (Ad=adult, juv=juvenile).

Wolf No.	Pack	Capture Date	Sex	Age Class	Fate	Date
2257	E. DIRTY NOSE	MAY 1999	M	AD	LOST SIGNAL	JAN 14, 2001
2258	WILLOW L	AUG 1999	M	AD	DISPERSED	MAR 16, 2000
2259	DIRTY NOSE	JULY 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 JUN 2003 - RECAPTURED	F	AD	DISPERSED	MAR 17, 2004
2141	INGUADONA	SEP 2002	F	JUV	DROPPED TRANSMITTER	SEP 22, 2002
2149	INGUADONA	MAY 2003	M	AD	SHOT	NOV 2003
2143	WILLOW	MAY 2003	M	AD	ON THE AIR	
2144	MORRISON BROOK	JUN 2003	F	AD	ON THE AIR	
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	ON THE AIR	

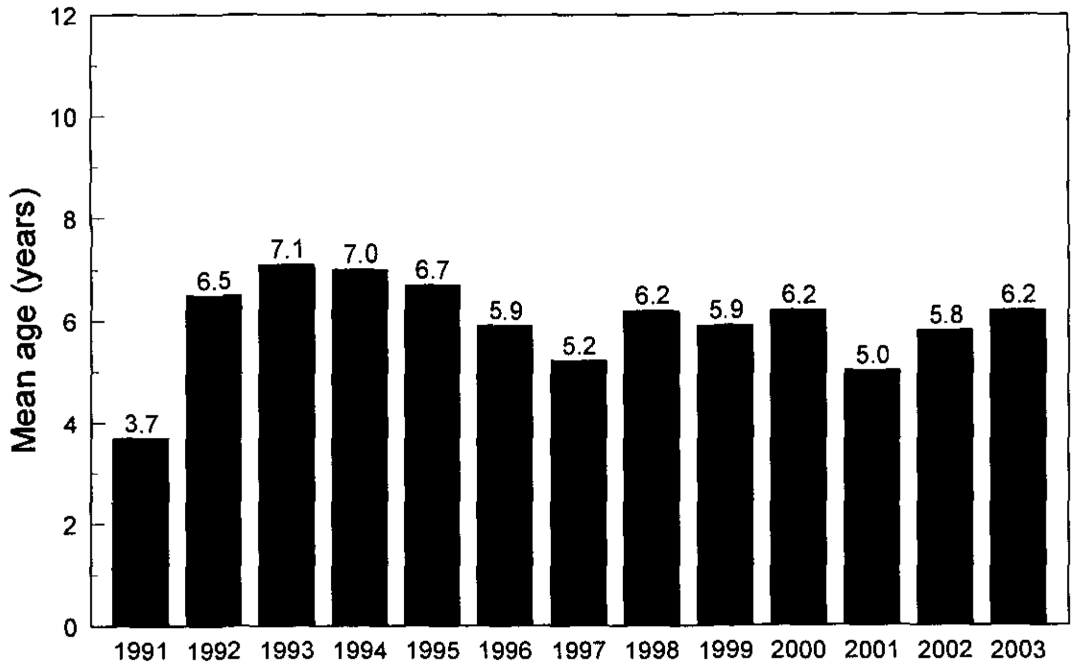


Figure 1. Mean age of radiocollared female white-tailed deer among years, north central Minnesota, 1 January 1991-31 December 2003. (Sample sizes were 22, 34, 62, 66, 54, 76, 74, 49, 55, 48, 89, 83, and 74, respectively.)

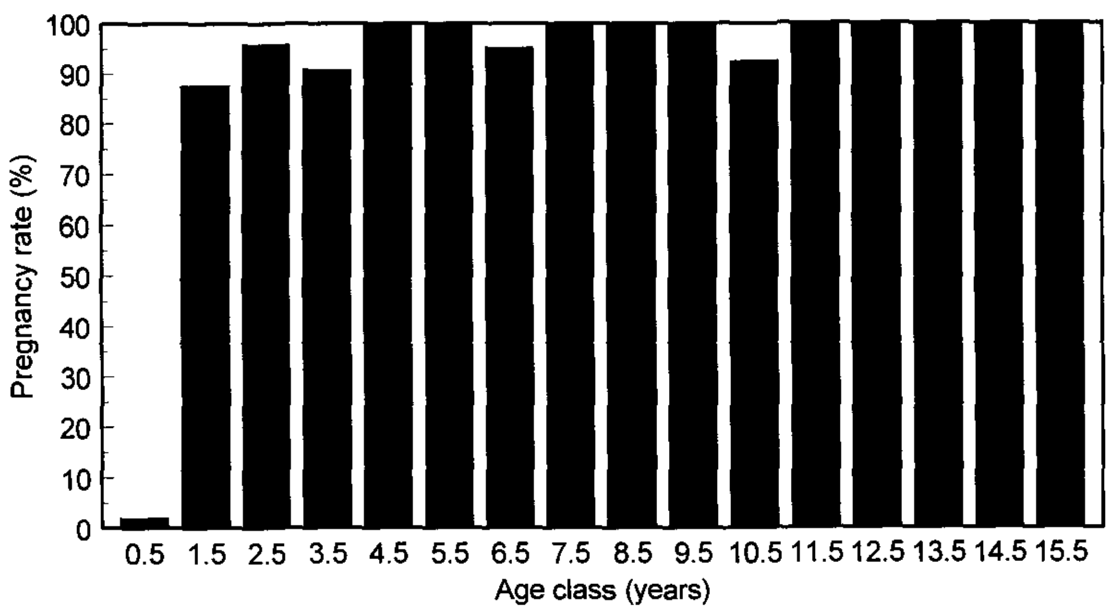


Figure 2. Age-specific pregnancy rate of radiocollared white-tailed deer (4 study sites pooled) in north central Minnesota, winters 1991 to 2002. (Sample sizes were 55, 48, 23, 21, 18, 21, 20, 13, 9, 11, 13, 8, 11, 5, 4, and 4 for yearly age classes, respectively.)

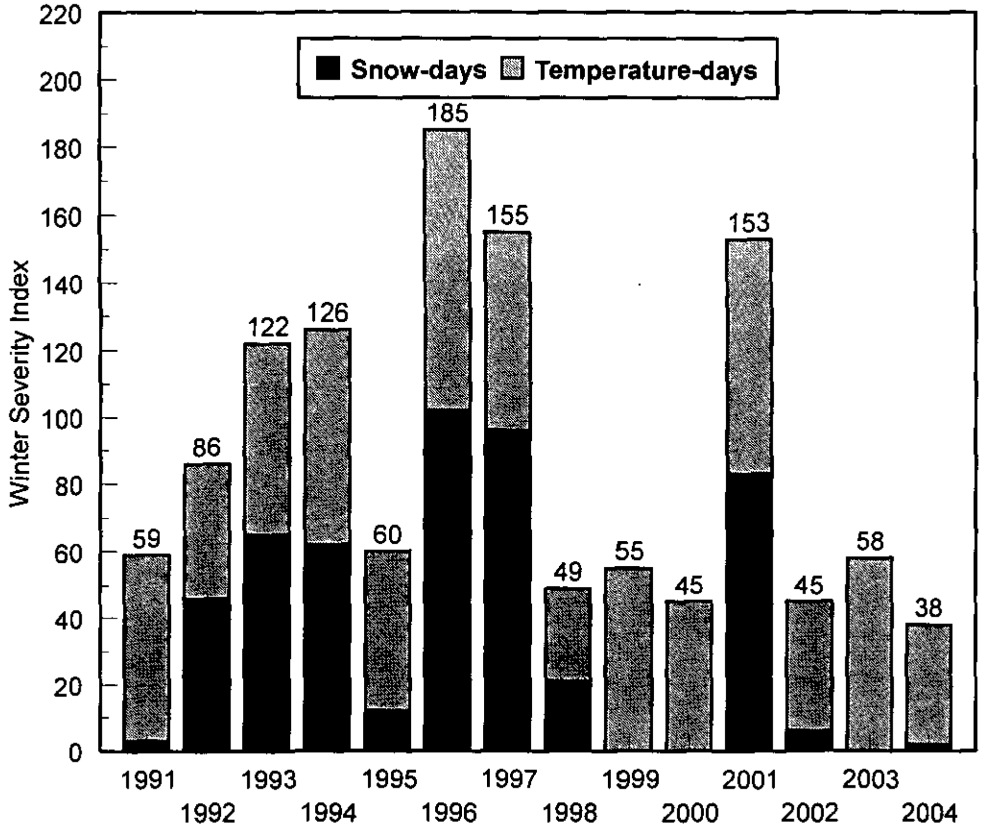


Figure 3. Winter severity index for white-tailed deer study sites, north central Minnesota, winters 1990-91 to 2003-04. One point is accumulated for each day with an ambient temperature $\leq$ -17.8° C (temperature-day), and an additional point is accumulated for each day with snow depths $\geq$38.1 cm (snow-day).

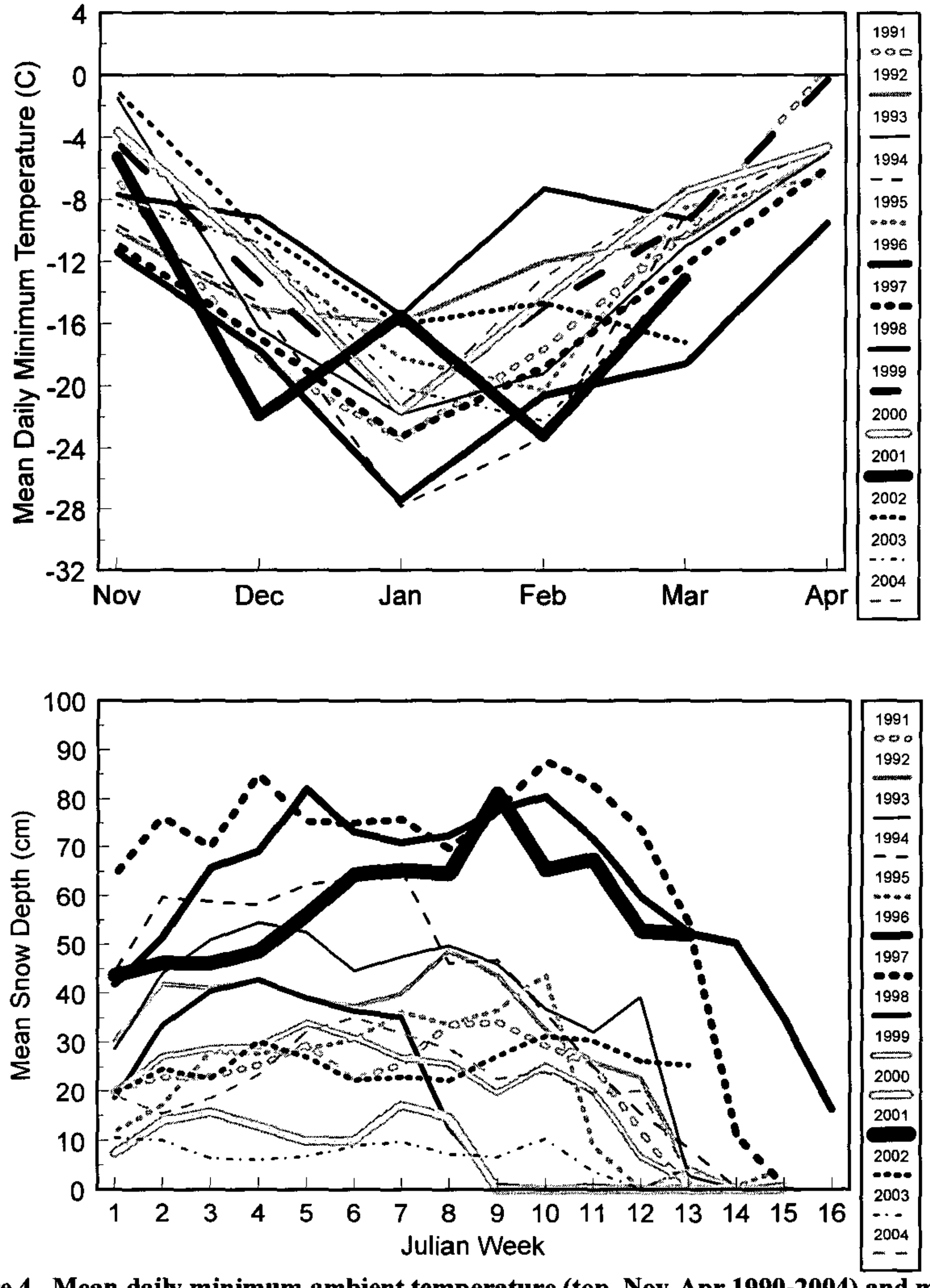
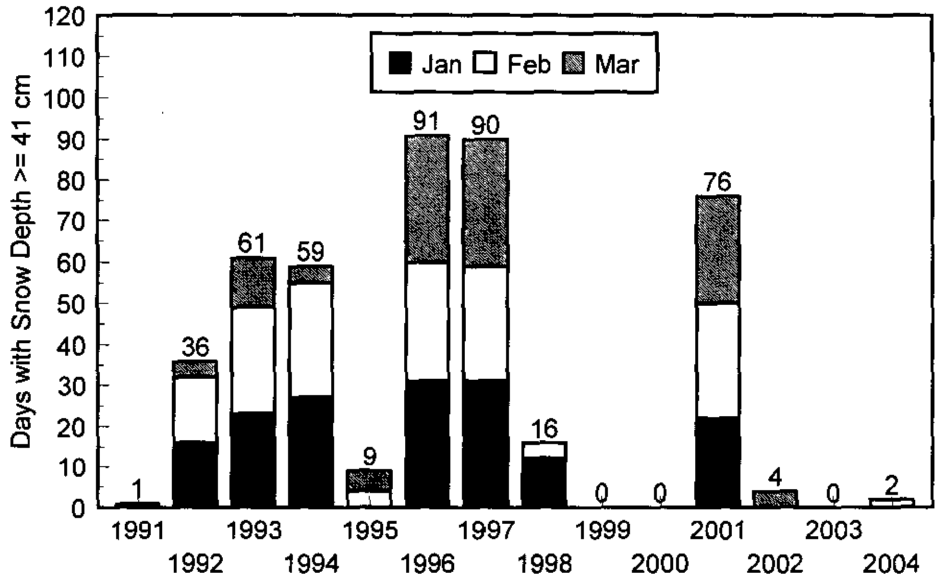


Figure 4. Mean daily minimum ambient temperature (top, Nov-Apr 1990-2004) and mean weekly (julian) snow depths (bottom, Jan-Apr 1991-2004) for white-tailed deer study sites, north central Minnesota.



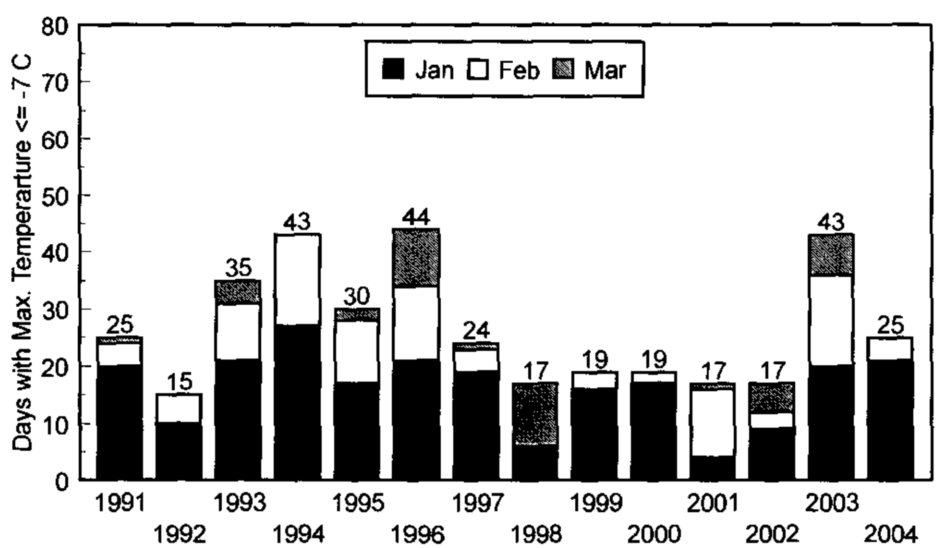


Figure 5. Number of days with snow depths ≥41 cm (top) and maximum ambient temperatures ≤ -7° C (bottom, *effective critical temperature* for an average size doe [60 kg]), north central Minnesota, January-March 1991-2004.

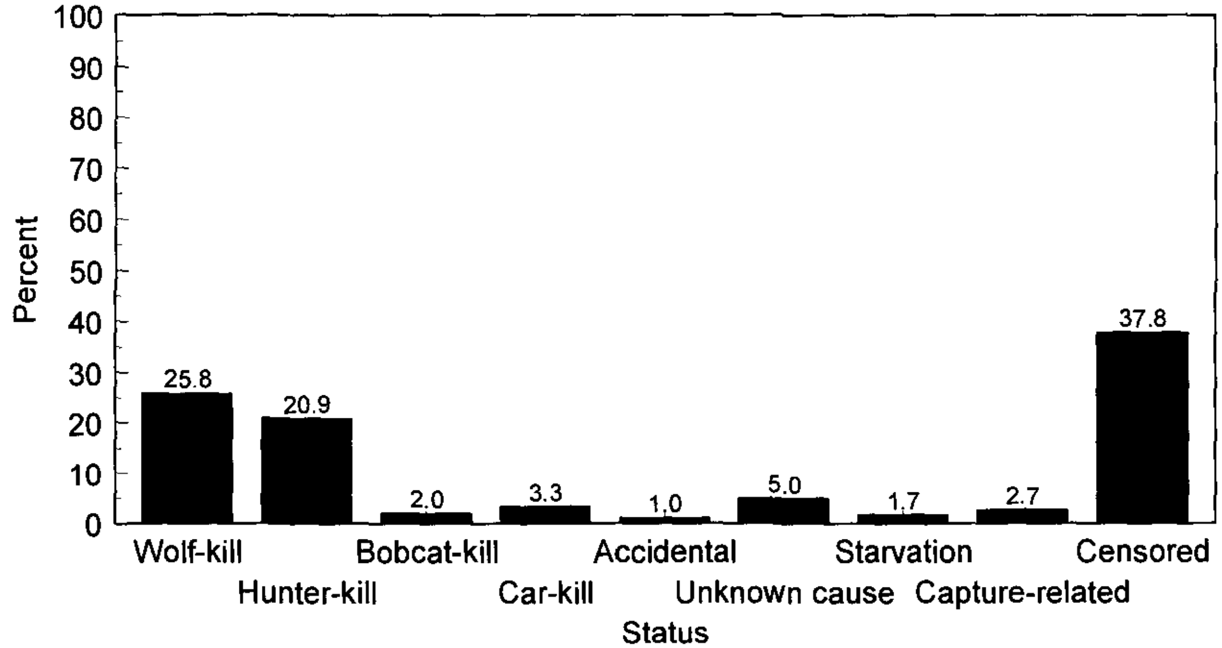


Figure 6. Status of radiocollared female deer, north central Minnesota, January 1991-December 2003. Censored deer include those that were still alive on 31 December 2003, or whose radio signals have been lost to monitoring (e.g., radio failure, dispersal from region of the study sites).

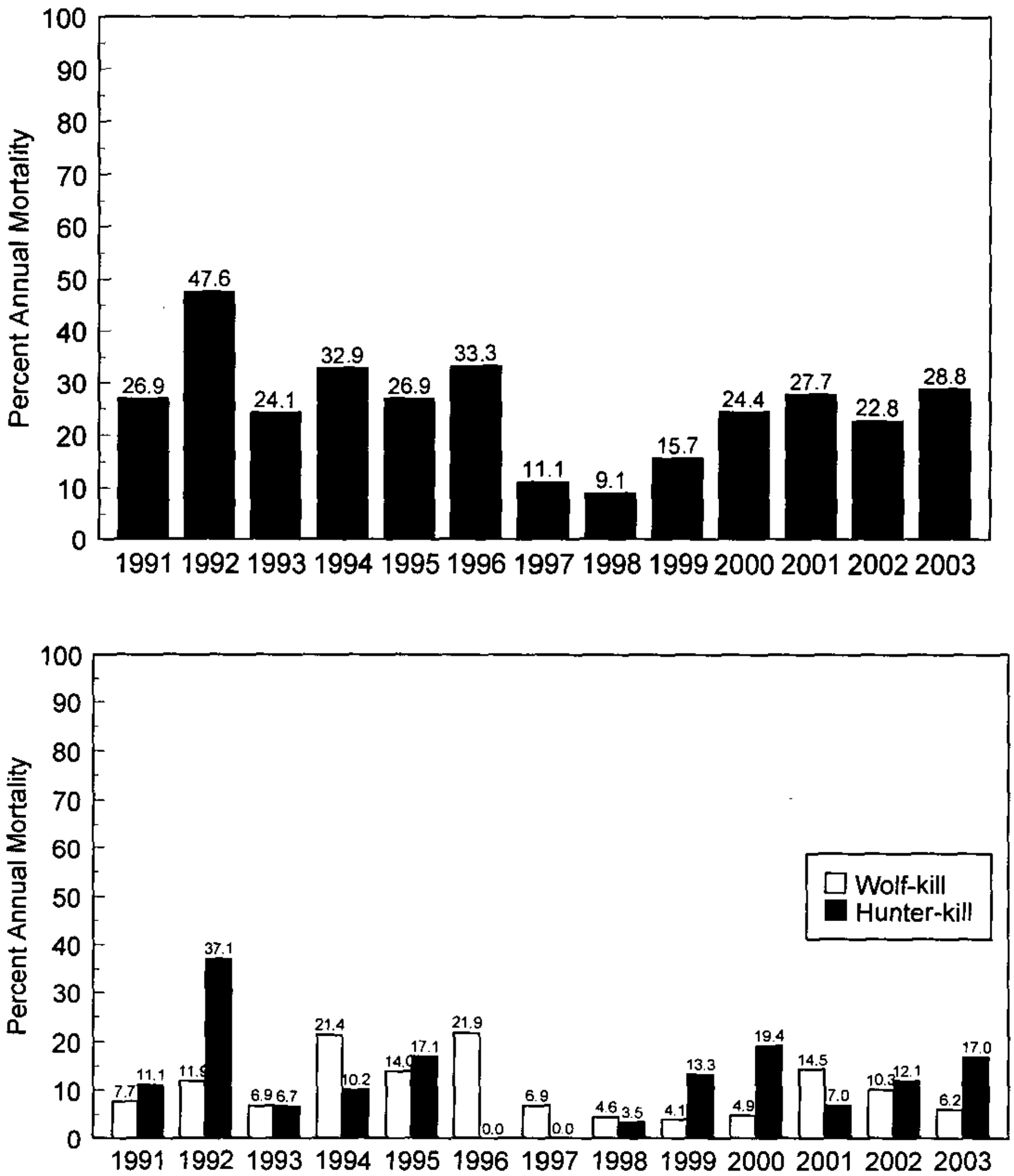


Figure 7. Annual (1 Jan-31 Dec) percent mortality of radiocollared, female white-tailed deer (top) and annual percent mortality attributable to wolf predation and hunter harvest (bottom, 4 sites pooled), north central Minnesota, 1991-2003. (Sample sizes were 26, 42, 58, 70, 52, 66, 72, 44, 51, 41, 83, 79, and 66, respectively. Hunter harvest was calculated with the maximum number of collared females entering Nov; no antlerless permits were issued in 1996 and 1997, and very few were issued in 1998.)

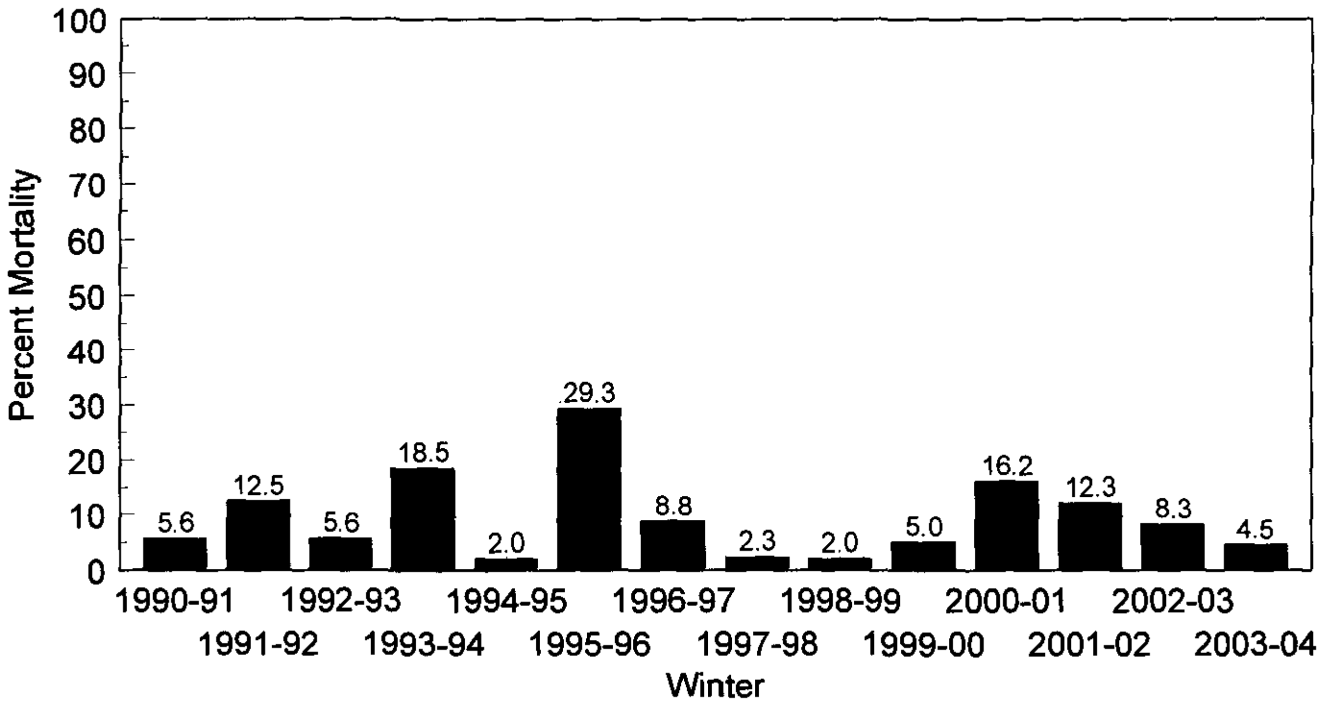


Figure 8. Percent winter mortality (Dec-May) of radiocollared, adult (≥1.0 year old) female white-tailed deer (4 sites pooled), north central Minnesota, winters 1990-91 to 2003-04. (Sample sizes were 18, 40, 54, 65, 50, 58, 68, 43, 49, 40, 68, 73, 60, and 67, respectively; no deer were radiocollared during Dec 1990.)

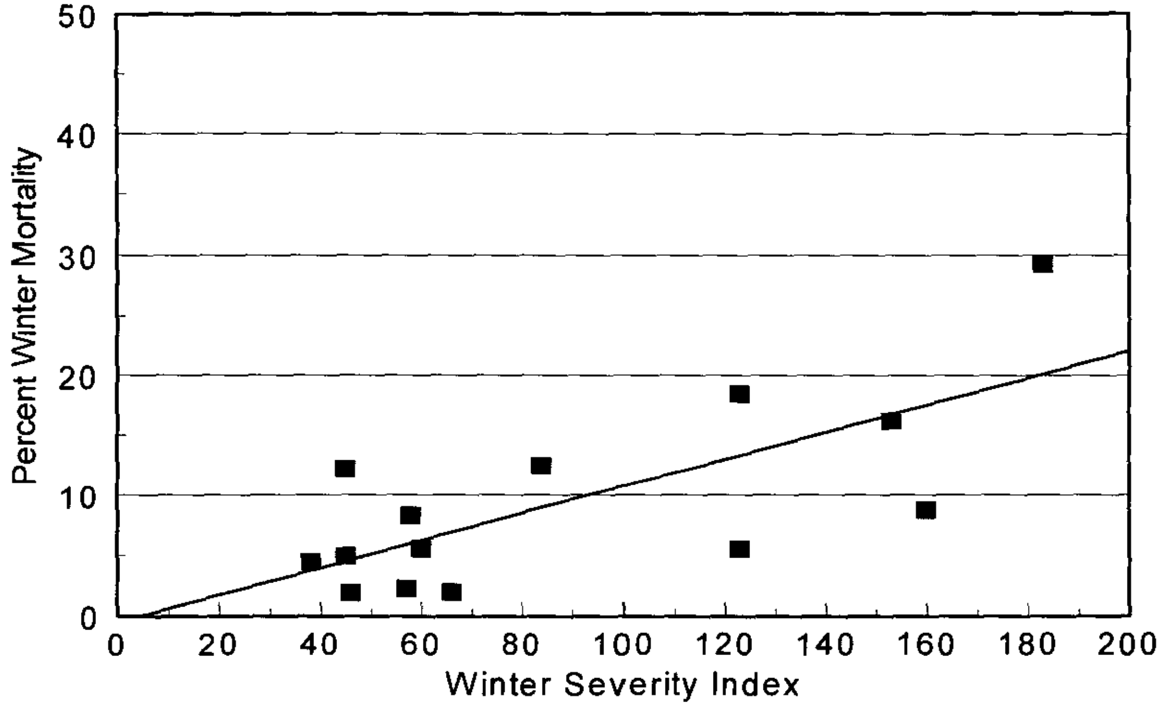


Figure 9. Relationship between MNDNR winter severity index (Nov-May) and percent winter (Dec-May) mortality ($Y = -0.0050 + 0.0011x$, $r^2 = 0.52$, $P = 0.004$) of radiocollared, adult (≥1.0 year old), female white-tailed deer (4 sites pooled), north central Minnesota, winters 1990-91 to 2003-04.

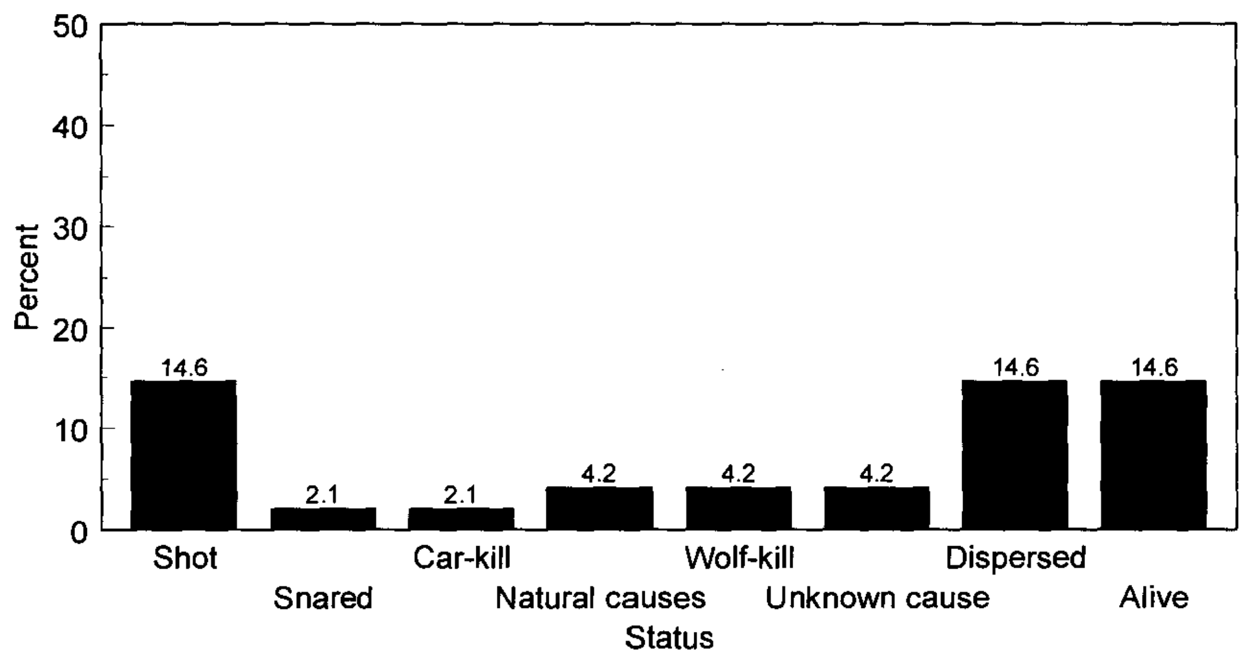


Figure 10. Status of radiocollared wolves, north central Minnesota, 1993-2004.

USING DOE BEHAVIOR AND VAGINAL IMPLANT TRANSMITTERS TO CAPTURE NEONATE WHITE-TAILED DEER IN NORTH CENTRAL MINNESOTA

Michelle Carstensen, Glenn D. DelGiudice, and Barry A. Sampson

Direct study of the survival and causes of mortality of neonates of deer (*Odocoileus* spp.) inhabiting forests has been limited because of the difficulty of locating newborns in dense vegetation. We compared our efforts and success in locating and capturing white-tailed deer (*O. virginianus*) neonates in the forest zone of Minnesota using movement behavior of radiocollared does versus radiocollared does fitted with vaginal implant transmitters. Using doe behavior, we located 25 fawns in springs 1997, 1999, and 2000 combined. Almost 100 people investing 5,310 person-hours (1,500–1,890 per year) were involved in search efforts using this doe behavior; average crew size was 5–8 people. The success rate (i.e., capture of ≥1 neonate) was 12 and 21% for 68 and 43 searches, respectively, during springs 1999 and 2000. During winter 2000–2001, we fitted 25 pregnant does with vaginal implant transmitters, which resulted in a capture success rate of 88% (14 of 16 searches) and 20 radiocollared neonates in spring 2001. We captured an additional 11 fawns using doe behavior. This season's total search efforts included 29 people investing 1,872 person-hours. The required effort-per-unit-capture was 2.4–3.5 times more using doe behavior than using vaginal implant transmitters to capture neonates. We recovered 16 of the 25 implants at birth-sites, 3 were inactivated or lost due to predation before fawning, 2 were expelled prematurely, and 4 were lost due to transmitter failure. Problems with the implant transmitters included battery failure and fluctuating pulse rate of signals (i.e., slow versus fast) while still in the doe. Though not perfected, the vaginal implant transmitter proved to be an effective tool for locating birth-sites and capturing neonates of a widely dispersed deer population inhabiting a densely vegetated landscape.

*Abstract of paper published in the Wildlife Society Bulletin 31: 634-641.

IMPROVING OUR UNDERSTANDING OF THE MARGINS OF SAFE CAPTURE, CHEMICAL IMMOBILIZATION, AND HANDLING OF FREE-RANGING WHITE-TAILED DEER

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

Improved understanding of the margins of safe capture, chemical immobilization, and handling of free-ranging animals for research and management relies on the documentation and examination of efforts involving various species, study designs, and environmental conditions. During 1991–2002, we captured white-tailed deer (*Odocoileus virginianus*), primarily by Clover trap, under a wide range of winter weather conditions and in an area saturated with wolves (*Canis lupus*). Our research objectives required prolonged immobilization times (mean = 96–98, range 5–243 minutes). With 984 captures and recaptures of females and males, "capture accidents" (e.g., trauma-induced paralysis, death) were limited to 2.9%. Focusing on 652 of the 984 captures and recaptures, involving 337 radiocollared females (0.5–15.5 years old), the incidence of capture accidents remained at 2.9%, and mortality that occurred within 14 days of release was 2.5%. Mean time to mortality for this latter group of individuals was 6 days (SE = 1.1 days, $n = 16$); wolf predation within 11 days was the proximate cause of 50% of these mortalities. A priori, we selected immobilization time for analysis by logistic regression to test for a potential effect of capture and handling on mortality, while controlling for known risk factors (age, winter severity), but found no significant effects. Additionally, subsequent comparisons of means and 95% confidence limits showed no differences between numerous aspects of the capture, immobilization, handling, or associated weather conditions. Relative success in capture and handling of free-ranging deer means smaller sample sizes of capture-related deaths ("events"), which makes it difficult to infer causal relations between environmental variables, handling procedures, and capture-related mortality. The strength of such studies is that they may serve to demonstrate a range of conditions (environmental variables and handling procedures) over which capture-related mortality can be controlled at acceptably low levels.

*Abstract of paper accepted by the Wildlife Society Bulletin.

COMPARISON OF VEGETATION TYPE SELECTION BY FEMALE WHITE-TAILED DEER DURING WINTER USING TWO METHODS OF ANALYSIS

Carolin A. Humpal, Robert G. Wright, and Glenn D. DelGiudice.

Radiotelemetry can be useful for investigations of use and selection of vegetation types. When radio-locations are treated as precise locations, rather than estimates, results may be inaccurate and misleading. Telemetry error can be addressed by analytical methods by generating a subsample of points from the error distribution of each estimated location. During January-March 2001 and 2002, 20 adult female white-tailed deer (*Odocoileus virginianus*) were located 3-7 times weekly using ground-based radiotelemetry with simultaneous triangulation on 2 study sites within Camp Ripley Army National Guard Training Site in central Minnesota. Ten vegetation types, based on dominant cover, were delineated, digitized, and one assigned to each point location and subsampled point. We used χ^2-goodness-of-fit tests and Bailey's confidence intervals to determine selection using both point locations and subsampled points for each location at both the study site and homerange levels of availability. Use of subsampled points increased the number of vegetation types that showed selection by an average of 7.8 and 5.7 types for the study site and home range levels of availability, respectively. However, when the point method showed selection, there was agreement between methods 94.5% and 98.3% of the time for the homerange and study site levels. Generating a subsample of points from the error distribution of each estimated location enables researchers to account for the imprecise nature of radiotelemetry and increase power when investigating vegetation selection.

*Presented at The Wildlife Society's Tenth Annual Conference, Burlington, Vermont.

BIRTH, MORPHOLOGICAL, AND BLOOD CHARACTERISTICS OF FREE-RANGING WHITE-TAILED DEER NEONATES

Michelle Carstensen and Glenn D. DelGiudice

Winter severity is a primary factor influencing deer survival and reproduction in northern climates. Prolonged, harsh winters can adversely affect body condition of does, resulting in depressed morphological development of neonates. In this study, we captured 59 white-tailed deer (*Odocoileus virginianus*) neonates (28 and 31 in 2001 and 2002), following two distinctly different winters; one severe and the other historically mild. Exact age was determined for 73% of neonates. Birth-date and morphological measurements of neonates (i.e., birth-weight, new hoof growth, hoof length) were compared by gender and capture year. For known-age neonates ($n = 43$), there was a year by sex interaction effect ($P = 0.01$) on birth-date and capture-date, both being later for females and earlier for males during spring 2001 (females: 153.1 julian day [$\pm$ 2.5], SE) and 153.2 [$\pm$ 2.5]; males: 146.0 [$\pm$ 2.0] and 146.4 [$\pm$ 1.9]) compared to 2002 (females: 145.9 [$\pm$ 2.0] and 147.8 [$\pm$ 1.9]; males: 151.4 [$\pm$ 2.5] and 152.8 [$\pm$ 2.3]). A significant ($P = 0.03$) year by sex interaction was also determined for total hoof length (22.3 mm [$\pm$ 0.9] and 20.3 [$\pm$ 0.8] for females and males in 2001; 19.9 [$\pm$ 1.0] and 22.1 [$\pm$ 1.0] for females and males in 2002). Interestingly, there was no effect of year on birth-weight or birth-date of known-age neonates. A year by sex interaction ($P = 0.04$) was determined for birth–dates of estimated age neonates ($n = 16$), with females born earlier than males in 2001 and later than males in 2002. Capture year also had little reportable effect on 20 hematological and serum characteristics that were examined; however, there were significant sex effects on red blood cell (RBC) counts ($P = 0.04$; females, 8.3 $10^6/\mu l$ [$\pm$ 0.3] and males, 7.4[($\pm$ 0.2]), serum cholesterol ($P = 0.01$; females, 46.7 mg/dl [$\pm$ 3.5] and males, 34.6[$\pm$ 3.4]), cortisol ($P = 0.01$; females, 7.6 µg/dl [$\pm$ 0.8] and males, 11.4 [$\pm$ 1.3]), and a year by sex effect ($P = 0.04$) on triglyceride (38.8 mg/dl [$\pm$ 11.5] and 167 [$\pm$ 0.0] for female and male neonates in 2001; 55.5 [$\pm$ 31.6] and 55.7 [$\pm$ 16.3] for female and male neonates in 2002). Mean corpuscular volume (MCV) was the only blood characteristic that differed ($P < 0.01$) between years, with higher values occurring in 2001. Also, we report a range of reference values for blood constituents that have not been previously documented for free-ranging neonates. Remarkably, winter severity appeared to have little overall effect on birth, morphological development, and blood characteristics of neonates in this study. Our findings suggest caution should be exercised when applying physiological models derived in captivity to free–ranging deer populations.

*Abstract of a paper accepted by the Journal of Wildlife Diseases.

RISK OF MARKING-INDUCED ABANDONMENT MAY BE MINIMAL FOR FREE-RANGING WHITE-TAILED DEER NEONATES

Michelle Carstensen, Glenn D. DelGiudice, and Barry A. Sampson

Marking-induced abandonment has been suggested as the most common cause of marking-induced mortality of free-ranging, newborn ungulates in North America. However, there has not been any direct study of marking-induced abandonment in free-ranging ungulates, and its relevance to neonate survival is inconclusive. We describe our capture, marking and subsequent monitoring of white-tailed deer (*Odocoileus virginianus*) neonates born to radiocollared dams in north central Minnesota over 5 springs (1997, 1999–2002), as it may relate to marking-induced or natural abandonment. We assumed that all neonates dying within 4 days post-marking had possibly been predisposed to abandonment. We captured 89 neonates; 6 (7%) died within 4 days (4 to predation, 2 to unknown causes). We found no conclusive evidence of marking-induced abandonment during this study. Duration of handling had no apparent impact on neonate survival, even though 40% of neonates were handled >15 minutes. The time of marking (post-parturition) was similar among survivors and nonsurvivors; 48% of the surviving neonates were captured < 24 hours of birth. Eleven neonates (12%) were chased prior to capture and all survived >4 days post-capture. Serum cortisol levels appeared unaffected by chase, but nonsurviving neonates had higher cortisol concentrations than survivors at the time of their capture (12.0 ± 4.7 [SE] and 7.5 ± 0.5 µg/dL). Hematology was similar among all neonates, as were serum indices of nutritional restriction and body fat content of dams of surviving and nonsurviving neonates. Dams of nonsurviving fawns were older than dams of survivors (8.5 ± 2.1 versus 5.6 ± 0.4 years). Neonates traveled a mean distance of 162 ± 8 m from their capture site < 4 days post-marking, and 76% of all radiocollared dam locations (n = 245) were < 200 m from their neonates. Mean distance traveled between capture and mortality sites for nonsurviors (159 ± 69 m, n = 6) was similar to the mean travel distance of the surviving, radio-tracked neonates (162 ± 8 m, n = 18) within the first 4 days post-marking. Our findings suggest the risk of marking-induced abandonment in white-tailed neonates is minimal, regardless of the time and duration of handling. We caution against the omission or censoring of suspected cases of marking-induced abandonment in white-tailed neonate survival studies, as this may underestimate natural mortality rates.

*Abstract of paper accepted by the Wildlife Society Bulletin.

WINTER SEVERITY, DEER NUTRITION, AND FAWNING CHARACTERISTICS

Michelle Carstensen and Glenn D. DelGiudice

The primary objective of this study is to survey survival of free-ranging, white-tailed deer (*Odocoileus virginianus*) neonates and examine its relation to winter severity, nutritional restriction (i.e., dietary), and condition (i.e., body composition) of the does during the prior winter. A principal question concerns how variable fawn survival is among years and whether their survival rate is predictable relative to severity of the winter previous to their birth. A secondary objective is to examine physiological responses of deer to natural, winter nutritional restriction and deteriorating body condition through *in vivo* body composition techniques, as well as by blood analysis, relating patterns of winter nutritional restriction and condition to subsequent reproductive success.

BACKGROUND

Winter Nutritional Condition of Does and Survival of Fawns

Winter in northern Minnesota (Nov-Apr) is the most nutritionally challenging season for white-tailed deer, which can strongly impact their population performance through survival and reproductive success (Mech et al. 1971; Nelson and Mech 1986; DelGiudice et al. 1991, 2002). The study of seasonal changes in nutrition and physical condition of deer and other cervids has received increasing effort involving both captive and free-ranging animals. Investigators have studied numerous indices of nutritional status, but there is almost no information on the specific changes of body composition of *free-ranging* animals relative to winter nutritional restriction.

Previous studies have examined the effects of protein and energy malnutrition and nutritional restriction on body mass and composition, physiological profiles, and reproductive success of deer and other cervids (Verme 1969; DeCalesta et al. 1977; Holter et al. 1977; Seal et al. 1978; Bahnak et al. 1979; Verme and Ozoga 1980; Warren et al. 1982; Watkins et al. 1982; DelGiudice et al. 1987, 1990). However, the majority of these data have been collected on captive animals, thus excluding the critical influences of *natural* energy and activity budgets and *natural* diets on assessed responses. Researchers have attempted to elucidate the relationship between fat reserves (and range quality) and fertility of free-ranging cervids, but information relating body composition to reproductive success is scant. According to studies on captive white-tailed deer, does fed a low plane of nutrition (similar to what might be expected during a severe northern winter) had fewer incidences of twinning, lower fawn birth weights, and prolonged gestational periods (Verme 1969, Verme and Ozoga 1981).

Further, little research has included examination of the potential effects of environmental variation, particularly winter severity, on body composition and reproductive success of does, or on subsequent survival of their fawns. Such studies of reproductive performance and survival provide insights concerning the influence of individual characteristics (age, condition) and extrinsic factors (resource availability, weather) on reproductive success of individuals and on the variation of productivity of populations. In this study, we closely examine the relation between winter severity, nutritional restriction, body composition and reproductive success of free-ranging, female white-tailed deer, and the survival of their fawns.

STUDY AREA

The study area for this research consists of 4 trapping sites (located between 46°52' and 47°15'N latitude and 93°45' and 94°07' W longitude) along the eastern and southern boundaries of the Chippewa National Forest in north central Minnesota. The sites range in size from 10 to 19 km^2. The physiography and habitat of these sites are very similar. Topography is undulant with elevations of 400-475 m. Deciduous and mixed coniferous-deciduous stands are associated primarily with the uplands, and conifer swamps predominate in the lowlands.

METHODOLOGY

Weather Data Collection

As part of a larger deer/winter thermal cover study (see DelGiudice, this Research Summary), winter severity was assessed by daily measurements of minimum and maximum ambient temperatures in openings (i.e., forest clearings) and dense conifer stands on the study area. Snow depth and penetration (index of snow density) were measured to the nearest centimeter in 27 locations (3 measurements along each of 3 transects in openings, mixed hardwood, and dense [≥70% canopy closure] conifer stands) on the study area.

Deer Capture, Handling, and Body Composition Determination

During winters 2000-01 and 2001-02, adult female white-tailed deer and fawns (male and female) were captured by Clover traps (55) and rocket net within the boundaries of the 4 study sites. These deer were injected intramuscularly with 1.4 mg xylazine HCl and 4.3 mg ketamine HCl per kg body mass, and handling included ear-tagging, extracting a last incisor for aging by cementum annuli, radiocollaring (primarily standard VHF collars, but 5 global positioning system [GPS] collars as well), weighing, blood- and urine-sampling, monitoring of body temperature, and morphological measurements. Body composition (i.e., water) of does and fawns was determined *in vivo* by intravenous injection of deuterium following a baseline blood sample; serial blood samples were collected out to 120 minutes post-injection to permit assessment of isotope equilibration. Predictive equations were employed to estimate body fat, protein, and ash. Additionally, 2 new techniques were used to assess body condition (i.e., fat reserves) of deer. Rump fat was measured with a portable real-time ultrasound device (Sonovet 600, Universal Medical Systems, Bedford Hills, NY). Previous studies using this technique to estimate body fat on cervids have reported encouraging results (Stephenson et al. 1998, Cook et al. 2001). This study was the first to use ultrasonography on white-tailed deer. Secondly, a body condition scoring (BCS) system was implemented. This involved palpation of the withers, rib and rump areas to assess deer condition. Pregnancy status of captured deer was determined in the field by portable dop-tone ultrasound, and pregnancy was confirmed in the laboratory by serum progesterone concentrations ≥1.8 ng/ml. Vaginal transmitter implants (Advanced Telemetry

Systems, Inc., Isanti, MN) were inserted into 50 pregnant does. These implants were expelled during fawning, and the pulse rate increased from 40 to 80 beats per minute when the transmitter was exposed to an ambient temperature < 95° F. Immobilizations were reversed by intravenous injection of 0.2 mg/kg yohimbine HCl.

Analytical Procedures

Extensive profiles of blood (hematology; serum chemistries, electrolytes, metabolic and reproductive hormones) were determined to assess each animal's overall health and metabolic status. Blood samples also were analyzed in the laboratory for deuterium concentration. Additionally, 15 deer (7 adult females, 4 female fawns, 4 male fawns) were euthanized following completion of the deuterium-dilution protocol to allow for direct chemical analysis of body composition. Carcass components were ground and homogenized. Subsamples of the homogenized mixtures were analyzed for dry matter (water derived), crude protein (macro-Kjeldahl N x 6.25), ash (combustion at 600°C for 12 hours), and fat (ether extract). Predictive equations derived from this chemical analysis were used to validate the accuracy of deuterium-dilution determinations of body composition.

Neonate Survival and Cause-specific Mortality

Constriction of spring-early summer home ranges or change in the pulse rate of the vaginal transmitter implants of does was used to determine if they had fawned. Neonates were captured, aged (in days) by hoof growth, weighed, blood-sampled, radiocollared, and released. Rectal temperatures also were recorded. Survival of all radiocollared fawns was monitored daily during the summer and 2-3 times per week throughout the following year via mortality switches built into the radio collars (Advanced Telemetry Systems, Inc., Isanti, MN). All deer mortalities were investigated < 24 hours of detection by a field crew to determine cause of death.

RESULTS

Deer Capture and Determination of Body Composition (January-March 2001 and 2002)

The 2 winters of this study were severe (2000–01) and historically mild (2001–02). To gauge winter severity, the Minnesota Department of Natural Resources (MNDNR) calculates a winter severity index (WSI) by accumulating 1 point for each day with an ambient temperature ≤ −17.8°C and 1 point for each day with snow depth ≥38 cm during November 1–May 31. The maximum WSIs for winters 2000–01 and 2001–02 were 153 and 45, respectively.

We estimated total body water by deuterium-dilution for 48 deer (24 adults, 24 fawns) in 2001 and 28 deer (15 adults, 13 fawns) in 2002. Using the predictive equations generated from the sacrificed deer, we calculated ingesta-free body composition for 39 adults and 37 fawns (18 females, 19 males). Mean total body 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%), which was accompanied by declines in body protein mass (0.6 and 1.0 kg for adults and 1.6 and 0.2 kg for fawns in 2001 and 2002, respectively). Interestingly, body 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 body fat (%) of adults, and serum urea nitrogen (in combination with julian day) was inversely related ($R^2 = 0.36$) to body fat (%) of fawns.

We also assessed the validity of using ultrasonic measurements of subcutaneous rump fat and body condition scoring to predict body fat reserves of deer. To determine if the ultrasonic measurements were predictive of body condition, we assessed their linear relationship to percent body fat. In adults, MidFat was most closely related ($r^2 = 0.71$, $P = 0.07$, $n = 5$) to percent body fat, followed by loin muscle ($r^2 = 0.66$, $P = 0.05$, $n = 6$). However, in fawns, MaxFat was the strongest predictor ($r^2 = 0.64$, $P = 0.05$, $n = 6$) of body fat, followed by SumFat ($r^2 = 0.53$, $P = 0.10$, $n = 6$) and loin muscle ($r^2 = 0.45$, $P = 0.10$, $n = 7$). Body condition was scored for 19 adults (5 and 14 in 2001 and 2002) and 15 fawns (5 and 10 in 2001 and 2002) during mid-late winter. In adults, Rump score was most closely related ($r^2 = 0.63$, $P = 0.01$, $n = 9$) to percent body fat, but in fawns, the Withers score was most strongly related ($r^2 = 0.45$, $P = 0.07$, $n = 8$) to body fat. Including all ultrasonic measurements and BCS components as predictors of body fat (%), Rump score alone was the best predictive model (y = 5.03 + 0.58[Rump score], $r^2 = 0.81$, $P = 0.05$, $n = 5$) for adults. In the fawn group, Rib score, in combination with MaxFat (y = 2.67 + 0.80[Rib score] – 0.16[MaxFat], $R^2 = 0.88$, $P = 0.03$, $n = 7$), was the best predictive model.
Survival of adult females was similar between years, with the majority of deaths occurring between February and April; wolf predation was the primary cause of predator-related deaths (Table 1). 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, as nearly half (47%) of fawns died during late-winter or early-spring 2001, compared to 100% survival of fawns in 2002. Fat reserves were not reliable predictors of survival in this study as nearly 80% and 86% of adults and fawns that were determined to have low fat reserves in mid- or late-winter (i.e., below the median body fat percentage of all animals sampled) survived winter and early spring. Absence of a biologically significant difference between winter body condition of deer in this study relative to winter severity may be a result of a cumulative effect of several mild winters preceeding 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 that encompasses several winters of varying severity may be warranted, as well as consideration of other potentially influential factors (e.g., migration behavior, predator density, habitat quality).

Newborn Fawn Capture and Survival

In early May 2001and 2002, does with vaginal implant transmitters (50) were monitored daily for a change in pulse rate, indicating the implants were expelled during parturition. Seventy-six percent (38) of implants remained active at the beginning (May) of these fawning seasons, 74% (28) of which led to the capture of 41 neonates (20 and 21 in 2001 and 2002). We were unable to monitor 12 implanted does by May due to premature expulsion of the implant (2 does in 2001) or predation of the doe (3 and 7 does in 2001 and 2002). In addition to the capture of neonates, 31 birth-sites were discovered (17 in 2001 and 14 in 2002). A technical advancement of the implants allowed us to document the exact time of parturition for 13 does in 2002; the majority of births (70%) occurred between 1200-1800 hours. Use of vaginal implant transmitters markedly increased our ability to efficiently and successfully locate and capture neonates; however, we also discovered the implants can be problematic (i.e., including battery failure, fluctuating signals, and monitoring schedules). An additional 25 neonates were captured during springs 2001 and 2002 using doe behavior alone.

The mean date of birth was 26 May (± 1.6 [SE] days, range = 5 May-19 June) for neonates captured in 2001 and 2002. Mean dates of birth and body masses were similar between neonates captured in springs 2001 (n = 31) and 2002 (n = 35) (Table 2). Neonate survival was similar between years; pooled mortality rates for neonates were 14, 25, and 45% at 0–1, 1–4, and 4–12 weeks of age, respectively. Predation accounted for 86% of mortality, whereas the remaining 14% of deaths were attributed to unknown causes (Table 3). Black bears (*Ursus americanus*) were responsible for 57 and 38% of predator-related deaths of neonates in springs 2001 and 2002; whereras, 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) ratios were related to neonate survival to 1, 4, and 12 weeks of age; with elevated values 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, creatinine, and SUN:C ratios 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 indicated that body mass at birth and key serum indices of neonate nutrition were associated with their survival. Of even greater significance, we were able to link winter severity and nutritional restriction of dams to reduced survival of their offspring.

UPCOMING STUDY RESULTS

Investigation into the importance of fawning-site characteristics (e.g., vegetative cover, predator pressure) and spatial relationships between does and their newborns on overall fawn survival was a new component added to the study during spring 2001. Little is known about birth-site selection by does in northern Minnesota. Location of birth-sites may play a role in neonatal mortality, particularly relative to the risk of predation. To maximize protection of fawns from potential predators, frequency and duration of contact between does and their fawns may be inversely related to the fawn's vulnerability, and the level of predator pressure may influence doe-fawn spatial relationships. To gain a better understanding of doe-fawn behavior and ultimately fawn survival, we began to characterize birth-site habitats and assess the spatial relationships between doe-fawn pairs from birth to 2-3 weeks post-parturition, which is typically how long it takes for fawns to be moving together with their does. Vaginal implant transmitters enabled us to locate specific birth-sites, in addition to the location of neonates (31 birth-sites were identified during springs 2001 and 2002). Aerial photographs were taken to allow habitat characterization of these birth-sites. Additionally, 8 doe-fawn pairs were intensively located from parturition to 2 weeks after birth. Analyses of these data are ongoing.

LITERATURE CITED

Bahnak, B. R., J. C. Holland, L. J. Verme, and J. J. Ozoga. 1979. Season and nutritional effects on serum nitrogen constituents in white-tailed deer. Journal of Wildlife Management 43: 454–460.

Cook, R.C., J. G. Cook, D. L. Murray, P. Zager, B. K. Johnson, and M. W. Gratson. 2001. Development of predictive models of nutritional condition for Rocky Mountain elk. Journal of Wildlife Management 65: 973–987.

DeCalesta, D. S., J. G. Nagy, and J. A. Bailey. 1977. Experiments on starvation and recovery of mule deer does. Journal of Wildlife Management 41: 81–86.

DelGiudice, G. D., L. D. Mech, and U. S. Seal. 1990*b*. Effects of winter undernutrition on body composition and physiological profiles of white-tailed deer. Journal of Wildlife Management 54: 539–550.

DelGiudice G. D., L. D. Mech, U. S. Seal, AND P. D. KARNS. 1987. Effects of winter fasting and refeeding on white-tailed deer blood profiles. Journal of Wildlife Management 51: 865–873.

DelGiudice, G. D., M. E. Nelson, and L. D. Mech. 1991. Winter nutrition and population ecology of white-tailed deer in the central Superior National Forest. United States Department of Agriculture, Forest Service, North Central Experiment Station, St. Paul, Minnesota, USA.

DelGiudice, G. D., M. R. Riggs, P. Joly, and W. Pan. 2002. Winter severity, survival and cause-specific mortality of female white-tailed deer in north central Minnesota. Journal of Wildlife Management 66: 698–717.

Holter, J. B., and H. H. HAYES. 1977. Growth in white-tailed deer fawns fed varying energy and constant protein. Journal of Wildlife Management 41: 506–510.

Mech, L. D., L. D. Frenzel, Jr., and P. D. Karns. 1971. The effect of snow conditions on the vulnerability of white-tailed deer to wolf predation. Pages 51-59 *in* L. D. Mech and L. D. Frenzel, Jr., editors. Ecological studies of the timber wolf in northeastern Minnesota. U. S. Department of Agriculture, Forest Service, North Central Forest Experiment Station, St. Paul, Minnesota, USA.

Nelson, M. E., and L. D. Mech. 1986. Mortality of white-tailed deer in northeastern Minnesota. Journal of Wildlife Management 50: 691–698.

Seal, U. S., L. J. Verme, and J. J. Ozoga. 1978. Dietary protein and energy effects on deer fawn metabolic patterns. Journal of Wildlife Management 42: 776–790.

Stephenson, T. R., K. J. Hundertmark, C. C. Schwartz, and V. V. Ballenberghe. 1998. Predicting body fat and body mass is moose with ultrasonography. Canadian Journal of Zoology 76: 717–722.

Verme, L. J. 1969. Reproductive patterns of white-tailed deer related to nutritional plane. Journal of Wildlife Management 33: 881–887.

Verme, L. J., and J. J Ozoga. 1980. Effect of diet on growth and lipogenesis in deer fawns. Journal of Wildlife Management 44: 315–324.

Verme, L. J., and J. J Ozoga. 1981. Appraisal of autumn-spring weather severity for northern deer. Wildlife Society Bulletin 9: 292–295.

Warren, R. J., R. L. Kirkpatrick, A. Oelschlaeger, P. F. Scanlon, K. E. Webb, Jr., and J. B. Whelan. 1982. Energy, protein, and seasonal influences on white-tailed deer fawn nutritionalindices. Journal of Wildlife Management 46: 302–312.

Watkins, B. E., R. F. Nachreiner, S. M. Schmitt, T. M. Coolley, and D. E. Ullrey. 1982. Thyroid function in fed and fasted white-tailed deer fawns. Canadian Journal of Zoology: 60: 1331–1338.

Table 1. Cause specific mortality of free ranging, radiocollared, adult (≥1.0 year old) female white-tailed deer, north central Minnesota, 16 June 2000-15 June 2001 and 16 June 2001-15 June 2002.

	2000–01 Crude mortality rate		2001–02 Crude mortality rate	
Fate	*n*	%	*n*	%
Wolf-kill	5	7.0	8	9.9
Bobcat-kill	2	2.8	1	1.2
Unknown predator	2	2.8	0	0.0
Hunter-kill[a]	6	8.5	5	6.2
Car-kill	1	1.4	2	2.5
Total deaths	16	22.5	16	19.8
Censored animals	8		13	
Alive	47		52	
Total 71	81			

Table 2. Birth characteristics of free-ranging white-tailed deer neonates, captured during springs 2001 (n = 31) and 2002 (n = 35), north central Minnesota. Ranges are presented in parentheses.

Year	Capture date Mean	SE	Estimated birth-date[a] Mean	SE	Age at capture[a] (days) Mean	SE	Capture mass (kg) Mean	SE	Estimated birth-mass[b] (kg) Mean	SE	New hoof growth (mm) Mean	SE	Hoof length (cm) Mean	SE
2001	29 May	1.5	26 May	1.7	2.5	1.1	3.2	0.2	2.8	0.1	2.6[c]	0.4	22.6	0.6
	(16 May–19 June)		(5 May–19 June)		(0–32)		(1.5–6.5)		(1.5–4.8)		(0.0–10.8)		(16–28)	
2002	30 May	1.1	26 May	1.3	3.7	0.7	3.7	0.2	3.0	0.1	3.7[c]	0.2	23.3	0.6
	(20 May–15 June)		(11 May–15 June)		(0–15)		(1.4–8.1)		(1.4–5.5)		(2.0–7.5)		(14–31)	

[a]Birth-dates and age at capture were determined from timing of vaginal implant transmitter expulsion, or for neonates of non-implanted dams, estimated using new hoof growth (Sams et al. 1996*b*).

[b]Birth-masses were estimated by assuming an average daily mass gain of 0.2 kg since birth (Verme and Ullrey 1984, Rawson et al. 1992).

[c]Means were significantly different at $P < 0.05$.

Table 3. Survival and cause-specific mortality rates of white-tailed deer neonates at 0–1 (0–7 days), 1–4 (8–28 days) and 4–12 (29–84 days) weeks of age, captured in springs 2001 (n = 31) and 2002 (n = 35), north central Minnesota.[a] Cumulative totals of cause-specific neonate mortalities by 12 weeks of age are presented in parentheses.

	0–1 week of age	1–4 weeks of age	4–12 weeks of age
2001			
Survival	.90	.90	.83
Number of deaths	3	3	5 (11)
Cause-specific mortality			
Wolf	.00	.00	.03 (1)
Bear	.00	.07	.07 (4)
Bobcat	.00	.03	.00 (1)
Red fox	.03	.00	.00 (1)
Unknown predator[b]	.00	.00	.07 (2)
Unknown cause[c]	.07	.00	.00 (2)
2002			
Survival rate	.82	.88	.78
Number of deaths	6	4	7 (17)
Cause-specific mortality			
Wolf	.00	.00	.00 (0)
Bear	.09	.00	.06 (5)
Bobcat	.00	.12	.13 (8)
Red fox	.00	.00	.00 (0)
Unknown predator[c]	.06	.00	.00 (2)
Unknown cause[d]	.03	.00	. 03 (2)

[a]One neonate slipped its radiocollar at 41 days of age and was censored from survival analyses at 4–12 weeks of age in 2001. Three neonates slipped their radiocollars at 22, 45 and 74 days of age and were censored from survival analyses at 1–4 (n = 1) and 4–12 (n = 2) weeks of age intervals in 2002.
[b]"Unknown predator" was assigned as cause of mortality when site evidence indicated a predator-kill, but the species of predator could not be determined conclusively.
[c]When evidence indicated death, but was not conclusive as to cause, it was recorded as "unknown cause."

MOOSE POPULATION DYNAMICS IN NORTHEASTERN MINNESOTA

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

BACKGROUND

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. Records from the Superior National Forest (Peek et al. 1976) suggest that moose numbers increased dramatically in the late 1920's, but plummeted in the mid 1930's, and remained low until the mid to late 1960's. Population estimates from aerial surveys in northeastern Minnesota, conducted since 1959, suggest that the population gradually began to increase in the 1970's and 1980's to a peak of 6,900 in 1988 and then dropped to 3,700 by 1990. In recent years, moose numbers have apparently stabilized around 4000 animals.

We can only speculate as to the causes of past fluctuations in the northeastern moose population. Undoubtedly, moose numbers were reduced in the early decades of this century by the cumulative effects of settlement: over-hunting and timber harvest followed by wide-spread wildfire. The increase in moose numbers in the late 1920's probably reflected the closure of the moose season in 1921 combined with the ideal habitat provided by the early stages of the second growth forest. It is less clear why the population declined so dramatically in the mid 1930's. Increased poaching associated with the Great Depression, maturation of the forest habitat, and increased exposure to "brainworm" (*Parelaphostrongylus tenuis*) from higher deer numbers, probably all contributed to the reduction in moose numbers. In the early 1970's, the gradual increase in moose numbers corresponded with record low deer numbers throughout the northeast possibly as a result of a reduced the incidence of *P. tenuis* related mortality. Predation was probably reduced as well, because wolf numbers declined in portions of the northeast in response to the reduced deer numbers (Mech 1986). It is also possible that hunter selectivity for bulls (beginning in 1971) may have increased the population growth rate by increasing the proportion of females in the herd. Between 1988 and 1990, moose numbers decreased over 50%. Circumstantial evidence suggested that much of the mortality was associated with massive infestations of an external parasite, the "winter tick" (*Dermacentor albipictus*). Research suggests that outbreaks of this parasite may be related to weather (Drew and Samuel 1985, Samuel and Welch 1991) and if so, are independent of moose density.

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

[1] United States Geological Survey, U.S. Geological Survey, Northern Prairie Wildlife Research Center, Jamestown, North Dakota, US
[2] Fond du Lac Resource Management Division, 1720 Big Lake Road, Cloquet, Minnesota 55720
[3] 1854 Authority, 4428 Haines Road, Duluth, Minnesota 55811

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 in recent years are biased low.

OBJECTIVES:

1) Determine annual rates of non-hunting mortality for northeastern moose.
Simulation modeling suggests that Minnesota's northeastern moose population should be increasing. The results from annual aerial surveys, however, indicate that numbers have remained relatively constant, despite conservative harvest levels. The proposed study will establish whether high levels of non-hunting mortality are preventing this population from increasing, identify causes of non-hunting mortality and determine whether is if feasible to develop an index that can be used to predict annual variation in this mortality.

2) Determine annual rates of reproduction in northeastern moose.
Research in northwestern Minnesota indicated that a low proportion of cow moose were pregnant and that this contributed to a decline in moose numbers. The proposed study will document annual pregnancy, twinning, and calf mortality rates to determine whether reduced reproduction is preventing the population from increasing and attempt to identify indices that predict annual variation in reproduction.

3) Calibrate aerial moose survey methodology
Aerial surveys assume that observers do not tabulate some proportion of moose. This proportion varies among observers, habitat types, snow conditions, and timing of the survey. The proposed study will document the magnitude of this proportion and identify ways to improve the survey methodology.

Methods

During 9-11 February 2004, 18 adult moose (6 bulls and 12 cows) were immobilized with a combination of carfentanil and xylazine delivered by a dart gun from a helicopter to replace moose that died during the previous year. A total of 84 moose (50 cows and 34 bulls) have been captured in northeastern Minnesota since February 2002 (Fig. 1). A radio-collar was attached and blood, hair, and fecal samples were collected from each moose. Beginning in 2003, a canine tooth was also extracted for aging. Details of condition assessment used on moose in 2003 and 2004 are provided elsewhere in this Research Summary (DelGiudice et al. 2004).

Mortality was determined by monitoring a sample of up to 60 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 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 half of the moose each week and a mortality check was conducted on the remaining moose.

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 2004, 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 (1.7 to 9.0 mi^2)was created around one or more collared moose and 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 to Date

As of 1 May 2004, 25 radio-collared moose (12 bulls and 13 cows) have died. The cause of death in 10 cases could be identified (3 hunter kill, 1 train, 2 trucks, 2 wolf predation, and 1 natural accident). We were unable to examine remains of 2 moose that died within BWCAW. Fourteen appear to have died from unknown non-traumatic causes. In eight 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 - Jan., 3 - April, 4 May, 1 June, 1 July, 2 August, 1 Nov., 2 Dec.). 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 mortality for bulls was 0% and 27% in 2002 and 2003,respectively. Only 7 bulls were collared during 2002, however. Annual non-hunting mortality for cows was 29% and 23% in 2002 and 2003, respectively. In both sexes, non-hunting mortality was substantially higher than documented for populations outside of Minnesota (generally 8 to 12%).

Serum samples from 61of the radio-collared moose were tested for the presence *P. tenuis*-specific antibodies using an enzyme-linked immunosorbent assay procedure (ELISA) (Ogunremi et al. 1999). Seven of the 61 moose (7 cows and 2 bulls) were sero-positive for antibodies against P. tenuis; two subsequently died from unknown non-traumatic causes and 1 was scavenged by wolves.

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

Radio collared moose were located 43 times in the process of developing a sightability model. In 23 cases, the collared moose was observed using the standard survey protocol; in 20 cases, the collared moose was not observed and telemetry had to be used to locate the collared moose. Because of the small sample, only 2 covariates (visual obstruction and temperature (F^{o})) were included in the analyses. The inclusion of both covariates resulted in a model that correctly classified 76% of the observations. Total population size based on this sightability model was substantially higher than previous estimates calculated using the "Gasaway" protocol (Gasaway et al. 1986, Lenarz, unpubl.).

LITERATURE CITED

Anderson, C.R. and F.G. Lindzey. 1996. Moose sightability model developed from helicopter surveys. Wildl. Soc. Bull. 24:247-259.

Cox, E.W., W.B. Ballard, D.L. Murray, H.A. Whitlaw, M.S. Lenarz, T.W. Custer, T.K. Fuller, T. Barnett. Pathogen and weather mediated decline in a northwest Minnesota moose population. Wildl. Monog. Submitted.

DelGiudice, G.D., M.S. Lenarz, M. Schrage, A.J. Edwards, and M. Nelson. 2004. Condition of moose (*Alces alces*) in northeastern Minnesota. Pages 89-95 *in*: M.W. DonCarlos, R.O. Kimmel, J.S. Lawrence, and M.S. Lenarz, *editors*, Summaries of wildlife research findings, 2003. Minnesota Dept. of Natural Resources, Wildlife Populations and Research Unit, St. Paul, Minnesota. In Press.

Drew, M.L. and W.M. Samuel. 1986. Reproduction of the winter tick, *Dermacentor albipictus*, under field conditions in Alberta, Canada. Can. J. Zool. 64:714-721.

Gasaway, W.C., S.D. DuBois, D.J. Reed, and S.J. Harbo. 1986. Estimating moose population parameters from aerial surveys. Biol. Pap. Univ. Alaska, No. 22. 108pp.

Haigh, J.C., E.W. Kowal, W. Runge, and G. Wobeser. 1982. Pregnancy diagnosis as a management tool for moose. Alces 18: 45-53.

Mech, L.D. 1986. Wolf population in the central Superior National Forest, 1967-1985. U.S. For. Ser. Res. Pap. NC-270. 6pp.

Ogunremi, O., M. Lankester, J. Kendall, and A. Gajadhar. 1999. Serological diagnosis of *Parelaphostrongylus tenuis* infection in white-tailed deer and identification of a potentially unique parasite antigen. J. Parasitol. 85(1): 122-127.

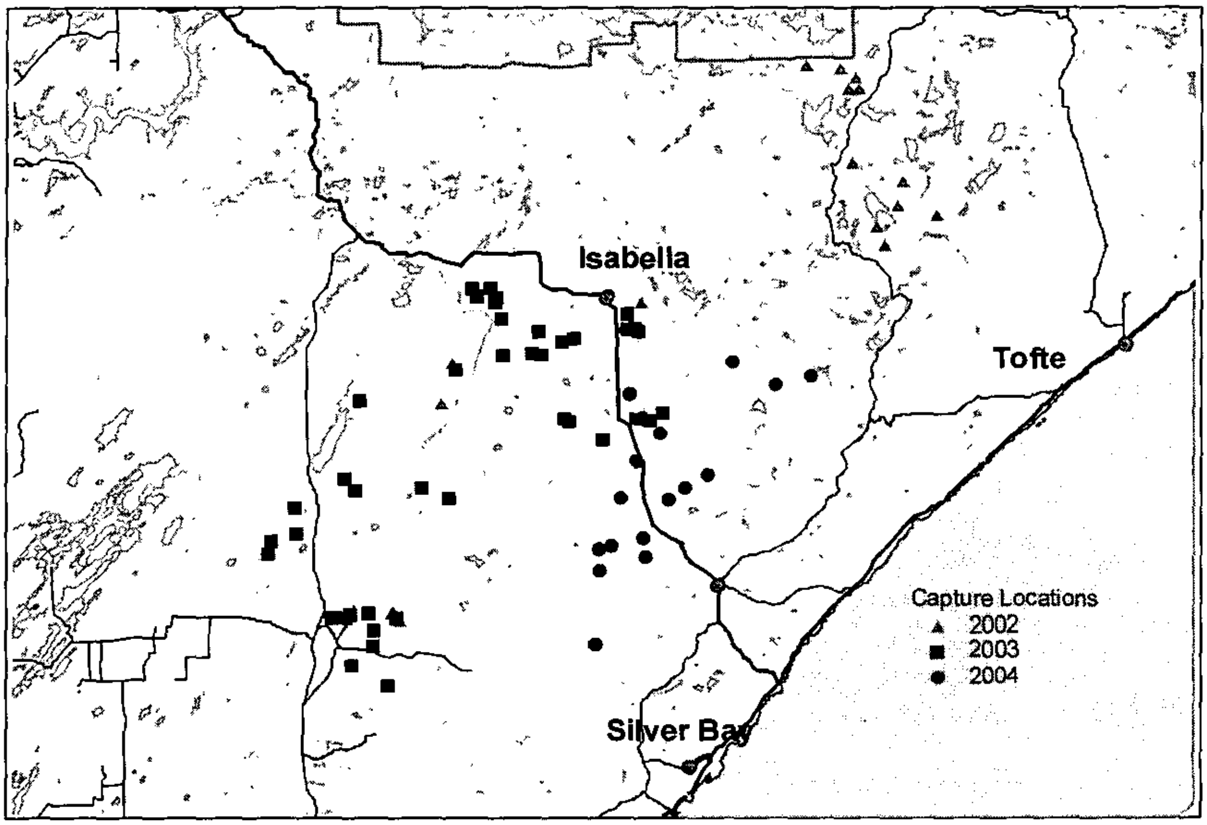
Peek, J.M., D.L. Urich and R.J. Mackie. 1976. Moose habitat selection and relationships to forest management in northeastern Minnesota. Wildl. Monog. 48. 65pp.

Pollock, K.H., C.T. Moore, W.R. Davidson, F.E. Kellogg, and G.L. Doster. 1989. Survival rates of bobwhite quail based on band recovery analyses. J. Wildl. Manage. 53:1-6.

Quayle, J.F., A.G. MacHutchon, and D.N. Jury. 2001. Modeling moose sightability in south central British Columbia. Alces. 37:43-54.

Samuel, W.M. and D.A. Welch. 1991. Winter ticks on moose and other ungulates: factors influencing the population size. Alces 27:169-182.

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



EFFECTIVENESS OF CHANGES TO THE AERIAL MOOSE SURVEY PROTOCOL IN NORTHEASTERN MINNESOTA*

Mark S. Lenarz

Abstract

The Minnesota Dept. of Natural Resources has conducted aerial surveys using a stratified random plot protocol (Gasaway et al. 1986) to estimate moose (*Alces alces*) numbers each year since 1982. Lenarz (1998) analyzed precision and bias in these surveys and made 5 recommendations to improve both the survey estimates and their precision. Since 1998, all aerial moose surveys began in early January so as to have a standard starting date. Two flight crews were used in the surveys to complete surveys earlier. The entire survey area was re-stratified and minor revisions are made on an annual basis. Optimal allocation has been used each year to establish the number of plots in each stratum. The number of re-survey plots has been increased so as to minimize the precision associated with the sightability correction factor and thereby increase the precision of the population estimate. The effectiveness of these changes to the survey protocol will be discussed.

* Abstract of paper presented at 39th North American Moose Conference and Workshop. 18-21 May 2003; Jackson, Wyoming.

NORTHEASTERN MINNESOTA MOOSE MANAGEMENT—A CASE STUDY IN COOPERATION*

Andrew J. Edwards[1], Michael W. Schrage[2], and Mark Lenarz

ABSTRACT

This paper provides an overview of moose management in northeastern Minnesota with an emphasis on relationships between the State and Tribal entities that share management responsibility. Specific topics discussed include settlement of treaty rights issues, harvest allocation and strategies, and the evolving State-Tribal partnerships that have been created during the past 15 years. Brief updates on the status of moose in Minnesota, population monitoring efforts, population goals, and the future direction of management are provided.

---

[1]1854 Authority, 4428 Haines Road, Duluth, Minnesota 55811
[2]Fond du Lac Resource Management Division, 1720 Big Lake Road, Cloquet, Minnesota 55720
* Abstract of paper presented at 39th North American Moose Conference and Workshop. 18-21 May 2003; Jackson, Wyoming.

SEROLOGICAL DIAGNOSIS OF *PARELAPHOSTRONGYLUS TENUIS* INFECTION IN MOOSE*

O. Ogunremi[1], M. Lankester[2], E. Cox[3], D. Murray[4], W. Ballard[5], H. Whitlaw[5], M. Lenarz[6], M. Schrage[7], M. Nelson[8], and A. Edwards[9]

ABSTRACT

Meningeal worm or *Parelaphostrongylus tenuis* (Family: Protostrongylidae) is a known cause of neurological signs and death in moose, and infection can be diagnosed antemortem by a newly developed enzyme-linked immunosorbent assay (ELISA) which detects serum IgG antibodies directed against the excretory-secretory products of the third-stage larvae of *P. tenuis*. For the purpose of obtaining base-line serological information on moose in a *P. tenuis* endemic area, serum samples were obtained from a total of 162 moose at the time of collaring. These were made available from 2 different studies: 65 moose from the northeast part of Minnesota (47E 30' and 91E 25') collared between 2002 and 2003, and 97 moose in from the northwest part of the state (48E 25' and 96E 15') collared between 1995 and 1998. Health status of the animals in the northwest part was monitored and carcasses of dead animals examined; observations and presumed cause of mortality were recorded. In the northeast, the proportion of animals with antibodies to *P. tenuis* (% seropositive, ELISA Optical Density > 0.386) was 20.0% (n=65). Clinical signs were recorded for 4 animals which were eventually necropsied. Two of the animals were euthanized because of marked disease processes including inability to stand, complete lack of fear of humans or emaciation, and on postmortem examination adult *P. tenuis* worms were recovered from the heads, which correlated with positive serology. Another 2 animals displayed clinical signs consistent with *P. tenuis* infection including emaciation or unsteady gait progressing to inability to stand, and were both were seropositive even though no worms were recovered at postmortem.

---

[1] Centre for Animal Parasitology, Canadian Food Inspection Agency Saskatoon Laboratory, Saskatoon, Canada.
[2] Department of Biology, Lakehead University, Thunder Bay, Ontario, Canada
[3] (Deceased) University of Idaho, RR 1, Box 74, Middle River, Minnesota, US
[4] Department of Biology, Trent University, Peterborough, Ontario, Canada.
[5] Department of Range, Wildlife, and Fisheries Management, Texas Tech University, Lubbock, Texas, US
[6] Minnesota Department of Natural Resources, Forest Wildlife Populations and Research Group, Grand Rapids, Minnesota
[7] Fond du Lac Resource Management Division, Cloquet, Minnesota
[8] United States Geological Survey, U.S. Geological Survey, Northern Prairie Wildlife Research Center, Jamestown, North Dakota, US
[9] 1854 Authority, Duluth, Minnesota, US
* 52nd Annual Conference of the Wildlife Disease Association, Saskatoon, Alberta, Canada. 11-14 August 2003.

In the northwest group, 54 of the collared animals had died by 2000 when monitoring was discontinued, 34 animals were still alive, and 9 were missing. Recorded causes of death (3 categories) and initial anti-*P. tenuis* antibody titers were: (i) loss of body condition and likely disease process including *P. tenuis*, 46.7% seropositive (n=15); (ii) non-infectious, 14.8% seropositive (n=27);, and (iii) possibly flukes-related (possibly*Fascioloides magna),* 25% seropositive, (n=12). Seven (20.6%) of the 34 animals still alive in 2000 were seropositive when bled 3-5 years earlier. Animals dying of a disease process where *P. tenuis* may be involved were more likely to be seropositive than those dying of non-infectious agents or where flukes may have played a role (P=0.04, Fisher=s Exact test; two tailed). Animals dying of disease process were more likely to be seropositive (46.7%) than surviving animals (20.6%) but this difference was not significant at 90% confidence level (P=0.09; Fisher=s Exact test; two tailed). From this preliminary study, the following conclusions can be made: (a) *P. tenuis* appears to play a significant role in moose mortalities in northern Minnesota, (b) some survive for a number years despite *P. tenuis* seropositive status, although some of the seropositive animals may be expected to succumb to *P. tenuis* infection over with time, (c) the serological test gave a higher estimate of exposure of moose to *P. tenuis* than past studies using traditional tests which used parasite recovery methods, (d) the serological test appears to be a worthwhile tool for the management of moose herds, (e) the test can now be used to directly investigate the hypothesis that moose survivorship in *P. tenuis* endemic areas is significantly associated with negative *P. tenuis* serology although a large sample size may be required because of the confounding effect of other causes of moose mortality.

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

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

BACKGROUND

A study of moose (*Alces alces*) in northeastern Minnesota was begun in 2002, because aerial survey estimates suggested the population was stable despite a very conservative harvest (Lenarz et al. 2002). 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. 2002). One of the primary objectives is to "determine annual rates of non-hunting mortality…" for moose in this part of the state (Lenarz et al. 2002). Because 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 1997), assessment of winter condition of moose recruited into the present study was deemed a worthwhile field objective. 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 and 2003-04 were feasible and successful. Herein, we report the results of those condition assessments for live-captured moose.

METHODS

During 26 February-2 March 2003 and 9-11 February 2004, adult ($\geq$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. 2003, 2004).

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.) 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).

[1] Fond du Lac Resource Management Division, 1720 Big Lake Road, Cloquet, Minnesota 55720
[2] 1854 Authority, 4428 Haines Road, Duluth, Minnesota 55811
[3] United States Geological Survey, U.S. Geological Survey, Northern Prairie Wildlife Research Center, Jamestown, North Dakota, US

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.

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 and 17 (12 females, 5 males) of 18 moose in winter 2003-04. Overall, mean Maxfat was 1.6 (SE = 0.16, range = 0-3.8 cm) and 2.1 cm (SE = 0.38, range = 0.58-4.6 cm) during these 2 winters. In captive moose, Maxfat measurements have ranged between 0 and 7.0 cm, and were directly related ($Y = 5.61 + 2.05x$, $r^2 = 0.96$, $P < 0.0001$) 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%) and 9.9% (range = 6.8-15.0%) during winters 2002-03 and 2003-04, 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 both winters; the differences were significant ($P \leq 0.05$) in winter 2003-04 (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 and 7.3 (range = 4-9) and 3.8 (range = 2.5-5.0) in winter 2003-04. According to both of these scoring systems, although not significantly, mean condition scores were lower for bulls than cows during both winters (Table 3). There were significant correlations between the FCC and BCS_r scores for all moose during winters 2002-03 ($r = 0.83$, $P < 0.0001$) and 2003-04 ($r = 0.75$, $P = 0.002$). Additionally, during both winters, Maxfat was significantly correlated to FCC scores ($r = 0.56$ and 0.71, $P \leq 0.01$) and BSS_r scores ($r = 0.53$ and 0.68, $P \leq 0.02$). The strength of the relationship between the scoring systems and Maxfat measurements is 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) 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 and moderate weather conditions that characterized winters 2002-03 and 2003-04, respectively, 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 captures, snow depths were typically approaching 90 cm. Conducting moose captures several weeks earlier in 2004 than in 2003 also likely contributed to the similar conditions assessed during the two winters. Our assignment of qualitative assessments of "very good," "good," and "fair-poor" to FCC scores as presented in Table 4, indicating that about 75-76% of the moose were in good to very good condition during the 2 winters, were consistent with our evaluations derived from fat measurements. The most notable case of a moose in poor condition during winter 2002-03, was a female with no rump fat (Maxfat = 0), the lowest FCC and BCS_r scores (3 and ≤ 2, respectively) of all 37 moose scored that year, and which died within hours of release, despite a typical, relatively rapid apparent recovery from the chemical immobilization.

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. Further, as this study progresses, annual condition assessments of new recruits of the study cohort may contribute to our understanding of the impacts of varying environmental conditions (e.g., winter severity/habitat quality, winter tick infestation) on performance (i.e., survival rates, reproductive success) of the local population over time.

LITERATURE CITED

Cook, R. C., J. G. Cook, D. L. Murray, P. Zager, B. K. Johnson, and M. W. Gratson. 2001. Development of predictive models of nutritional condition for RockyMountain elk. Journal of Wildlife Management 65:973-987.

DelGiudice, G. D., R. O. Peterson, and W. M. Samuel. 1997. Trends of winter nutritional restriction, ticks, and numbers of moose of Isle Royale. Journal of Wildlife Management 61:895-903.

Franzmann, A. W. 1977. Condition assessment of Alaskan moose. Proceedings of the North American Moose Conference and Workshop. 13:119-127.

Lenarz, M. S., M. Schrage, A. Edwards, and M. E. Nelson. 2002. Moose population dynamics in northeastern Minnesota. Pages 168-172 *in* M. W. DonCarlos, M. A. Hanson, R. O. Kimmel, and M. S. Lenarz, *editors*, Summaries of wildlife research findings, 2001. Minnesota Department of Natural Resources, Wildlife Populations and Research Unit, St. Paul, Minnesota.

Lenarz, M. S., M. Schrage, A. Edwards, and M. E. Nelson. 2003. Moose populationdynamics in northeastern Minnesota. Pages 172-176 *in* M. W. DonCarlos, M. A. Hanson, R. O. Kimmel, and M. S. Lenarz, *editors*, Summaries of wildlife research findings, 2002. Minnesota Department of Natural Resources, Wildlife Populations and Research Unit, St. Paul, Minnesota.

Lenarz, M. S., M. Schrage, A. Edwards, and M. E. Nelson. 2004. Moose populationdynamics in northeastern Minnesota. Pages 79-84 *in* M. W. DonCarlos, M. A. Hanson, R. O. Kimmel, and M. S. Lenarz, *editors*, Summaries of wildlife research findings, 2002. Minnesota Department of Natural Resources, Wildlife Populations and Research Unit, St. Paul, Minnesota. In Press.

Stephenson, T. R., V. C. Bleich, B. M. Pierce, and G. P. Mulcahy. 2002. Validation of mule deer body composition using *in vivio* and post-mortem indices of nutritional condition. Wildlife Society Bulletin 30:557-564.

Stephenson, T. R., K. J. Hundertmark, C. C. Schwartz, and V. Van Ballenberghe. 1998.Predicting body fat and body mass in moose with ultrasonography. Canadian Journal of Zoology 76:717-722.

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) and 2003-04 (11 females, 5 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 all 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) and 2003-04 (10 females, 4 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) and 2003-04 (11 females, 5 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] (0-3.8)	0.24	7.4 (3.0-10.0)	0.4	3.6 (2.0-4.5)	0.2
Males	1.5[b] (0.3-2.6)	0.20	7.0 (4.0-9.0)	0.3	3.2 (2.0-4.3)	0.1
Winter 2003-04						
Females	2.9[c] (1.5-4.6)	0.42	7.8 (5.0-9.0)	0.3	4.1[d] (3.0-5.0)	0.2
Males	1.1[c] (0.6-2.6)	0.38	6.2 (4.0-8.0)	0.8	3.1[d] (2.5-4.0)	0.3

[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.

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

	Franzmann's Condition Score			
	≥ 8 (Very Good)	$7 \leq x < 8$ (Good)	≤ 6 (Fair-Poor)	Total
Winter 2002-03				
Number of moose	15	13	9	37
Percent of total	41	35	24	100
Winter 2003-04				
Number of moose	9	3	4	16
Percent of total	56	19	25	100

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

RIVER OTTERS IN SOUTHEASTERN MINNESOTA: ACTIVITY PATTERNS AND AN INDEX OF POPULATIONS BASED ON AERIAL SNOW-TRACK SURVEYS.

Daniel J. Martin[1], John D. Erb, Brock R. McMillan[1], John R. Fieberg, and Thomas A. Gorman[1]

Background and Justification

The river otter (*Lontra canadensis*) is a semi-aquatic member of the family Mustelidae and is indigenous to the state of Minnesota. Over the past century, populations of river otter declined nationwide due to environmental degradation (including drainage and pollution of wetlands) and unregulated trapping (Nowak 1999). Recently, however, river otter numbers have increased throughout much of their original range in North America (Miller 1992, Serfass et al. 1993, Chilelli et al. 1996). This rebound can be attributed to successful reintroduction programs, increased legal protection of river otters and their habitat, and more effective management (Melquist and Dronkert 1987, Chilelli et al. 1996, Raesly 2001).

Due to previous low abundance, otter harvest has been prohibited in the southern half on MN. However, based on anecdotal evidence (e.g., non-target capture by trappers), it appears that abundance of river otters has been increasing in southeastern Minnesota for some time. Though otter numbers appear to have increased in southeastern Minnesota, otter habitat in southern Minnesota is less extensive and more isolated compared to northern Minnesota. This, combined with moderate fecundity, suggests river otter distribution and abundance in southern Minnesota could be more affected by negative environmental impacts and over-harvest (Tabor and Wight 1977, Melquist and Dronkert 1987, Nowak 1999, Erb and DePerno 2001).

The ability to detect trends in abundance of river otters will enable improved management and conservation of this species in Minnesota. In addition, the Convention on the International Trade of Endangered Flora and Fauna (CITES) agreement requires countries that export otter pelts to monitor their populations (CITES 1973). Methods for monitoring trends in abundance of river otter have historically focused on harvest data or latrine surveys (Reid et al. 1987, Chilelli et al. 1996). However, indices based on harvest data may be inaccurate due to compounding economic effects (e.g., increase in price paid for pelts) and trapper behavior, while latrine surveys can be time-intensive and subject to low and variable detection rates (Kruuk and Conroy 1987, Romanowski et al. 1996).

Various forms of aerial snow-track (AST) surveys have successfully been used to monitor populations of other furbearer species, including the gray wolf (*Canis lupus*), coyote (*Canis latrans*), marten (*Martes americana*), red fox (*Vulpes vulpes*), Canada lynx (*Lynx canadensis*), and others (Golden 1994, Becker et al. 1998, Squires 2002, Anthony et al. 2002). AST surveys allow the coverage of large areas in a short amount of time relative to alternative methods. Because river otters primarily inhabit aquatic ecosystems, restricting survey routes to these systems should optimize efficacy of the survey as compared with similar surveys for other

[1]Dept. of Biol. Sci., MSU; 242 Trafton Science Center S.; Mankato, MN 56001.

furbearer species. In addition, river otters traveling on snow often leave tracks that are easily distinguishable from sign of other species. These factors, combined with a climate that consistently produces adequate snowfall, further warrant testing of an AST survey for monitoring trends in population size of river otters in Minnesota.

In addition to helping understand sources of within-year variability in AST surveys, monitoring the movement and activity patterns of river otters may help refine protocol for operational use of the survey. Several standards, including the definition of days since snowfall and the independence of river otter sign, will be evaluated based on the timing, distribution, and intensity of river otter activity. Furthermore, information describing the spatial ecology of river otters should improve the quality of decisions affecting management of both river otters and their habitat.

Objectives

AST Surveys: For snow-track surveys to be comparable through time, it is important that variation in abilities of detecting river otter sign among observers is small. In addition, quantifying other sources of variability (e.g., differences among survey routes, date of survey, days since snowfall) within a winter will improve standardization and maximize the ability to detect multi-annual trends in abundance. Thus, our primary objectives are to quantify within-year variability caused by different observers, date of survey, and days since snow. We will also evaluate how the spatial scale of data recording influences measures of variability, and thus which scale or metric (total track areas, proportion of blocks with sign, etc) is most desirable as a trend indicator. The index will be based initially on data collected during preliminary surveys conducted in 2001-2002 (Erb and DePerno 2001) and subsamples from 2003-2004 surveys. Relations among survey variables will then be compared for similarity to additional survey data collected in the 2004 winter.

Activity Patterns: Descriptions of activity patterns of river otters will enable managers to evaluate seasonal habitat requirements and spatial relations between conspecifics. This information will also be helpful for standardizing spatial parameters of the AST survey design. We will determine diel activity patterns of river otters in all seasons and in relation to ambient environment conditions (e.g., various weather parameters), habitat characteristics, and their sympatric relationships.

Study Area

The study area is located in the Paleozoic Plateau of southeast Minnesota, a subunit of the Eastern Broadleaf Forest ecosystem. This includes the Blufflands area along the western edge of the Southern Mississippi River Basin, which consists of a loess-capped plateau furrowed with river valleys (MN DNR Ecological Services). AST survey routes were flown over the Zumbro and Whitewater Rivers, and two sections of the Mississippi River (Fig. 1). Radio-marked river otters used for determining activity patterns are located throughout the study area. A sub-sample of river otters from the Whitewater River Valley was used to determine movement patterns.

Methods

AST Surveys: Aerial snow-track surveys were conducted from a Bell OH-58A+ helicopter. Although more expensive than a fixed-wing aircraft, a helicopter was used because of increased visibility (e.g. larger windows), speed control (sustained stable flight at low speeds), and maneuverability. Each observer was required to have a minimum of 3 hours of training in detection of river otter sign in snow from an aircraft prior to participation in the study. During 2002, observers were trained by conducting AST surveys over the Mississippi River Valley and the Minnesota River Valley

Global Positioning System (GPS) waypoints were collected directly above permanent visual landmarks in order to delineate AST survey routes (Fig. 1). On routes along the Mississippi River, additional waypoints were recorded as necessary in order to locate redirections in the survey flight path. Because the flight paths were restricted to the main channels, only beginning and end waypoints were recorded on smaller rivers (i.e., the Whitewater and Zumbro Rivers).

Variability among observers was determined using 2 or 3 different observers individually surveying the same routes on the same day. In order to negate bias from prior observations, the helicopter pilot was neither involved in observation nor confirmation of river otter sign in 2003. Several environmental factors, such as cloud cover, were recorded for each survey in order to test their potential impacts on observer variability. Weather permitting, aerial snow-track surveys were flown after snowfalls greater than 2.5 cm in depth, and from 1–4 days after a snowfall. Location of river otter sign was recorded using a Garmin 150 Global Positioning System (GPS), and any sign observed >5 seconds (flight time) from the previous recorded sign was logged. Integrated variation in the standardized speed limits and altitude for each survey route was a necessary compromise between effective viewing distance, safety, and tortuosity of a given river (Table 1).

With respect to conditions necessary for flight and recording of otter sign, AST surveys flown in 2004 followed the 2003 protocol. In addition, both the pilot and recorder were simultaneous observers in 2004. This change was made in order to refine and test future operational protocol. All survey flight speeds and altitudes were standardized to control for possible variability in the surveys caused by dissimilar flight speeds and altitudes set during 2003 flights (Table 1). Also, the relative role of each observer (pilot plus main observer) was documented for each survey route by marking which observer first located each otter activity area.

Movement and Activity Patterns: River otters were captured and tagged with a radio-transmitter as described by Erb and DePerno (2001). Radio-tracking was performed using an ATS Challenger Model R400 radio telemetry receiver with a 3-element Yagi hand-held antenna. River otters were relocated via triangulation using ≥2 azimuths recorded within a ≤20-minute period when possible. Each river otter was relocated between 2 and 6 times during a 6-hour tracking session. Tracking sessions were conducted based on a randomized, stratified-block sampling design (Table 2).

All individual river otters marked in the Whitewater River and McCarthy Lake State Wildlife Management Area were included in the sample. Activity of an individual river otter was determined as either active or inactive based on variation in signal cutout during individual locations attempts. Two estimators of activity were used to determine activity patterns, 1) the distance covered (movement) during the diel period (an indirect measure), and 2) the percentage of fixes (i.e., location attempts) coinciding with activity (a direct measure).

Results

AST Surveys: Sixty AST surveys were conducted among 4 survey routes from 15 January–11 March 2003 (Table 4). Final analysis of this data has not been completed. Preliminary results suggest that variability among observers was low compared to variation between days after snowfall and among sites (see Fig. 2 for example from survey on Whitewater River). Cloud cover appears to have the greatest impact on variability among observers. Twenty-three AST surveys were flown on 6 dates during winter 2004 (Table 5); no results are available at this time.

Activity Patterns: Seven radio-implanted river otters were tracked in the Whitewater River Valley, and two were tracked in the McCarthy State Wildlife Management Area to determine movement patterns (Table 4). River otters were radio-tracked for over 650 hours between 1 June 2002 and 31 October 2003 (Table 3). Most tracking sessions to determine movement patterns involved between 2 and 6 locations of the river otters. Based on a direct measure of activity, river otters in the Whitewater Valley appear to be active primarily at night, but also show relatively high levels of activity during morning and evening hours in winter months (Fig. 3).

Preliminary analysis includes 7 individual river otters that were radio-tracked during October 2002–February 2003 (Table 2). These otters were radio-tracked for a total >281 hours over 53 tracking sessions in winter months. We ran an ANOVA and found no significant difference (p=0.503) among mean activity levels of river otters. Therefore, we grouped all otters prior to running the full-factorial model. The remaining factors included in the model were: sun (i.e., photoperiod), month, and time-block. These factors and their interactions were all significant at α=0.05.

The following mean activity values are the mean percentage of fixes active for each factor, and the associated standard deviation (s.d.). Mean activity was 32% (s.d.=30%), 63% (s.d.=29%), 58% (s.d.=34%), and 79% (s.d.=26%) for diurnal, evening crepuscular, morning crepuscular, and nocturnal tracking blocks, respectively. Mean activity for each month was: October=58% (s.d.=39%), November=58% (s.d.=33%), December=51% (s.d.=27%), January=58% (s.d.=30%), and February=65% (s.d.=39%). Mean activity for sun phase was: Down=71% (s.d.=29%), Up=38% (s.d.=33%), Rising=62% (s.d.=27%), Falling=54% (s.d.=30%).

Discussion and Future Plans

AST Surveys: Further statistical analysis will be conducted to summarize the influence of observer, days since snowfall, date, and weather conditions on survey data and index values. In addition, we will determine how much variability changes when varying the size of the sample 'block' for recording track presence/absence. This information will provide the basis for further development of a sampling design and an index with the greatest potential to detect changes in numbers of river otter. We will continue AST survey flights in winter 2005. Exact protocol has yet to be determined, but may include preliminary evaluation of fitting occupancy models (MacKenzi et al. 2002, 2003) to snow track surveys along transects.

Activity Patterns: River otters were most active during nocturnal and crepuscular hours, and also when the sun was down or during the hour of sunrise or sunset. Mean diel activity of otters was lowest in December as compared to other survey months. Further analysis of environmental measures may provide a more complete explanation of this result.

Location estimates from recorded azimuths will be generated using program Locate II and further spatial analysis will be completed using ESRI ArcView. In order to determine factors that potentially affect diel activity and movement patterns of river otters, we will examine: Julian date, biological season (Table 3), hour of day (Table 2), mean hourly surface temperature, mean hourly water temperature, mean daily precipitation, mean hourly barometric pressure, moon phase, snow depth, river water level, sex of river otter, and age-class of river otter. Movement patterns will be analyzed by comparing maximum, minimum, range, standard deviation, and mean distance moved by individuals, social groups, and demographic groups (i.e., sex and age-class) of river otters, as well as turning angles and tortuosity of movement paths (Turchin 1998). Activity levels estimated with movement patterns (indirect activity estimate) for each river otter will be compared to direct measures in order to determine the sensitivity of these methods for estimating activity patterns.

Acknowledgements

The authors would like to thank J. Lopez, G. Sargeant, M. Sovada, C. DePerno, J. Fieberg, J. Giudice, J. Krenz, and R. Engeman for providing ideas, feedback, and/or logistical support. Special thanks to B. Maas and M. Trenholm, whose expertise in piloting the helicopter allowed for safe completion of the AST surveys. We appreciate the efforts of M. Fischer and S. Gappa for their help in capturing river otters for this study. The Water Resources Center at Minnesota State University, Mankato, allowed use of their GIS lab, and C. Berg provided invaluable guidance with the GIS. We are also grateful to J. Cole, J. McIntire, and other staff at Whitewater State Wildlife Management Area for their continued support. The project was funded by the Minnesota Department of Natural Resources.

Literature Cited

Anthony, M., H. Golden, N. Guldager, R. Skinner, E. Becker, J. Burch, W. Erickson, K. Fox, P. Ladegard, and B. Strauch. Development and testing of aerial videography techniques to monitor furbearer populations. Interagency Collaborative Project Report. 18 October 2002.

Becker, E. F., M. A. Spindler, and T. O. Osborne. 1998. A population estimator based on network sampling of tracks in the snow. The Journal of Wildlife Management 62:968-977.

Chilelli, M. B., Griffith, and D. J. Harrison. 1996. Interstate comparisons of river otter harvest data. Wildlife Society Bulletin 24:238-246.

CITES. 1973. Convention on the International Trade of Endangered Flora and Fauna, Appendix II. IUCN. 3 March 1973.

Erb, J. and C. DePerno. 2001. Research proposal: river otter population and habitat ecology in Southern Minnesota. Minnesota Department of Natural Resources, Farmland Wildlife Populations and Research, Madelia, Minnesota, USA.

_____, B. Berg, M. DonCarlos, and C. DePerno. 2000. River otter population monitoring in southern Minnesota in Summaries of Wildlife Research Findings 1999 (A. Berner et al., eds.). Minnesota Department of Natural Resources, Wildlife Populations and Research Unit – Report 1999:45-54, 2000.

Golden, H. N. 1994. Furbearer Track Count Index Testing and Development. Alaska Department of Fish and Game Division of Wildlife Conservation, Study 7.17 Technical Report.

Kruuk, H., and J. W. H. Conroy. 1987. Surveying otter *Lutra lutra* populations: a discussion of problems with spraints. Biological Conservation 41:179-183.

MacKenzie, D. J., J. D. Nichols, G. B. Lachman, S. Droege, J. A. Royle, and C. A. Langtimm. 2002. Estimating site occupancy when detection probabilities are less than one. Ecology 83:2248-2255.

MacKenzie, D. J., J. D. Nichols, J. E. Hines, M. G. Knutson, and A. B. Franklin. 2003. Estimating site occupancy, colonization, and local extinction when a species is detected imperfectly, Ecology 84: 2200-2207.

Melquist, W. E., and A. E. Dronkert. 1987. River otter. Pages 627-641 *in* Wild furbearer management and conservation in North America.

Miller, M. C. 1992. Reintroduction of river otters into Great Smoky Mountains National Park. Thesis, University of Tennessee. 70 p.

Nowak, R. M. 1999. New world river otters. p 740-741 *in* Walker's Mammals of the World, 6th edition. Johns Hopkins University Press, Baltimore, MD, USA.

Raesly, E. J. 2001. Progress and status of river otter reintroduction projects in the United States. Wildlife Society Bulletin 29:856-862.

Reid, D. G., M. B. Bayer, T. E. Code, and B. McLean. 1987. A possible method for estimating river otter, *Lutra Canadensis*, populations using snow tracks. Canadian Field-Naturalist 101:576-580.

Romanowski, J., M. Brzeziński, and J. P. Cygan. 1996. Notes on the technique of the otter field survey. Acta Theriologica 41:199-204.

Serfass, T. L., R. L. Peper, M. T. Whary, and R. P. Brooks. 1993. River otter (*Lutra canadensis*) reintroduction in Pennsylvania: prerelease care and clinical evaluation. Journal of Zoo and Wildlife Medicine 24:28-40.

Squires, J. R. Snow-tracking protocol used to delineate lynx populations. USDA Forest Service, Rocky Mountain Research Station Technical Report. January 2002.

Tabor, J. E., and H. M. Wight. 1977. Population status of river otter in western Oregon. The Journal of Wildlife Management 41:692-699.

Turchin, P. 1998. Quantitative analysis of movement: measuring and modeling population redistribution in animals and plants. Sinauer Associates, Sunderland, Massachusetts, U.S.A.

Table 1. Altitude and flight-speed limit standards used during 2003 Aerial snow-track surveys. All surveys in 2004 were flown at 56-65 kph and the altitudes designated below.

AST survey route	*Altitude* (m)	*Ground-speed* (kph)
Lower Mississippi River	~76	72-81
Whitewater River	~46	48-57
Upper Mississippi River	~91	72-81
Zumbro River	~46-61	72-81

Table 2. Stratified-block sampling scheme used for tracking of river otters via radio-telemetry in southeastern Minnesota.

Time-block	*Hours*
Morning Crepuscular	04:00-09:59
Diurnal	10:00-15:59
Evening Crepuscular	16:00-21:59
Nocturnal	22:00-03:59

Table 3. Biological seasons of river otters and number of tracking sessions completed to date by time-block (1 June 2002—8 May 2003; *n* = 78).

Time-block	*Birthing / breeding* 1 Mar.—31 May	*Pup-rearing* 1 June—15 Oct.	*Winter maintenance* 16 Oct.—29 Feb.
Morning Crepuscular	2	4	12
Diurnal	4	2	14
Evening Crepuscular	3	5	13
Nocturnal	5	2	12
Total	14	13	51

Table 4. River otters radio-tracked to determine activity and movement patterns, with sex and age-class (at capture) listed for each.

Otter ID	Sex	Age-class[1]
161	Male	Yearling
347	Female	Juvenile
505	Female	Adult
757	Female	Adult
827	Male	Yearling
851	Male	Adult
867	Female	Adult
942	Male	Adult
962	Male	Adult

[1] Juveniles < 1 yr., Yearlings =1-2 yr., Adults >2 yr.

Table 5. Number of observers that completed Aerial snow-track surveys by route and date, 2003 (n = 60) and 2004 (n = 23).

	AST Survey Route				
# Observers	Lower Miss.	Upper Miss.	Whitewater	Zumbro	Days Since Snowfall
15 Jan. 2003	2	0	2	0	2
29 Jan. 2003	3	0	3	0	1
30 Jan. 2003	3	3	3	0	2
7 Feb. 2003	2	1	2	1	2
8 Feb. 2003	3	3	3	3	3
4 Mar. 2003	2	2	1	2	1
5 Mar. 2003	3	3	1	3	2
11 Mar. 2003	2	2	0	2	3
29 Jan. 2004	1	1	1	1	2
3 Feb. 2004	1	1	1	1	1
4 Feb. 2004	1	1	1	1	2
21 Feb. 2004	1	1	1	1	1
6 Mar. 2004	1	1	1	1	1
9 Mar. 2004	1	1	1	0	1
Total	26	20	21	16	

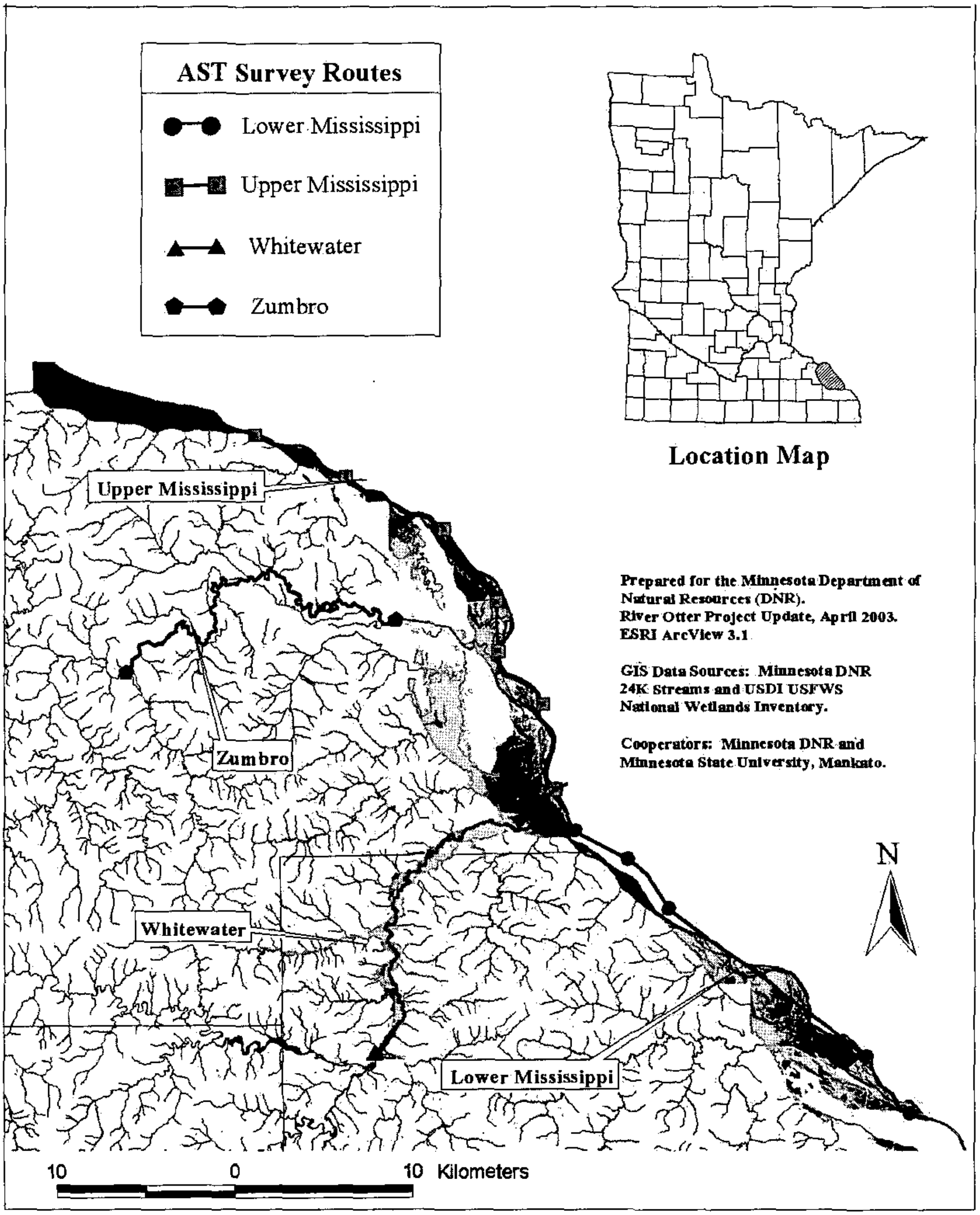


Figure 1. Aerial Snow-track survey routes used to survey for sign of river otters in southeastern Minnesota, 2003–2004.

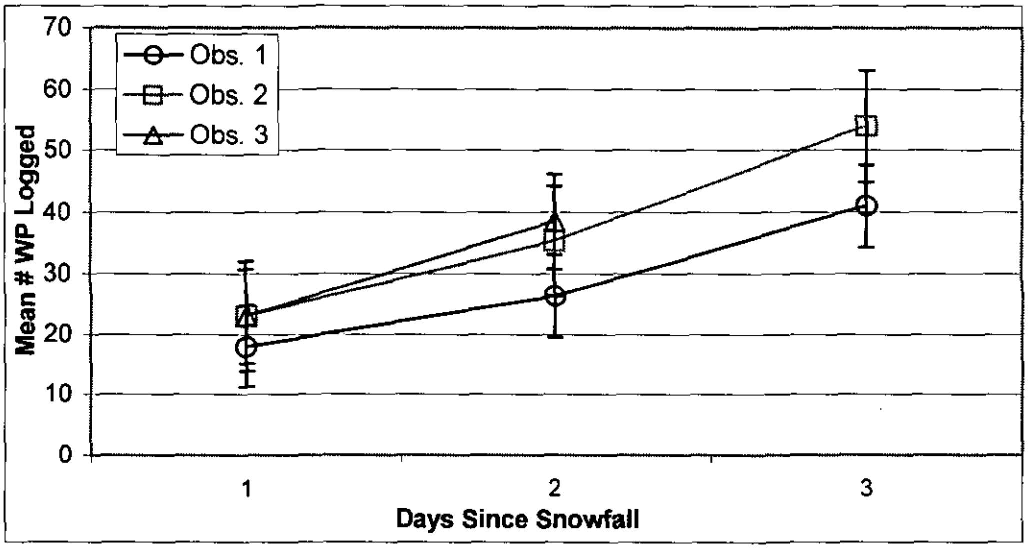
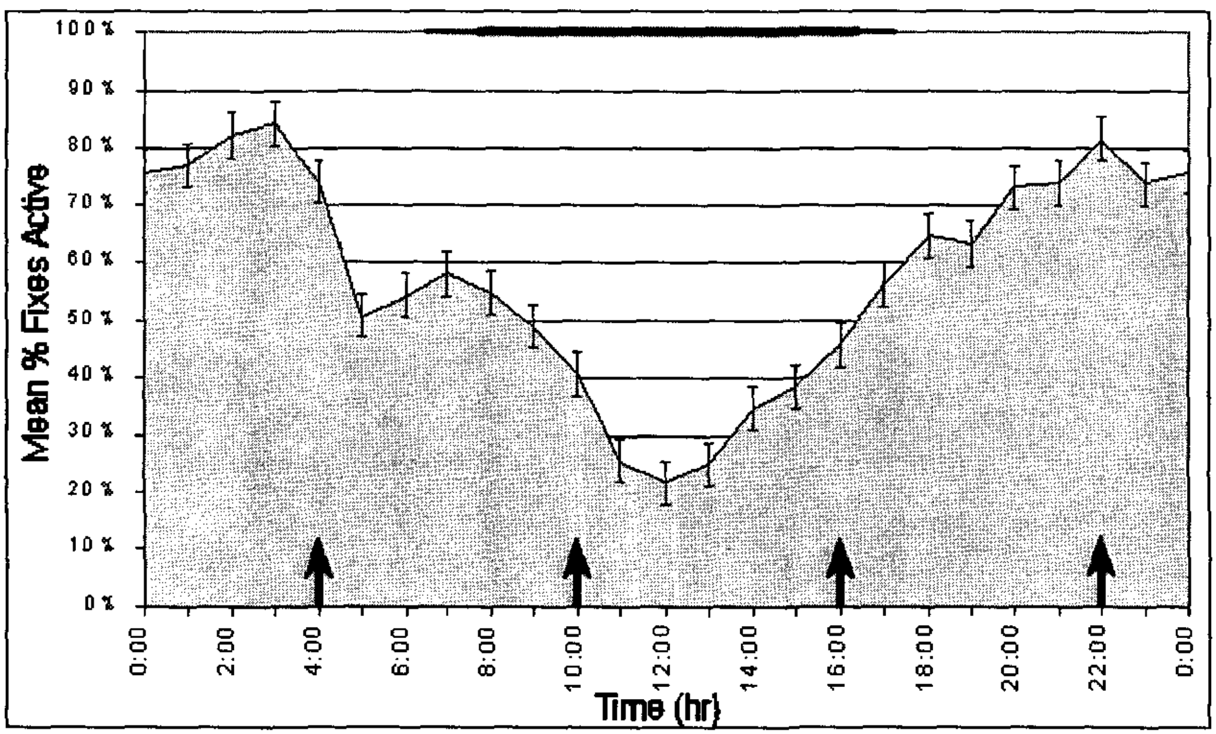


Figure 2. Example of Days Since Snowfall effect on Mean # Waypoints logged, by observer for the Whitewater route. Number of river otter sign recorded by observers varies more among DSS than among Observers. As expected, the number of river otter sign recorded is positively correlated with DSS. Note that 1 DSS is adequate for conducting surveys.



a. Arrows designate the beginning of each time-block.

b. Horizontal bars designate the beginning and end of sunlight; long bar represents the longest day during the study period, short bar represents the shortest day during the study period.

c. Error bars are 1 S.E.

Figure 3. Mean diel activity pattern of river otters in southeastern Minnesota, 16 October 2002–28 February 2003. Over 280 hours tracking data was used for this analysis.

SURVIVAL, HOME RANGE CHARACTERISTICS, AND HABITAT SELECTION OF RIVER OTTER IN SOUTHEASTERN MINNESOTA

Thomas A. Gorman[1], John D. Erb, Brock R. McMillan[1], Christopher S. DePerno, and Daniel J. Martin[1].

INTRODUCTION

River otters *(Lontra canadensis)* are a top-predator in aquatic ecosystems (Ben David et al. 2001) and a valuable furbearer species (Melquist and Hornocker 1983). River otter populations in many regions of North America experienced a population decline, likely caused by anthropogenic factors such as land use changes, unregulated harvest, and water pollution (Nilsson 1980, Towell and Tabor 1982, Melquist and Dronkert 1987, Raesly 2001). River otter populations may be slow to recover from impacts, partly a result of relatively low natality. Improvements in water quality, habitat management, population monitoring, and successful reintroduction programs, have contributed to the recovery of river otters across much of their historical range (Raesly 2001). The recovery of populations of river otter across their historic range may have strong implications for a rebound in proper ecosystem function in both terrestrial and aquatic systems (Bowyer et al. 2003).

River otters are indigenous to Minnesota and were historically distributed statewide (Swanson et al. 1945). The river otter was an unprotected species in Minnesota prior to 1917, at which point the species received complete protection. In 1943, limited trapping began and was legal in only three years until 1953, at which point a two-week, annual season was implemented in the northern portion of Minnesota (Landwehr 1985). The southern region of the state has remained closed to legal harvest. The protected status of river otter in the southern half of Minnesota has contributed to an increase of the population in the region (Erb and DePerno 2000), though this increase appears to have been slow. Survival estimates and determination of cause-specific mortality are essential to determine specific factors that may limit population growth (White and Burnham 1999) and to adequately manage wildlife populations (White and Garrot 1990).

River otters have few natural predators, with most mortalities a result of anthropogenic factors such as car collisions and trapping (Melquist 2003). The river otter's piscivorous diet may expose populations to indirect mortality agents as a result of water pollution (Ben David et al. 2001). Diseases may also contribute to mortality, but are not thought to be a major factor (Melquist and Dronkert 1987, Melquist 2003).

Home range estimates are used to understand the organization of animals through space and time (Kernohan et al. 2001) and provide information on the spatial needs of river otters (Kernohan et al. 2001). Otter are typically more social than other mustelids (Melquist and Dronkert 1987). Thus, knowledge about individual home range characteristics will assist in describing home range overlap between individuals, spatial requirements, site fidelity, and resource use for otter in southern Minnesota.

[1] Dept. of Biol. Sci., MSU; 242 Trafton Science Center S.; Mankato, MN 56001

The objectives of this study were to determine the influence of age, sex, and season on survival, cause-specific mortality, home range characteristics, and habitat selection of river otters in southeastern Minnesota.

STUDY AREA

This study was conducted along the Mississippi River drainage in Winona and Wabasha Counties in southeast Minnesota (Figure 1). The majority of the research was performed on the McCarthy Lake Wildlife Management Area (MWMA), Whitewater Wildlife Management Area (WWMA), and the Upper Mississippi River National Wildlife and Fish Refuge (UMNWFR). Also, the backwaters of the Mississippi River and the major tributaries (i.e., primarily the Zumbro River and the Whitewater River) which flow into the Mississippi River from the west were included in the study area.

The topography of the study area was predominantly blufflands with as much as 183 meters of relief. The blufflands are a bedrock plateau covered with a windblown layer of silt that has been significantly eroded by rivers. Historically, the study area was dominated by black oak (*Quercus velutina*), jack pine (*Pinus banksiana*), shagbark hickory (*Carya ovata*), and American basswood (*Tilia americana*) on poorly drained slopes; red oak (*Quercus rubra*), American basswood, and black walnut (*Juglans nigra*) in the deep valleys; and tallgrass prairie on the ridges and in the drier valleys.

Annually southeast Minnesota receives 87.8 cm of precipitation with an annual mean temperature of 8.0 °C, a mean minimum annual temperature of 2.8° C and a mean maximum annual temperature of 13.1° C (Garoogian 2001).

METHODS

Animal Capture and Monitoring

River otters were captured in fall and spring beginning in fall 2001 and ending in fall 2003. All handling procedures were approved by the Minnesota State University, Mankato Institutional Animal Care and Use Committee. Otter trapping occurred at areas with a high-intensity of use, such as crossover trails (e.g., trails traveling across land between two bodies of water) and latrine sites. Sleepy Creek® #11 double-jawed foothold traps (Sleepy Creek Manufacturing, Berkley Springs, WV) were used to capture otters (Shirley et al. 1983, Blundell et al 1999). After an otter was captured it was transferred from the trap to a transport tube, and taken to Plainview Veterinary Clinic for surgical implantation of a radio transmitter (Models: 1245 2-stage, 1250 2-stage, 1250 3-stage; Advanced Telemetry Systems, Inc.(ATS), Isanti, MN).

Prior to surgery, otters were administered a combined intramuscular injection of ketamine ($\overline{X}$ = 16.77 mg/kg) and xylazine ($\overline{X}$ = 10.20 mg/kg). The radio transmitters were surgically implanted into the peritoneal cavity through a paralumbar incision. While under anesthesia, an upper premolar was extracted for aging by cementum annuli (Kuehn and Berg 1984), and a blood sample was drawn for DNA and toxicology analysis. Sex, weight, head circumference, chest circumference, length of right hind foot, total body length, condition and wear of teeth, and overall body condition were recorded. Otters were ear tagged with number 1 monel ear tags and web tagged with number 3 monel web tags (National Band and Tag Company, Newport, KY). To minimize infection the otters received 2cc of long acting penicillin, 1cc of baytril, and 2cc of clostridium anti-toxin. Otters were allowed to naturally recover from anesthesia, and were released at the site of capture within 6 to 74 hours.

River otters were radio-tracked 2 days per week from the ground using an ATS R4000 scanning receiver and a three-element Yagi antenna via triangulation and homing methods. Triangulation was conducted using ≥ 2 bearings from known locations within 15 minutes. Locations obtained using triangulation data were analyzed using Locate II (Nams 1990). Also, radio tracking was conducted at approximately 7-10 day intervals via Cessna Skylane 182 equipped with a four-element Yagi antenna on each wing. Locations of river otters were collected during all seasons for the duration of the study.

Mortality

Cause specific mortality was determined for all animals within 48 hours of the mortality, except for one juvenile female that was located 89 days after the previous location. This animal had moved a considerable distance from the study area and was not located for several fights. A necropsy was performed by a licensed veterinarian if the cause of death of each otter was not obvious from field observations.

Survival

We used Program MARK (White and Burnham 1999) to model factors that influenced survival and then used these models to estimate survival on the study area. We *a priori* selected 9 models (Table 1) that we felt were the most biologically relevant to river otter survival on our study area. Models were based on age, sex, and seasonal parameters. Otters were placed in 1 of 4 age and sex classes; adult male (males > 2 years old), adult females (females > 2 years old), sub-adult males (males < 2 years old), and sub-adult females (females < 2 years old). Seasons were established based on biological and managerial considerations. We used an information theoretic approach using Akaike's Information Criteria (AIC) to examine the relative strength of each model. The model with the lowest AIC value was considered the model with the best balance between statistical parsimony and goodness of fit for the empirical data. We calculated AIC corrected for small sample size (AIC_c). Based on guidelines established by Burnham and Anderson (2002) we assumed that models with Δ AIC_c values < 2.0 represented a model with relative support. Model averaging was used to estimate survival for each group.

Home ranges

We estimated annual 95% home ranges and 50% core areas using a fixed kernel estimator with least squares cross validation. Home ranges and core areas were calculated for animals with > 35 locations over > 4 months of tracking. Fixed kernel home ranges and core areas were estimated using Animal Movements extension (Hooge and Eichenlaub 1997) in ArcView 3.3 (Environmental Systems Research Institute, Inc., Redlands, CA).

RESULTS

We captured 39 river otters 42 times (20 females; 19 males) (Table 2). The number of otter captured per 100 trap nights decreased two-fold from 1.2 otter captured per 100 trap-nights in Fall 2001 to 0.57 otter captured per 100 trap-nights in Fall 2003 (Figure 2). We monitored survival of river otters over a 2 year period from April 2002 through March 2004, and recorded a total of 403 river otter-months during this period.

Mortality

The main causes of river otter mortality on the study area were from human related activities (Table 3). We encountered a total of 9 mortalities of which, incidental captures by fur-harvesters was the primary cause and resulted in 6 deaths (Table 3). One animal was killed after it was stuck by an automobile, 1 animal died from an infection, which was attributed to natural causes, and 1 animal died from unknown causes (Table 3).

Survival

There were three models that had Δ AIC_c values ≤ 2, and therefore warrant further explanation (Table 3). The *sex model* received the greatest amount of support from the empirical data (Δ AIC_c = 0.0, W_i = 0.38) relative to the *a priori* selected group of models. This model represented a comparison between male and female survival and did not include an effect of time or age. The second highest ranking model (Δ AIC_c = 1.26, W_i = 0.20), *Trapping*Sex*, represented a 2-season year and highlighted the influence of the trapping season versus the non-trapping season as an influence on otter survival and the interaction of this season on otter sex. The next model in the set (Δ AIC_c = 2.02, W_i = 0.14), was the same as the previous model except with out the sex interaction. The sex parameter had an effect size of 0.68, which was 1.5 times greater than the effect size of the 2-season trapping parameter, and 3.6 times greater than age parameter. However, because the other 6 models accounted for 0.36 of the model weights we chose to model average to estimate the sex and age specific survival. Survival of sub-adult females (S = 0.709, SE = 0.132) was similar to adult females (S = 0.733, SE = 0.122), but was lower than sub-adult males (S = 0.891, SE = 0.088) and adult males (S = 0.889, SE = 0.086; Figure 3) when estimated on an annual basis.

Home ranges

From 1 June 2002 through 31 May 2003 we collected 1480 locations on 19 individuals ($\overline{X}$ = 77.9/individual; range = 46 - 103/ individual). Fixed kernel core areas of females ($\overline{X}$ = 176.26ha, SE = 70.30ha, n = 9) on average were smaller than males ($\overline{X}$ = 395.67ha, SE = 142.65ha, n = 10) (Figure 4), but there was a large amount of variation in the sizes of core areas for both groups. Likewise, home ranges were also smaller for females ($\overline{X}$ =1134.56ha, SE = 320.97ha, n = 9) than for males ($\overline{X}$ = 2862.43ha, SE = 1078.64ha, n = 10; Figure 4) but there was also a large amount of variability.

FUTURE PLANS

Using our survival data, and previously reported natality estimates from northern Minnesota (n = 267 carcasses) and Wisconsin (n = 787 carcasses), we will evaluate population growth in the study site using a Leslie population matrix. Parameter sensitivity and elasticity will be examined to better understand factors contributing to population fluctuations.

Estimated locations from triangulation data, homing, and aerial telemetry from the second year of data collection (ending in May 2004) will be combined to estimate home ranges for each individual otter using the fixed kernel estimator. Fixed kernel home ranges will be estimated using the same methods as described above. Spatial overlap between otters will be examined using permutation tests in Blossom Statistical Software (Cade and Richards 2001). Individual otter overlap will be investigated between season, sex, and age. We will also investigate home range fidelity for animals that were tracked for the entirety of the study.

We will examine habitat selection at the second order, or landscape scale, and at the third order, or home range scale (Johnson 1980). For these analyses, we will compare used locations to a random set of locations using a logistic regression approach (Manly et al. 2002). Variables incorporated will focus on diversity of wetland types, stream characteristics, and type and amount of edge. ArcView 3.3 will be used to quantify habitat features for both random and observed locations at both spatial scales.

ACKNOWLEDGEMENTS

We thank the Minnesota Department of Natural Resources, United States Fish and Wildlife Service, and Minnesota State University, Mankato, for supporting this research. We thank M. Fischer, S. Gappa, J. Homyack, and J. McIntire for field assistance. We thank D. Carpenter, B. Maas, M. Trenholm, and J. Heineman for their aerial telemetry expertise, and Dr. Mike Morath for surgically implanting otters with radio transmitters. We thank J. Fieberg, J. Giudice, and J. Homyack for their statistical assistance, and R. Wright for GIS assistance. We thank the personnel at Whitewater Wildlife Management Area and the Winona District United States Fish and Wildlife Service office for all of their cooperation and assistance.

LITERATURE CITED

Ben-David, M., L. K. Duffy, G. M. Blundell, and R. T. Bowyer. 2001. Exposure of coastal river otters to mercury: relation to age, diet, and survival. Environmental Toxicology and Chemistry 20:1986-1992.

Berg, W. E. 1984. Otter Status Report, 1983-1984. Minnesota Department of Natural Resources, Grand Rapids, MN. 4pp.

Blundell, G. M., J. W. Kern, R. T. Bowyer, and L. K. Duffy. 1999. Capturing river otter: a comparison of Hancock and leg-hold traps. Wildlife Society Bulletin 27:184-192.

Bowyer, R. T., G. M. Blundell, M. Ben-David, S. C. Jewett, T. A. Dean, and L. K. Duffy. 2003. Effects of the *Exxon Valdez* oil spill on river otters: Injury and recovery of a sentinel species. Wildlife Monographs. 153:1-53.

Burnham, K. P. and D. R. Anderson. 2002. Model Selection and Inference: A Practical Information-Theoretic Approach. Second Edition. Springer – Verlag, New York, NY.

Cade, B. S., and J. D. Richards. 2001. User Manual for Blossom Statistical Software. Midcontinent Ecological Science Center, U. S. Geological Survey, Fort Collins, CO.

Erb, J. D., and C. S. DePerno. 2000. Distribution and relative abundance of river otter in southern Minnesota. Pages 19-26 *in* B. Joselyn (ed.), Summaries of Wildlife Research Findings. Section of Wildlife, Minnesota Department of Natural Resources, St. Paul, MN.

Garoogian, D., editor. 2000. Weather America: a thirty year summary of statistical weather data and rankings. Grey House Publishing, Inc., Millertown, NY.

Hooge, P. N., and B. Eichenlaub. 1997. Animal movement extension to ArcView. version 1.1. Alaska Science Center - Biological Science Office, U.S. Geological Survey, Anchorage, AK.

Johnson, D. H. 1980. The comparison of usage and availability measurements for evaluating resource preference. Ecology 61:65-71.

Kernohan, B. J., R. A. Gitzen, and J. J. Millspaugh. 2001. Analysis of animal space use and movements. Pages 125-166 *in* J. J. Millspaugh and J. M. Marzluff (eds.), Radio Tracking and Animal Populations. Academic Press, San Diego, CA.

Kuehn, D. W. and W. E. Berg. 1983. Use of radiographs to age otters. Wildlife Society Bulletin 11:68-70.

Landwehr, T. J., compiler. 1985. A summary of Minnesota game regulations, harvests, and population statistics, 1858-1983. Unpublished Report. Section of Wildlife. Minnesota Department of Natural Resources. St. Paul, MN.

Manly, B. F. J, L. L. McDonald, D. L. Thomas, T. L. McDonald, and W. P. Erickson. 2002. Resource Selection by Animals: Statistical Design and Analysis for Field Studies (2nd ed.). Kluwer Academic Publishers, Dordrecht, The Netherlands.

Melquist, W. E, P. J. Polechla Jr., and D. Toweill. 2003. River otter. Pages 708-734 *In* G. A. Feldhamer, B.C. Thompson, and J. A. Chapman (eds.), Wild Mammals of North America: Biology, Management, and Conservation, 2nd edition. Johns Hopkins University Press, Baltimore, MD.

Melquist, W. E. and A. E. Dronkert. 1987. River otter. Pages 625-641 *in* M. Novak, J. A. Baker, M. E. Obbard, and B. Malloch (eds.), Wild Furbearer Management and Conservation in North America. Ontario Trappers Association.

Melquist, W. E. and M. G Hornocker. 1983. Ecology of river otters in west central Idaho. Wildlife Monograph 83:1-60.

Nams, V. O. 1990. LOCATE II user's guide (version 1.82). Pacer, Truro, Nova Scotia, Canada.

Nilsson, G. 1980. River otter research workshop. Florida State Museum. 26 pp. Gainesville, FL.

Raesly, E. J. 2001. Progress and status of river otter reintroduction projects in the United States. Wildlife Society Bulletin 29:856-862.

Shirley, M. G., R. G. Linscombe, and L. R. Sevin. 1983. A live trapping and handling technique for river otter. Proceedings of the Animal Conference for the Southeast Association of Fish and Wildlife Agencies 37:182-189.

Swanson, G., T. Surber, and T. S. Roberts. 1945. The Mammals of Minnesota. Minnesota Department of Conservation Technical Bulletin No. 2, St. Paul, MN. 108pp.

Toweill, D. E., and J. E. Tabor. 1982. The northern river otter *Lutra canadensis* (Schreber). Pages 688-703 *in* J. A. Chapman and G. A. Feldhamer (eds.), Wild Mammals of North America. Johns Hopkins University. Press, Baltimore, MD.

White, G. C. and K. P. Burnham. 1999. Program MARK: Survival estimation from populations of marked animals. Bird Study 46 Supplement, 120-138.

White, G. C. and R. A. Garrot. 1990. Analysis of Wildlife Radio-Tracking Data. Academic Press, San Diego, CA.

Table 1. Nine models used to assess factors influencing survival in southeastern Minnesota, 2002 - 2004.

Model	Definition
Sex	Male and female
Age	Sub-adult and adult
Sex*Age	Sex and age interaction
Trapping	Legal trapping season (November – May)
Trapping*Sex	Trapping season and sex interaction
Trapping*Age	Trapping season and age interaction
Trapping, November	Trapping season with emphasis on Nov.
Biological Seasons	Pup rearing, summer, and winter
Global	Sex, age, and biological season interaction

Table 2. Sex and age class of river otters captured during spring and fall trapping periods, 2001 - 2003.

	Age Class	
Sex	Sub-adult	Adult
Male	6	13
Female	12	8

Table 3. Cause-specific mortality of river otters in southeastern Minnesota, 2002-2004

Cause of Death	Age Class Sub-adult	 Adult
Trapping	3F	3F
Road-kill	-	1M
Unknown	1F	-
Natural	-	1F

Table 4. The relative strength of evidence to support factors influencing survival of radio-marked river otters (*Lontra canadensis*) based on Akaike's Information Criterion (AIC), the change in AIC (Δ AICc), model weight (W_i), and the number of parameters in each model (K) for southeastern Minnesota, 2002-2004.

Model	AIC_c	$\Delta\ AIC_c$	W_i	K
Sex	78.50	0.00	0.38	2.0
Trapping*Sex	79.76	1.26	0.20	3.0
Trapping	80.52	2.02	0.14	2.0
Sex*Age	81.22	2.72	0.12	4.0
Trapping*Age	82.07	3.58	0.10	3.0
Trapping, November	82.45	3.95	0.06	3.0
Biological Seasons	82.73	4.23	0.05	3.0
Age	83.86	5.37	0.03	2.0
Saturated	303.25	224.76	0.00	101.0

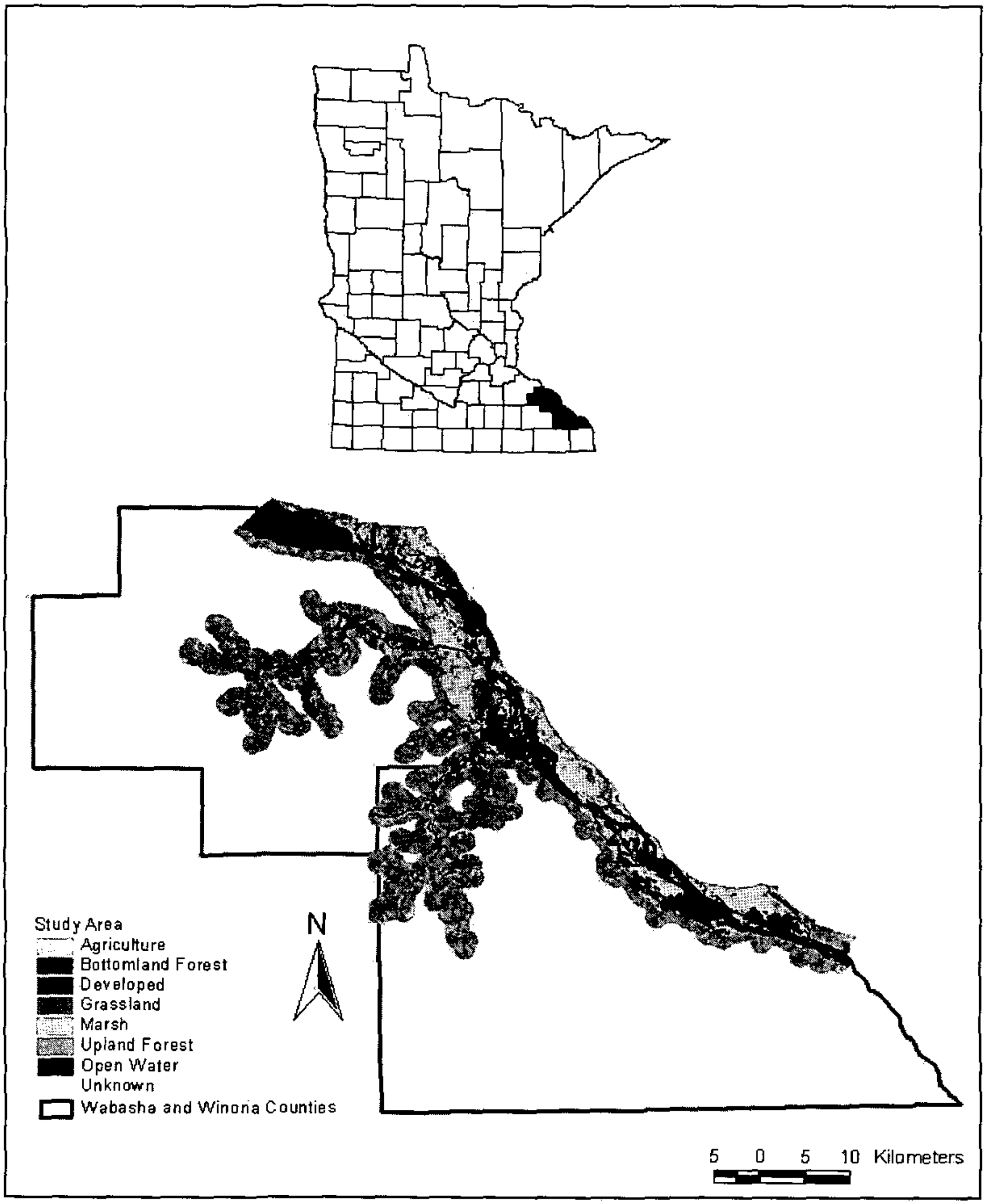


Figure 1. River otter study area in southeastern Minnesota, 2002 - 2004.

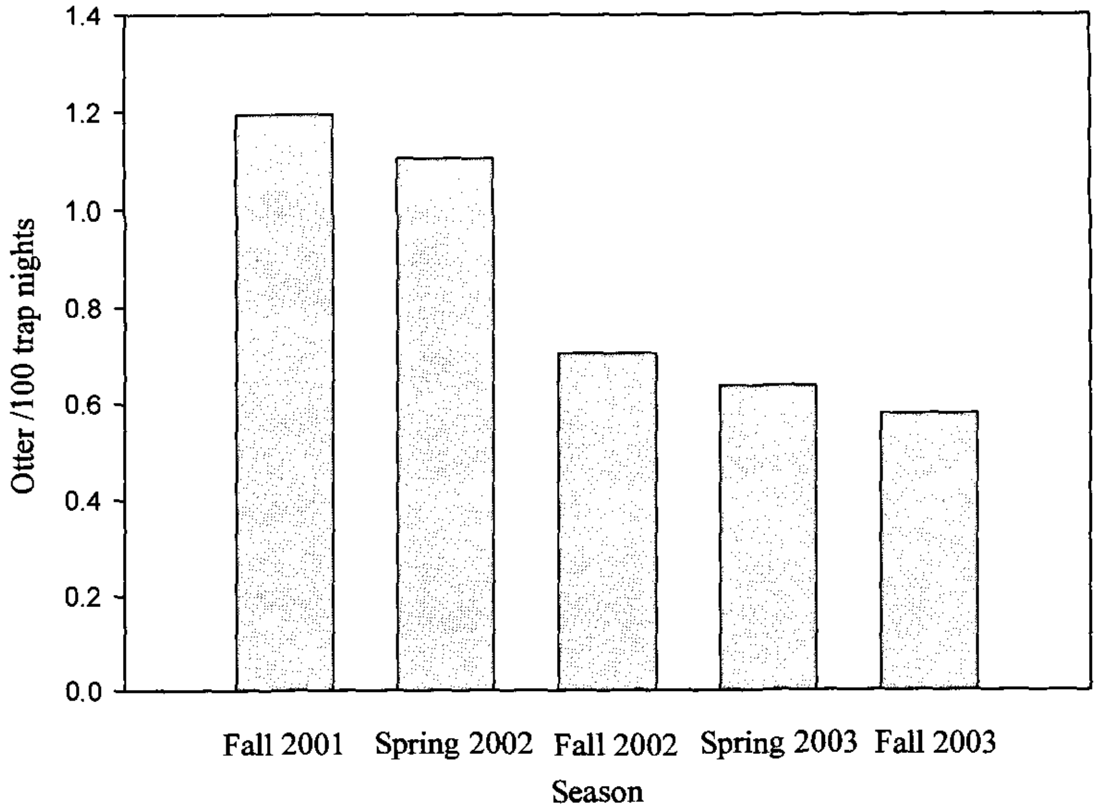


Figure 2. Number of river otters captured/100 trap nights during trapping events, beginning in fall 2001 and ending in fall 2003, in southeastern Minnesota.

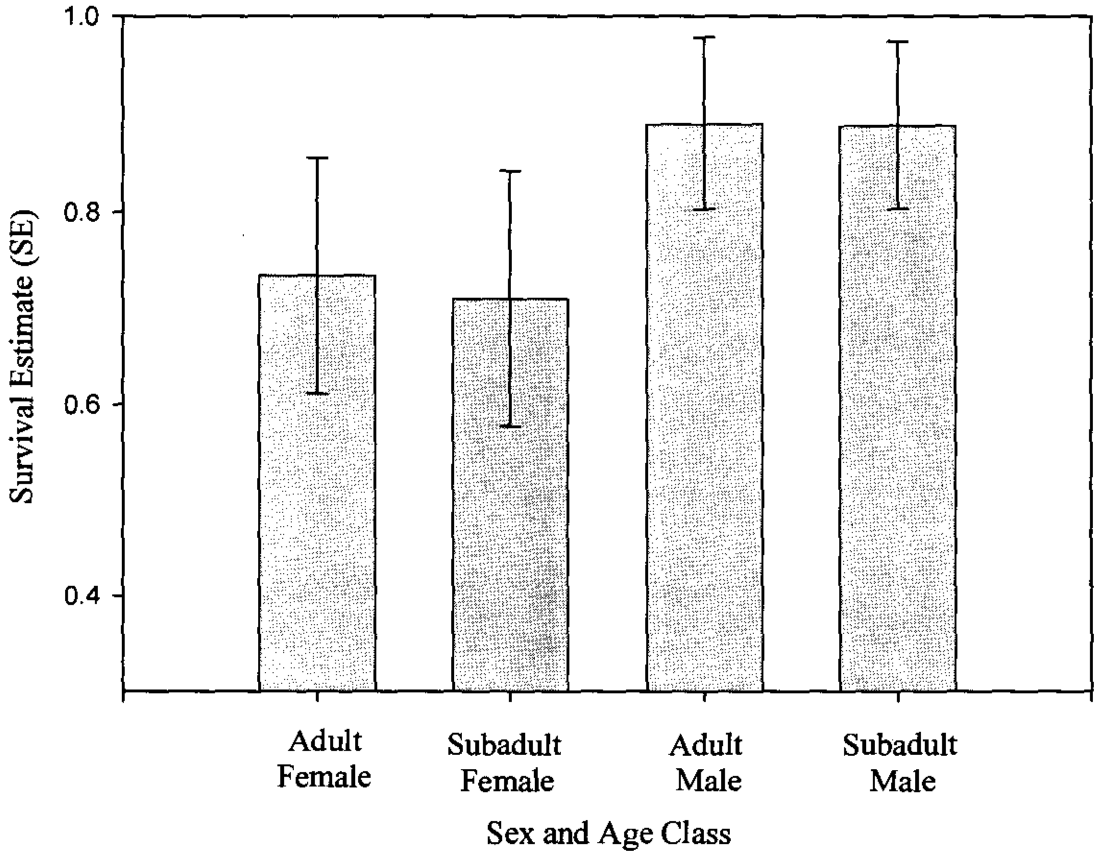


Figure 3. Annual survival and standard errors (SE) of 39 radio-marked river otters (*Lontra canadensis)* by sex and age class (adult males (males > 2 years old), adult females (females > 2 years old), sub-adult males (males < 2 years old), and sub-adult females (females < 2 years old)) based on a 12-month average from a 26-month period in southeastern Minnesota, April 2002- May 2004

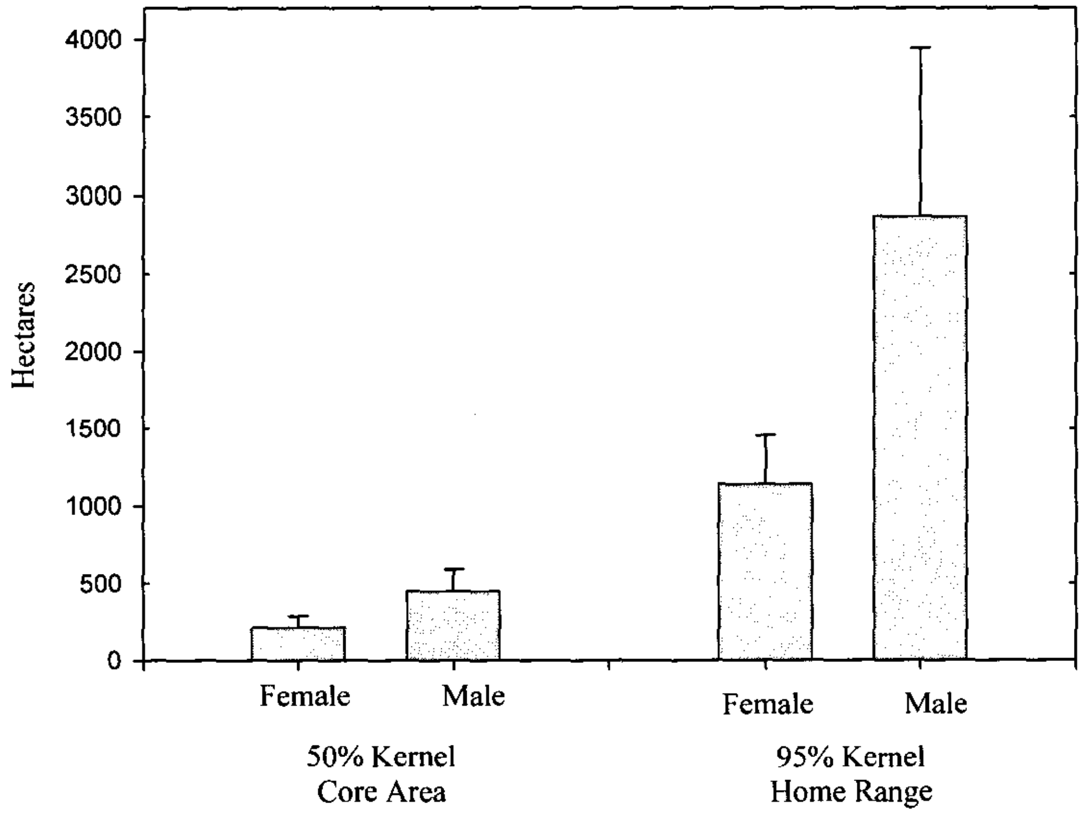


Figure 4. Annual 95% fixed kernel home ranges and 50% core areas of male and female river otters in Southeastern Minnesota, 2002 - 2003.

ECOLOGY AND POPULATION DYNAMICS OF BLACK BEARS IN MINNESOTA

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

Since the summer of 1981, we have been conducting a telemetry-based study of black bears in the Chippewa National Forest (CNF), near the center of the Minnesota bear range. Other satellite bear projects were initiated in 1991 near the southern fringe of the Minnesota bear range (Camp Ripley Military Reservation and Pine County), and in 1997 in Voyageurs National Park (VNP), on the Canadian border. All but the Pine County study have been continued to date.

We routinely visited radiocollared bears once or twice in their winter dens (December – March), mainly to observe condition and reproduction, and also periodically checked their status (alive, dead, dispersed) during the active season (April – October). From 1981 through completion of den visits in March 2003, a total of 613 individual bears were handled in and around CNF, 74 at Camp Ripley, 71 at VNP, and 50 in Pine County. As of April 2003, the start of the current year's work, we were monitoring 21 collared bears in the CNF, 9 at Camp Ripley, and 5 in VNP.

Principal objectives of this study include: (1) continued monitoring of reproduction and cub survival, (2) additional (improved) measurements of body condition, heart function, and wound healing, (3) examination of habitat use and movements with GPS telemetry, and (4) investigation of female dispersal near the southern fringe of the expanding bear range.

RESULTS

Trapping and Collaring

Trapping efforts this summer focused on capturing more female bears inhabiting lowland habitats in CNF and recapturing bears that had dropped radiocollars at Camp Ripley. We caught 6 bears and collared 4 in CNF and collared 2 bears in Camp Ripley. Another collared bear was killed during the hunting season outside Camp Ripley, and we caught and collared one of her orphaned cubs.

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. Four GPS-collared bears at Camp Ripley provided data this year, although 2 of these consisted of just a few months of information.

One GPS-collared adult male remained almost entirely within the bounds of Camp Ripley over the course of the year, but he made routine excursions across the Mississippi River along the eastern boundary to visit birdfeeders in early spring and also made frequent visits to probable hunter bait sites outside Camp during the fall (Fig. 1). He denned in Camp on November 9. Another adult male in this area, who had a GPS collar since late August, denned in Camp on October 12.

A 2-year-old female, whose mother was a resident of the Camp, was caught outside of Camp when she was visiting birdfeeders and coming up on decks of houses. Other young females monitored during this study have rarely dispersed from the vicinity of their natal home range. We attached a GPS collar to monitor this bear's movements in relation to human development and possible further dispersal from Camp. Her home range was centered about 6 miles west of Camp, straddling a 4-lane highway (Fig. 1). She crossed this highway a minimum of 31 times during a 4-month period; 75% of crossings were during 2000–0530 hours. This bear, pregnant in the fall of 2003, denned on October 18, about ¼ mile from a house.

One other uncollared bear, an ear-tagged 1-year-old male from Camp Ripley, also provided noteworthy movement data. He was seen by a hunter 42 miles south of Camp Ripley in September, 2003.

Mortality

Legal hunting has been the predominant cause of mortality among radiocollared bears in this study (>90%). 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, 7 of 25 (28%) collared bears from the CNF and 1 of 6 collared bears at Camp Ripley were shot by hunters.

In addition to these hunter-related mortalities, 1 young male from Camp Ripley was hit by a car 32 miles north of its previous year's den site, and 2 adult females died of unknown causes. The death of these 2 females was particularly puzzling, as natural mortalities of adult bears have been exceedingly rare during this study. Unfortunately, when we located the carcasses of these 2 bears, they were too decomposed to investigate causes of death; however, none of the bones were broken suggesting that they were likely not struck by a vehicle, and a metal detector found no evidence of them being shot.

Reproduction

Three CNF females produced their first litters in 2004; two were 4 years old and 1 was an unknown age (tooth not yet sectioned). Neither of the two collared 3-year-olds in CNF produced cubs. Since the beginning of this study, 43% of 4-year-olds in the CNF have produced cubs (plus 3% of 3-year-olds). At Camp Ripley, where hard mast (especially oak) is more abundant, bears have a somewhat earlier age of first reproduction than in CNF. This year, like last year, one of two 3-year-olds had cubs when we visited her den in March. The other 3-year-old had no cubs present, but her mammae showed signs of having been suckled. A scat at her den site had cub claws, indicating that she had consumed her litter. In VNP, all 5 collared adult bears failed to produce in 2002, but did produce in 2003 and all were with yearlings this year (so were unavailable to produce cubs).

Litter size tends to be less responsive to food conditions than age of first reproduction. However, litter sizes in VNP, where food conditions are poorest, appear to be somewhat lower than in CNF and Camp Ripley (Tables 1–3). Litter size also appears to be most variable in VNP. Average litter size in CNF has remained stable at 2.6 cubs since the mid-1980s.

We checked litters in their mother's den a year after they were born to assess cub mortality; we assume that all missing cubs died. Since 1981, 83% of cubs born to collared mothers in the CNF survived. Sample sizes were too small to compare yearly cub survival (Table 1). Overall cub survival at Camp Ripley (75%; Table 2) was similar, but cub survival at VNP (67%; Table 3) was significantly lower than CNF ($P = 0.04$). This year's cub survival in VNP, however, was 92% (12 of 13).

We have generally been unable to determine causes of mortality of cubs because we did not collar them. However, for the first time during this study, an eartagged cub from Camp Ripley was reported struck and killed by a car.

Mortality of male cubs has averaged about twice that of females in all areas (23% M vs 10% F in CNF; 33% M vs 17% F in Camp Ripley; 40% M vs 25% F in VNP). Sex ratios at birth were skewed towards males in all areas (52–57%; Tables 1–3), so the higher cub mortality for males resulted in a near 50:50 sex ratio among yearlings.

Heart Function and Wound Healing of Hibernating Bears

Since 2001 we have been collaborating on a study of heart function in hibernating bears with two experts in the field, Dr. Paul Iaizzio (University of Minnesota) and Dr. Tim Laske (Medtronic). We continued that work this year. Five bears were studied in December 2003 and then again in March 2004. Heart function was measured with ultrasound imaging and a 12-lead EKG. Although these bears were in deep hibernation, no differences were observed between December and March in electrophysiological parameters. Heart rates showed a marginal increase (mean = 113 bpm to 129 bpm).

Tests of wound healing were conducted by removing a plug of skin (~0.5 cm diameter) and subsequently examining the healing process. In all cases, these wounds completely healed from December to March, with no evident scarring.

Current Monitoring

After completion of den visits in spring 2004, 35 bears (22 in CNF, 5 in Camp Ripley, 8 in VNP) were radiocollared, including 9 with GPS collars. These bears will be monitored for mortality periodically during the active season, and then tracked to their 2004–5 den sites.

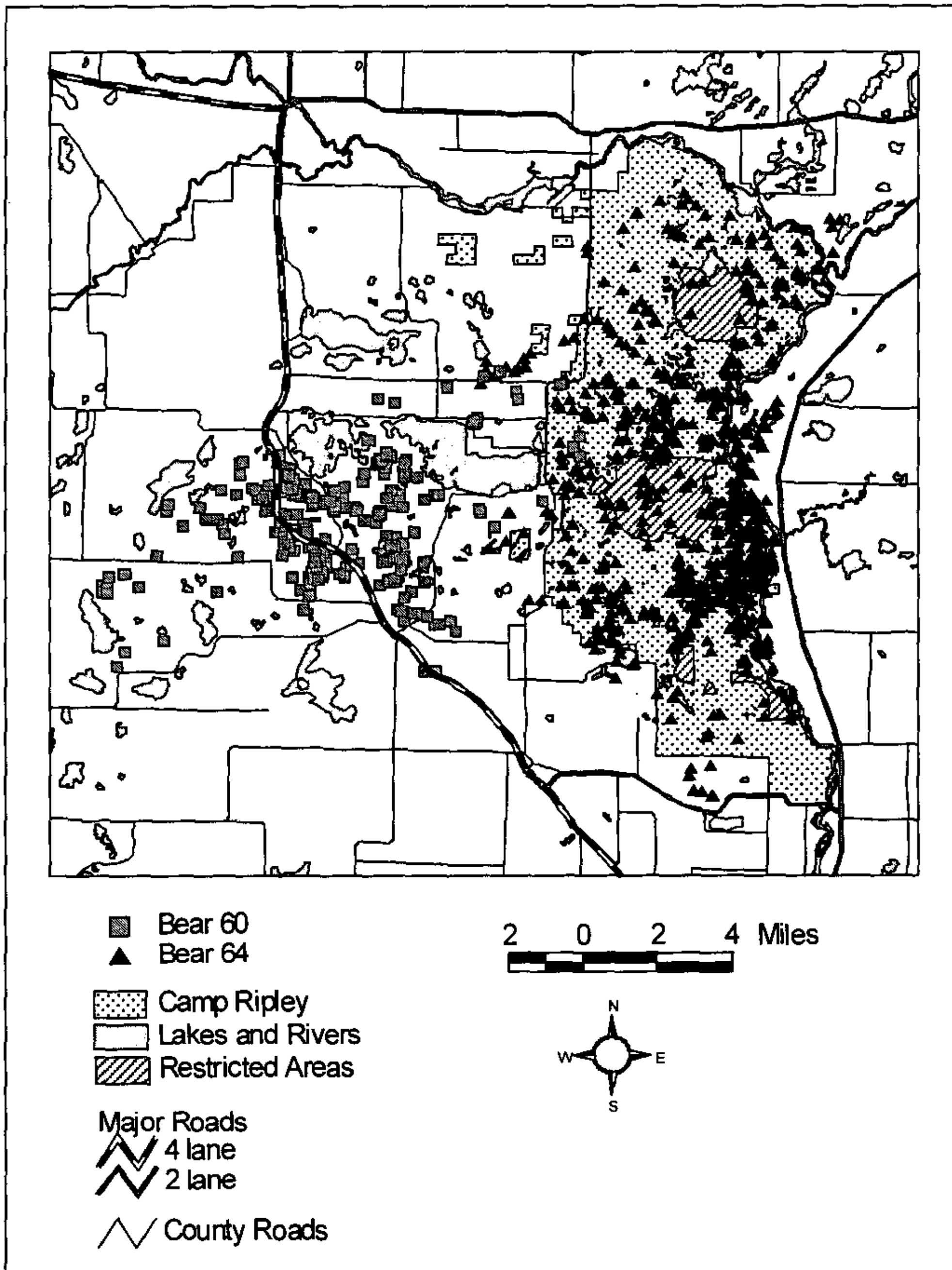


Figure 1. GPS locations of a 2-year-old female bear (bear 60) that illustrate her dispersal from her natal home range on Camp Ripley and her extensive use of both sides of a 4-lane highway and the movements of an adult male bear (bear 64) that stayed predominantly within Camp Ripley boundaries.

Table 1. Black bear cubs examined in dens of radiocollared mothers in or near the Chippewa National Forest during March, 1982B2004.

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%	—
Overall	**162**	**419**	**2.6**	**52%**	**17%**

[a] Cubs that were absent from their mother=s den as yearlings were considered dead.
[b] Excluding 1 cub that was killed by a hunter after being translocated away from its mother.

Table 2. Black bear cubs examined in dens of radiocollared mothers in Camp Ripley Military Reserve during March, 1992B2004.

Year	Litters checked	No. of cubs	Mean cubs/litter	% Male cubs	Mortality after 1 yr
1992	1	3	3.0	67%	0%
1993	3	7	2.3	57%	43%
1994	1	1	1.0	100%	—[a]
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%[b]
2004	1	2	2.0	50%	—
Overall	**15**	**36**	**2.4**	**56%**	**25%**

[a] The only cub born to a collared female left its mother in early spring, due to human disturbance.
[b] Excluding 1 litter that surely died when the mother died in mid-May.

Table 3. Black bear cubs examined in dens of radiocollared mothers in Voyageurs National Park during March, 1999B2004.

Year	Litters checked	No. of cubs	Mean cubs/litter	% Male cubs	Mortality after 1 yr
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	—	—	—
Overall	**15**	**30**	**2.0**	**57%**	**33%**

Wetlands Wildlife Populations and Research Group
102 – 23rd Street
Bemidji, Minnesota 56601
(218) 755-2973

HUNTER ACTIVITY AND GOOSE HARVEST DURING THE SEPTEMBER 2003 CANADA GOOSE HUNT IN MINNESOTA

Stephen J. Maxson and Margaret H. Dexter

This report documents results of the 2003 September goose hunter mail questionnaire survey.

Methods

The Canada goose season in the four zones encompassing the majority of Minnesota was 6-22 September 2003 (17 days). A 10-day (6-15 Sep) season was held in the Northwest Goose Zone (Fig. 1). The daily bag limit was 5 geese per day, except in the Northwest Goose Zone and the Southeast Goose Zone where the daily bag was two. Shooting hours were 1/2 hour before sunrise to sunset. Taking of Canada geese was prohibited on or within 100 yards of all surface waters in the Northwest, Southeast, and Twin Cities Metro Goose Zones, in the Carlos Avery Wildlife Management Area and in the Swan Lake Area. This was the first year hunting on or within 100 yards of surface water was allowed in the Remainder of State Zone. In the Twin Cities Metro Zone and goose refuges open to goose hunting, hunting was not allowed from public road rights-of-way. Goose hunters were required to obtain a $4.00 permit to participate in the September season.

Permittees were randomly selected to receive a post-season hunter survey. Questionnaires were sent to 3,100 permittees following the season. Questionnaires were individually numbered, and up to 3 questionnaires were mailed to individuals who had not responded. Completed questionnaires were double key-punched to reduce errors.

This year the questionnaire form was shortened and simplified compared to that used in previous years. The questionnaire asked hunters which zone they hunted, number of days they hunted, number of geese taken, and number of geese knocked down and not retrieved for the season as a whole. Prior questionnaires had asked hunters to report the County hunted and harvest for each day of the season. The 2003 questionnaire also asked whether hunters in the West or Remainder of State Zones had hunted over water or within 100 yards of water and if so, how many geese they had taken.

Statistical Analysis Systems (SAS Institute Inc. 1999-2001, Version 8.2) computer programs were written to summarize responses to the questionnaire survey.

Results and Discussion

The DNR License Bureau reported that 42,009 Special Canada Goose Season permits were sold prior to 22 September, 2003. Response rate to the survey was 70.4% and 72.4% of the respondents indicated that they hunted during the September season. The majority of the hunters indicated they hunted in the Remainder Zone, followed by the West, Twin Cities Metro, Northwest, and Southeast goose zones (Table 1). The Remainder and West zones are the largest zones. Active hunters were afield an average of 2.8 to 3.8 days, and retrieved 2.0 to 3.0 geese, when totaled according to their hunt zone. Success was lowest for hunters hunting in the Twin Cities Metro Zone (59.1%) and highest in the Southeast Zone (75.6%) (Table 1).

A total of 80,988 Canada geese was harvested with approximately 59% of the harvest in the Remainder Zone and 22% in the West Zone (Table 1). This pattern has remained rather consistent during the 2000-2003 September seasons (Table 2). The U.S. Fish and Wildlife Service adjusts their mail survey statistics by a memory and prestige response bias factor of 0.848 for geese bagged in the Mississippi Flyway (Voezler et al. 1982:56). Multiplying September Canada goose harvest by the adjustment factor would indicate a 2003 harvest of 68,678.

Of those hunters who indicted that they hunted in the West or Remainder of State Zones (24,277 hunters, Table 1), 43.1% reported that they hunted over water or within 100 yards of water. Of the 65,936 geese harvested in these two Zones (Table 1), 31.7% were taken over water or within 100 yards of water. This was similar to the proportion of geese taken over water in the West Zone during the 2000-2002 September seasons (Table 3).

The September Canada goose season continues to provide an important part of Minnesota's total Canada goose harvest, and the expanded zones and experimental extensions have both helped increase harvest. Continued monitoring of both the magnitude of the harvest and size of the Canada goose breeding population will be important to ensure that management objectives are met.

LITERATURE CITED

Voelzer, J. F., E. Q. Lauxen, S. L. Rhoades, and K. D. Norman, editors. 1982. Waterfowl status report 1979. U.S.D.I. Fish Wildl. Ser. Spec. Sci. Rep. - Wildl. No. 246. 96pp.

Table 1. Permit sales, hunter activity, and harvest[a] by zone during the September Canada Goose season (6-22 September) in Minnesota, 2003.

Parameter	Northwest	West	Southeast	Twin Cities Metro	Remainder	Total
ALL ZONES						
Total permits sold						42,009
Questionnaires delivered						3,036
Useable questionnaires returned						2,138
% responding						70.4
Active hunters						1,548
% active hunters						72.40
BY ZONE						
% Distribution of hunters by primary hunt zone	4.52	24.46	2.60	13.06	55.36	100
%successful	71.2	64.5	75.6	59.1	65.4	64.9
Days/active hunter	2.84	3.29	3.48	3.58	3.81	
Geese/active hunter	2.04	2.39	2.98	2.49	2.86	
Unretrieved harvest/active hunter	0.22	0.36	0.90	0.36	0.44	
% unretrieved harvest	9.7	13.1	23.2	12.6	13.3	
EXPANDED:						
Active hunters	1,375	7,439	791	3,972	16,838	30,415
Hunter days	3,905	24,474	2,753	14,220	64,153	109,505
Retrieved Harvest:	2,805	17,779	2,357	9,890	48,157	**80,988**
Est. unretrieved harvest	302	2,678	712	1,430	7,408	12,530
Total harvest	3,107	20,457	3,069	11,320	55,565	93,518

[a]Harvest estimates not adjusted for memory/exaggeration bias.

Table 2. Retrieved harvest estimates by zone during the September Canada Goose season in Minnesota, 2000-2003.

Year	Northwest	West	Southeast	Twin Cities Metro	Remainder	Total
2000	2,750	18,909	1,183	15,594	51,685	**90,121**
2001	2,047	27,663	538	8,164	62,608	**101,021**
2002	1,568	22,075	848	8,504	50,769	**83,764**
2003	2,805	17,779	2,357	9,890	48,157	**80,988**

Table 3. Proportion of hunters hunting over water[1] and the proportion of Canada geese taken over water in the West Zone during the September season 2000-2002.

Year	% Hunting over water	% Geese taken over water
2000	46.7	30.6
2001	43.2	37.4
2002	44.9	35.1

[1]Over water or within 100 yards of water.

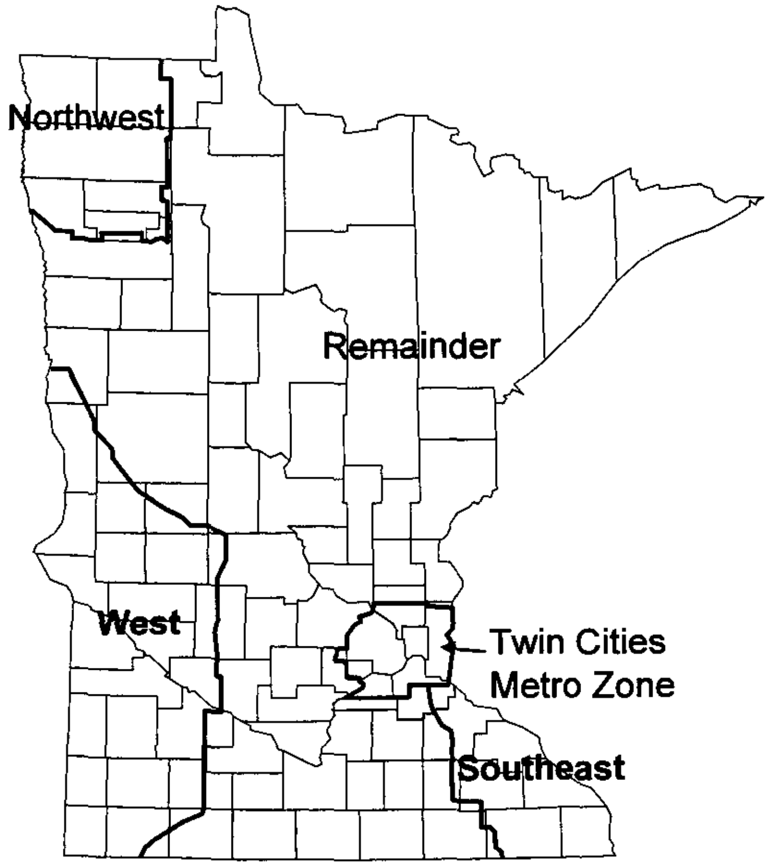


Figure 1. September season Goose Zones in Minnesota.

INFLUENCE OF LAND USE ON MALLARD NEST STRUCTURE OCCUPANCY

Michael C. Zicus, David P. Rave, Abhik Das[1], Michael R. Riggs[2], and Michelle L. Buitenwerf

ABSTRACT

We hypothesized that nest structure occupancy by mallards was a function of the type of nest structure and a number of landscape variables. From this model, we predicted that occupancy probability would increase as the amount and attractiveness of the surrounding nesting cover decreased and the size of the open water area in which a structure was deployed increased. Further, we predicted that the probability of structure occupancy would increase as the number of nearby structures decreased and the number of mallard hens with access to a structure increased. We also suspected that as the distance from a nest structure increased any landscape influence would diminish. We conducted an observational study to investigate the relationship between landscape composition and mallard occupancy of 2 types of nest structures. We used a geographic information system to quantify the covariates associated with the landscape and modeled nest occupancy as a function of several covariates using a hierarchical logistic regression model that allowed for both temporal and spatial correlation. Model deviance indicated that the model based on a landscape buffer with a 1.6 km radius performed the best. There were strong temporal correlations among periods during the nesting season, but no spatial correlations. The final model included an aggregate visual obstruction measurement (VOM) by period interaction, a quadratic effect for the size of the open-water area around a structure, and a year by period interaction. There was also a marginally significant nest structure type by period interaction. Neither pairs with access to a structure, nor the numbers of nearby structures were significantly associated with nest occupancy. Aggregate VOMs were positively associated with the probability of nest occupancy in the earlier part of the nesting season, but the pattern was reversed later in the nesting season. The probability of nest occupancy increased with the size of open-water area in which the structure occurred and reached an asymptote for open-water areas of ~15 ha or larger. Ultimately, we would like to develop a model whereby GIS data can be used to predict the probability of nest structure occupancy in different wetland locations. If we can combine this model with nest structure cost data, managers would have a tool that would enable them to deploy nest structures in a way that would maximize the benefit:cost ratio.

INTRODUCTION

Increased waterfowl production from private and public lands is an objective of the Prairie Pothole Joint Venture (United States Prairie Pothole Joint Venture Implementation Plan. 1989. Prairie Pothole Joint Venture Steering Committee. 64pp.). For mallards (*Anas platyrhynchos*), nest success appears to be a primary factor limiting recruitment in the prairies (Duebbert and Kantrud 1974, Cowardin et al. 1985, Klett et al. 1988).

[1]Statistical Research Division, Research Triangle Institute International, 6110 Executive Blvd., Suite 420 Rockville, MD 20852-3903, USA
[2]Statistical Research Division, Research Triangle Institute, 3040 Cornwallis Road, Research Triangle Park, NC 27709

The erection of mallard nest structures has become a popular management technique to improve mallard nest success. Extensive use and high nest success have been reported in some locations where structures are deployed. However, the effectiveness of structures in producing additional recruits in a landscape is likely to vary depending on the landscape.

A mallard management model (Johnson et al. 1987, Cowardin et al. 1988) will predict that mallards use a higher proportion of nest structures in landscapes having little attractive nesting cover versus landscapes that are rich in attractive cover (M. C. Zicus, unpubl. data). This prediction is reasonable, assuming structure density with respect to mallard pairs is similar, because structures are likely the most attractive nest sites hens have available to them in landscapes having little alternative cover. However, questions remain as to how to best distribute nest structures in the landscape. Guidelines to augment use of the mallard model have been developed for specific management practices (Habitat and Population Evaluation Team. 1993. Workshop information to develop waterfowl management strategy for Tewaukon Wetland Management District, North Dakota. unpubl. rep.). The guidelines suggest structures should be placed where pair water exists but where cover attractiveness in the immediate surrounding is marginal. These suggestions are largely untested, but form a starting point to assure that various management practices are used in what are believed to be appropriate settings.

We hypothesized that nest structure occupancy might be a function of: 1) type of nest structure under consideration, 2) amount and attractiveness of surrounding nesting cover, 3) amount of open water associated with a structure, 4) number of nearby nest structures, and 5) number of mallard pairs having access to a structure. From this model, we predicted that the probability of nest structure occupancy would increase as the amount and attractiveness of the surrounding nesting cover decreased. We expected the opposite relationship with the size of the open water area in which a structure was deployed because open water does not provide nesting cover. Likewise, we suspected that the probability of structure occupancy would increase as the number of nearby structures decreased and the number of mallard hens with access to a structure increased. We also suspected that as the distance from a nest structure increased any landscape influence would diminish and that landscape scale might be a question of interest. We conducted an observational study to investigate the relationship between landscape composition and mallard occupancy of 2 types of nest structures. We used a geographic information system (GIS) to quantify the covariates associated with the landscape. Our analysis also was designed to determine the size of the area around a structure that best predicted nest structure occupancy if a relationship between landscape composition existed. This knowledge should be useful to waterfowl managers interested in maximizing mallard use of nest structures.

K. Kotts and L. Lewis constructed nesting cylinders to replace those damaged at ice out. K. Kotts, L. Lewis, and D. Wells helped with project logistics. K. Andersson , J. Ferraro, J. Schlueter, and C. Vollbrecht measured vegetation height and helped maintain and check nest structures. K. Brennen provided access to concentrations of ground nesting mallards inside predator exclusion fences on the Fergus Falls Wetland Management District while T. Rondeau provided vehicles for nest searching. R. Wilken assisted with the nest searching and monitoring of mallard ground nests. L. Dedee helped obtain Global Positioning System (GPS) locations of nest structures on the study area. We also thank the landowners, who generously allowed access to their land, and D. Miller for digitizing and labeling land use polygons for our geographic information system (GIS).

STUDY AREA

The study area included 658 km^2 ($254mi^2$) in southern Grant and northern Stevens counties in western Minnesota (Fig. 1). The area is intensively cultivated and remnants of virgin prairie are extremely rare. Overall, landscape composition was quite similar during the study years (Table 1). Upland nesting cover for mallards was restricted primarily to scattered tracts of mixed native and exotic grasses and forbs on state (WMA) and federally (WPA) managed wildlife areas, grassy fields in agricultural set-aside programs such as the United States Department of Agriculture Conservation Reserve (CRP) and Water Bank (WBP) programs, and cover in roadside right-of-ways (ROW). Wetland drainage also has been extensive through the use of surface ditches and sub-surface drainage tiles (Prince 1997). Most remaining wetlands have permanent or semi-permanent water regimes due to the consolidation of temporary and seasonal wetlands into deeper, more permanent basins. Few mallard nest structures were present on the study area prior to the beginning of this study, and all were located on publicly managed areas. Mallard breeding pair density averaged ~4 pairs/km^2 (R. Johnson, unpubl. data).

METHODS

We used aerial photos to select wetlands throughout the study area that were candidates for nest structure placement. We subjectively assessed whether a wetland had much, a moderate amount, or little cropland within ~1 km. We then selected approximately equal numbers of wetlands from each cropland class for nest structure placement. We used this strategy to assure that structures with much or little surrounding cropland would be well represented in our sample. During spring 1996, one structure was placed in each of 78 wetlands while 2 structures were placed in each of 16 of the largest wetlands for a total of 110 structures. Nest structure type, either a single or double nest-cylinder structure (Delta Waterfowl Research Station, unpubl. rep.), was assigned randomly each time a structure was deployed resulting in 53 single- and 57 double-cylinder structures being placed in semi-permanent and permanent wetlands. Nest structures were deployed without a predator guard. We inspected each structure ≥ 4 times annually to record all nesting attempts and to determine exact hatch dates.

Geographic Information System

We developed a geographic information system (GIS) that included information for all of the covariates that we included in the analysis. These included descriptions of study area land use in 1997-1999, locations of all nest structures, and numbers of mallard pairs with access to each nest structure.

Delineating Land Use.--Tagged Image Format (TIF) files for each land survey section, created by scanning July-August U.S. Department of Agriculture Farm Services Agency (FSA) aerial photography, were geo-referenced and rectified. Root mean square (RMS) error (Environmental Systems Research Institute, Inc. 1996:288) for the rectified TIF files was assessed in meters. We initially performed 75 rectifications on 3 images to assess RMS error. Average RMS for the trial rectifications was 7.4m. Based on this trial, we set a maximum acceptable RMS error of 5.0m. When RMS exceeded 5.0m, we re-rectified the files to obtain a lower RMS and better edge matching among sections. Geo-referenced and rectified TIF files for each land survey section then served as a backdrop for digitizing land use polygons into shapefiles. We field inspected portions of most townships in the study area during March and April the following year to aid our photo interpretation of cropping patterns during the previous summer.

Certain types of linear features such as road surfaces, vegetated portions of road ROWs, and fence lines were created using buffering. Minnesota Department of Transportation road coverage was used to establish road locations, while location of features such as fence lines and ditches were established from a combination of the FSA images, Minnesota Department of Natural Resources (MNDNR) public land survey 40-acre parcel data, National Wetlands Inventory data, MNDNR hydrology data, and field notes using heads-up digitizing. Widths of small linear features that were buffered and transformed into polygons were determined from field measurements, visual estimates, or values used previously in Minnesota (Zicus and Rave 1998).

Nest Structure Locations.--Location of each nest structure on the study area was determined with a Rockwell Precise Lightweight Global Positioning System receiver. Differential corrected locations were obtained as Universal Transverse Mercator (UTM) coordinates using the NAD83 convention.

Mallard Pairs with Access to Structures.--The abundance of Mallard pairs with access to each nest structure was also modeled with a GIS (Reynolds et al. 1996; R. Johnson, U.S. Fish and Wildlife Service Habitat and Populations Evaluation Team, unpubl. data). We had no means to model each year separately, thus we assumed that the number of pairs with access to a structure was unchanged from year to year.

Cover Attractiveness.--We used visual obstruction measurements (VOMs) to index nesting cover attractiveness (Robel et al. 1970). VOMs were taken 5 times at 2-week intervals to describe chronologic changes in cover height and density throughout the nesting period. We estimated VOMs for those cover types that we believed to be most attractive to nesting mallards. We did not estimate VOMs for certain cover types such as woodlands, wetland vegetation, and cropland. For these cover types, we used date specific VOMs suggested for western Minnesota (Mack, G. D. 1991. The mallard model handbook. U.S. Fish and Wildlife Service, Bismarck Wetland Habitat Office. Unpublished Report revised by T. L. Shaffer and L. M. Cowardin, Zicus and Rave 1998). We used the median dates for first and second hay cutting reported by the Minnesota Crop and Livestock Reporting Service to reflect points in time when most hay was harvested.

We estimated VOMs in upland fields bisected by or included within an arc 0.4 km from a nest structure. We measured grassy fields in CRP, WBP, WMA, WPA, Reinvest In Minnesota (RIM) easements, and other grassy areas such as pastures not planted for conservation purposes (OG). Three sampling clusters were established along the longest straight-line diagonal across a field (Fig. 2). Sampling cluster starting points were established at the 3 quarter-points along the diagonal, and these were permanently marked with stakes. Each sampling cluster had 4 sampling points that were 20 m north, east, south, and west of a starting point. At each sampling point, vegetation height and density was measured in each cardinal direction (Robel et al. 1970). This provided 48 measurements from each field on a given date. VOMs were also determined for the portions of ROWs vegetated by upland cover. Six sampling clusters were established for each 1.6 km road segment bisected by the 0.4 km radius arc around a nest structure. Sampling cluster starting-points were located at the 3 quarter-points along each side of the road segment. Each ROW sampling cluster had either 1 or 3 sampling points. If the ROW ditch had a front and back slope, then 3 points were sampled. These were located at the quarter-points between the road edge and the field edge. If the ROW had only a front slope, then 1 point was sampled halfway between the road edge and the field edge. At each ROW sampling point, VOM measurements were taken in the 2 directions parallel to the road. This provided a minimum of 12 measurements and a maximum of 36 measurements for each road segment. We also measured the distance between the road edge and the field edge at each ROW sampling cluster. The width of the road surface was measured at the halfway point along the road segment. We combined CRP, RIM, and WBP fields into a single class (hereafter CRP) because of the similarity of these land uses. For each field or road segment that was sampled, date-specific means were calculated. From these values, overall date-specific means were estimated for each field or road segment type, and means were assigned to fields or ROWs throughout the study area that we did not measure.

Model Fitting

We used logistic regression to model nest occupancy, a binary (yes/no) outcome, as a function of several covariates. This outcome was assessed on the same nesting structures during different time periods, presumably producing temporal correlation. In addition, we were also concerned about possible spatial correlation that might be present due to land use features having spatial structure (Cressie and Chan 1989, Breslow and Clayton 1993, Ver Hoef and Cressie 2001). Since standard logistic regression used to model binary outcomes (McCullagh and Nelder 1989) does not allow for any correlations in the data, we accounted for such spatio-temporal correlations by using a hierarchical logistic regression model (Breslow and Clayton 1993). This method used a logistic regression framework, which is natural for binary data, while the hierarchical aspect allows for the likely presence of (positive) temporal and spatial correlations. Though we needed to account for these correlations, they were not our primary interest and were essentially nuisance parameters that hindered proper inference. The regression formulation enabled us to reduce these correlations by the inclusion of relevant time trends and appropriate covariates in the model. This hierarchical logistic regression allowed us to spatio-temporally model the occupancy of nesting structures by mallards as a function of land use features, type of nesting structure, number of nearby nesting structures, number of mallard pairs with access to the structures, and area of open water. It also enabled prediction and cross-validation through the fitted regression model.

Vegetative cover height and density measurements and nest establishment dates were temporally misaligned because VOMs were available for only 5 dates during the nesting season. To address this temporal misalignment, we partitioned each year into 4 time periods within which we assessed whether nesting occurred in a structure. For each year, these periods were March 15 to April 20, April 21 through April 30, May 1 through May 20, and May 21 through June 30. These intervals were chosen after preliminary exploration of the data, so that temporal misalignment was minimized and enough nest establishments occurred during each time period to make statistical modeling computationally feasible.

The model we fitted can be expressed as follows. Suppose Y_{ij} is a binary indicator variable taking the value 1 when a nest initiation occurred in a structure i during time point (year/period) j (i=1,…,n; j=1,…,t), and 0, otherwise. Letting $p_{ij} = \Pr(Y_{ij} = 1)$, our model can be expressed as:

$$(Y_{ij}|\theta_i,\lambda_j) \sim \text{Bernoulli}(p_{ij}), \text{ where } \log\left(\frac{p_{ij}}{1-p_{ij}}\right) = \underline{x}_{ij}{}'\underline{\beta} + \theta_i + \lambda_j,$$

$$\underline{\theta} \sim N(\underline{0}, \mathbf{V}_s(\sigma_s^2, \rho_s)) \text{ and } \underline{\lambda} \sim N(\underline{0}, \mathbf{V}_t(\sigma_t^2, \rho_t)). \quad \ldots (1)$$

In the above regression, the covariate vector $\underline{x}_{ij}$ contains: (i) VOMs encapsulating land use information, the effects of which are allowed to vary from period to period, (ii) the type of nesting structure (i.e., single- vs. double-nest cylinder), the effect of which is also allowed to vary from period to period, and (iii) linear and squared terms for the size of open area in wetlands in which the structure was deployed. In addition, the model also included temporal effects, so that the outcome was allowed to vary from year to year, and from period to period within each year.

θ and λ in model (1) respectively indexed the spatial and temporal components of the process Y. Specifically, these random effects accounted for possible spatial and temporal correlations (ρ_s and ρ_t respectively) that were likely to be present. For our data, we conducted exploratory analyses to choose suitable structures for the spatial ($\mathbf{V}_s$) and temporal ($\mathbf{V}_t$) covariances. Since we had only 3 time points (i.e., 3 years) to model the pattern of temporal variation, we assumed a simple exchangeable correlation structure for $\mathbf{V}_t$ with 4 parameters (Diggle et al. 1994). Thus, the $(j,k)^{\text{th}}$ element of this covariance matrix is given by:

$$((\mathbf{V}_t))_{jk} = \text{cov}(Y_{ij}, Y_{ik}) = \sigma_{ti}^2 \rho_t, \forall\ j \neq k, i = 1,2,3.$$

Graphical semi-variogram analysis (Kaluzny et al. 1997:101-107, Cressie 1993) revealed that a Gaussian spatial covariance structure with 2 parameters provided an appropriate and parsimonious fit for these data. Thus, the $(i,i')^{\text{th}}$ element of this covariance matrix was given by:

$$((\mathbf{V}_s))_{ii'} = \text{cov}(Y_{ij}, Y_{i'j}) = \sigma_s^2 \exp\left(-\frac{d_{ii'}^2}{\rho_s^2}\right), (i,i') = 1,2,3, i \neq i',$$

where $d_{ii'}$ is the Euclidean distance between structures i and i'. Our hierarchical model 1 belongs to the class of generalized linear mixed models (GLMM) (Breslow and Clayton 1993) for which model parameters can be estimated using different approaches (Clayton and Kaldor 1987, Breslow and Clayton 1993, Yasui and Lele 1997, McCulloch 1997, Diggle et al. 1998). We chose the penalized quasi-likelihood approach (Breslow and Clayton 1993) because it has been widely used and has the important advantage of being implemented in the SAS GLIMMIX macro program (Wolfinger and O'Connell 1993). GLIMMIX makes the fitting of a wide range of covariance structures to correlated categorical data relatively straightforward.

We calculated an aggregate VOM for each nesting structure each year for each of the 4 time periods by creating buffers having radii of 0.8, 1.6, 2.4, and 3.2 km around each structure. GIS was used to determine the amount of the area within the buffer that corresponded to each land use category. For each land use polygon in a buffer, we calculated a mean VOM for the time period. The aggregate VOM was the mean VOM of all land use categories weighted by the area of each land use polygon in the buffer. In order to identify the buffer size surrounding the nesting structure that had the most influence on structure occupancy, we fit the model 4 times. The aggregate VOMs for the buffer and the number of nearby structures included in the buffer changed in 4 models, while the values of other covariates did not. We used deviance function, a traditional likelihood based measure of goodness of model fit (McCullagh and Nelder 1989), as well as prediction-based performance measures for binary outcomes, such as sensitivity (i.e., percentage of nesting events accurately classified by the fitted model) and specificity (i.e., percentage of non-nesting events accurately classified by the fitted model) to choose the best model (Agresti 1990).

RESULTS

Each year from 47 – 54 mallard nest initiations occurred that could be used in the analysis (Table 2). A greater proportion of double-cylinder than single-cylinder structures was available for nest initiations in most time periods because the second nest-cylinder was often available for a nest initiation even though a nest might have been active in the first cylinder. GIS modeling indicated each structure was accessible to 61 – 274 mallard pairs (Fig. 3), and nest structures were located in open-water areas of 0 – 27.4 ha (Fig. 4). The number of nearby nest structures (Fig. 5) and aggregate VOMs (Fig. 6) during each time period depended on the size of the buffer around each structure.

Buffer Size

Results from fitting the full model using the data from the 4 buffer sizes produced similar results in terms of goodness of fit (deviance), predictive ability (sensitivity and specificity), and amount of reduction in spatio-temporal correlation (Table 3). Based on the deviance, which is a widely used measure of goodness of fit summarizing unexplained variability, the model based on a buffer with a 1.6 km radius seemed to perform the best and was chosen for inference testing.

Final Model

There were strong temporal trends in the nest initiation data. The final reduced model for the 1.6 km radius buffer included an aggregate VOM by period interaction ($F_{4,\,1012} = 3.97$, $P = 0.003$), a quadratic effect for the size of the open-water area around a structure ($F_{1,\,1012} = 8.42$, $P = 0.004$), and a year by period interaction ($F_{8,\,1012} = 3.03$, $P = 0.002$). There was also a marginally significant nest structure type by period interaction ($F_{4,\,1012} = 2.28$, $P = 0.059$). Lastly, neither pairs with access to a structure ($P = 0.7$), nor the number of nearby structures ($P = 0.8$) were significantly associated with a nest initiation.

Estimated regression coefficients from the final reduced model indicated that aggregate VOMs were positively associated with the probability of nest occupancy in the earlier part of the nesting season, but that this pattern was reversed towards the end of the nesting season (Fig. 7). Thus, in the 1st period, the odds of a nest structure being occupied increased ~12 times for each dm increase in the aggregate VOM around a nest structure, whereas by the 4th period those odds decreased by a factor of >1.5 for each dm increase in the aggregate VOM. The size of open-water area in adjacent wetlands was also significantly associated with nest initiations; for median sized open-water areas (0.63 ha), each hectare increase in size increased the odds of a nest structure being occupied by a factor of 1.3. However, this effect reached an asymptote for open-water areas of 16 ha or larger (Fig. 8). A double-cylinder nest structure also was ~4 times more likely to have a nest initiated in the 3rd period than was a single-cylinder structure, although the effect was only marginally significant. Lastly, the year by period interaction reflected differences in nest initiation phenology where more nests were initiated earlier in the season in 1998 than in the other years.

DISCUSSION

The probability of nest structure occupancy was related significantly to characteristics of the landscape. However, our analyses indicated that the relationship was complex. Some of our *a priori* predictions were supported while others were not. We had expected that structure use would be related to the number of pairs with access to the structure, but this was not the case in our landscape. It is possible that pair numbers were correlated with other landscape variables that we included in the model; thus, pairs were excluded from the final model because they provide information that was in effect redundant. Alternatively, our pair measure was model-based and might not have reflected actual pair numbers. Likewise, we detected no relationship between the probability that a structure would be used and the number of nearby nest structures in the buffer having a 1 km radius, suggesting that competition for nesting hens by nearby structures was low. Zicus et al. (2003) reported that specific nest structures in a landscape had clutches that were the product of intraspecific nest parasitism more often than other nearby nest structures. Possibly, mallard hen nest site selection is influenced by behavior in addition to the effects of the landscape features that we measured.

Cover height and density had a significant effect on the probability that a nest structure would be occupied, but the effect changed as the season progressed. Structures in buffers that had the highest aggregate VOM early in the season were those most likely to be used. In contrast, later in the nesting season the effect was reversed and probability of use was greatest when the VOMs in the surroundings were the lowest. We suspect that the difference between the cover effect in the earliest period and that in the later periods might be related to the initial settling pattern of mallard pairs returning in the spring. Perhaps significantly, aggregate VOMs were low throughout the entire landscape during the first period of the nesting season when mallards first initiated nests. Structure occupancy also was influenced by the size of the open water area in which the structure was erected. Occupancy increased with increasing area of open water up to ~15 ha.

Ultimately, we would like to develop a model whereby GIS data can be used to predict the probability of nest structure occupancy in different wetland locations. If we can combine this model with nest structure cost data (M. Zicus, unpubl. data), then managers would have a tool that would enable them to deploy nest structures in the landscape in a way that would maximize the benefit:cost ratio.

LITERATURE CITED

Agresti, A. 1990. Categorical data analysis. John Wiley, New York, New York, USA.

Besag, J. E. 1974. Spatial interaction and the statistical analysis of lattice systems. Journal of the Royal Statistical Society Series B 35:192-236.

Breslow, N. E., and D. G. Clayton. 1993. Approximate inference in generalized linear mixed models. Journal of the American Statistical Association 88:9-25.

Clayton, D., and J. Kaldor. 1987. Empirical Bayes estimates of age-standardized relative risks for use in disease mapping. Biometrics 43:671-681.

Cowardin, L. M., D. S. Gilmer, and C. W. Shaifer. 1985. Mallard recruitment in the agricultural environment of North Dakota. Wildlife Monographs 92:1-37.

Cowardin, L. M., D. H. Johnson, T. L. Shaffer, and D. W. Sparling. 1988. Applications of a simulation model to decisions in mallard management. U.S. Fish and Wildlife Service, Fish and Wildlife Technical Report 17. 28pp.

Duebbert, H. F., and H. A. Kantrud. 1974. Upland duck nesting related to land use and predator reduction. Journal of Wildlife Management 38:257-265.

Cressie, N., and N. H. Chan. 1989. Spatial modeling of regional variables. Journal of the American Statistical Association 84:393-401.

Cressie, N. 1993. Statistics for Spatial Data. John Wiley, New York, New York, USA.

Das, A., S. R. Lele, G. E. G. Glass, T. Shields, and J. A. Patz. 2002. Modeling a discrete spatial response using generalized linear mixed models: application to Lyme disease vectors. International Journal of Geographic Information Science 16:151-166.

Dempster, A. P., N. M. Laird, D. B. and Rubin. 1977. Maximum likelihood for incomplete data via the EM algorithm (with discussion). Journal of the Royal Statistical Society, Series B 44:1-38.

Diggle, P. J., K. Y. Liang, and S. L. Zeger. 1994. Analysis of Longitudinal Data. Oxford University Press, London, U.K.

Diggle, P. J., J. A. Tawn, and R. A. Moyeed. 1998. Model-based geostatistics (with discussion). Journal of the Royal Statistical Society Series C 47:299-350.

Duebbert, H. F., and H. A. Kantrud. 1974. Upland duck nesting related to land use and predator reduction. Journal of Wildlife Management 38:257-265.

Environmental Systems Research Institute, Inc. 1996. ArcView GIS: the geographic information system for everyone. Environmental Systems Research Institute Inc., Redlands, CA, USA.

Johnson, D. H., D. W. Sparling, and L. M. Cowardin. 1987. A model of the productivity of the mallard duck. Ecological Modelling 38:257-275.

Kaluzny, S., S. Vega, T. Cardoso, and A. Shelly. 1997. S+ spatialstats user's manual. Springer-Verlag, Berlin, Germany.

Klett, A. T., H. F. Deubbert, C. A. Faanes, and K. F. Higgins. 1986. Techniques for studying nest success on ducks in upland habitats in the prairie pothole region. U. S. Fish and Wildlife Service, Resource Publication 158.

McCullagh, P., and J. Nelder. 1989. Generalized linear models. Chapman and Hall, London, U.K.

McCulloch, C. E. 1997. Maximum likelihood algorithms for generalized linear mixed models. Journal of the American Statistical Association 92:62-170.

Prince, H. 1997. Wetlands of the American Midwest: a historical geography of changing attitudes. University of Chicago Press, Chicago, Illinois, USA.

Reynolds, R. E., D. R. Cohan, and M. A. Johnson. 1996. Using landscape information approaches to increase duck recruitment in the prairie pothole region. Transactions North American Wildlife and Natural Resources Conference 61:86-93.

Robel, R. J., J. N. Briggs, A. D. Dayton, and L. C. Hulbert. 1970. Relationships between visual obstruction measurements and weight of grassland vegetation. Journal of Range Management 23:295-297.

Ver Hoef, J., and N. Cressie. 2001. Spatial statistics: analysis of field experiments. *In:* S. Scheiner and J. Gurevitch (eds.) Design and analysis of ecological experiments (2nd Edition). Chapman and Hall, New York, New York, USA.

Wolfinger, R., and M. O'Connell. 1993. Generalized linear mixed models: a pseudo-likelihood approach. Journal of Statistical Computation and Simulation 48:233-243.

Yasui, Y., and S. Lele. 1997. A regression method for spatial disease rates: an estimating function approach. Journal of the American Statistical Association 92:21-32.

Zicus, M. C., and D. P. Rave. 1998. Assessment of a mallard model in Minnesota's Prairie Coteau. Minnesota Wildlife Report 12.

Zicus, M. C., J. S. Fieberg, and D. P. Rave. 2003. Does mallard clutch size vary with landscape composition: a different view. Wilson Bulletin. In Press.

Table 1. Land use composition (%) of the study area, 1997 – 1999.

Land use	1997	1998	1999
Row crops	54.5	55.8	60.9
Small grain	18.9	16.6	11.8
Wetland/open water	10.3	11.2	12.3
Conservation Reserve Program[a]	4.4	3.3	3.6
Odd areas[b]	3.9	4.0	3.7
Wildlife areas[c]	1.8	1.9	1.8
Hay	1.5	2.0	1.5
Transportation right-of-ways	1.4	1.5	1.4
Other grass	1.1	1.6	1.0
Barren	1.1	1.2	1.1
Woodland	0.6	0.6	0.6
Pasture	0.3	0.4	0.3

[a]Includes Conservation Reserve Program, Reinvest in Minnesota, and Waterbank Program land.
[b]Patches of nesting cover (<2 ha) described by other land use types, and linear and point features (e.g., rockpiles in cropland or strips of grass cover between plowed fields).
[c]Includes U.S. Fish and Wildlife Service Waterfowl Production Areas and Minnesota Department of Natural Resources Wildlife Management Areas.

Table 2. Mallard nest initiations by period of the nesting season for 53 single- and 57 double-cylinder nest structures in western Minnesota, 1997-1999.

		1997 Nest type[a]		1998 Nest type		199 Nest type	
Period	Initiation	Single-	Double-	Single-	Double-	Single-	Double-
March 15 - April 20	No	46	51	38	42	43	44
	Yes	7	6	14	15	10	13
	NA[b]	0	0	1	0	0	0
April 21 - April 30	No	38	47	33	52	37	46
	Yes	8	10	4	4	5	10
	NA	7	0	16	1	11	1
May 1 - May 20	No	33	45	33	51	38	47
	Yes	2	11	0	5	0	9
	NA	18	1	20	1	15	1
May 21 - June 30	No	39	50	44	54	45	49
	Yes	4	5	2	3	2	5
	NA	10	2	7	0	6	3

[a]Nest structures with either 1 or 2 nest cylinders on a single pole.
[b]Nest structure occupied by a nesting mallard or wood (*Aix sponsa*) duck(s) and thus not available for a nest initiation during this period.

Table 3. Model performance based on buffer sizes of 0.8, 1.6, 2.4, and 3.2 km around each of the 110 nest structures in the western Minnesota study area, 1997 – 1999.

Buffer radius (km)	Temporal Correlation	Spatial Correlation	Deviance	Sensitivity (%)	Specificity (%)
0.8	0.56[a]	0	557.48	80.4	84.6
1.6	0.55[a]	0	554.07	81.8	85.1
2.4	0.57[a]	0	555.42	82.4	84.5
3.2	0.57[a]	0	558.35	80.4	84.4

[a]Correlation statistically significant (*P*<0.0001).

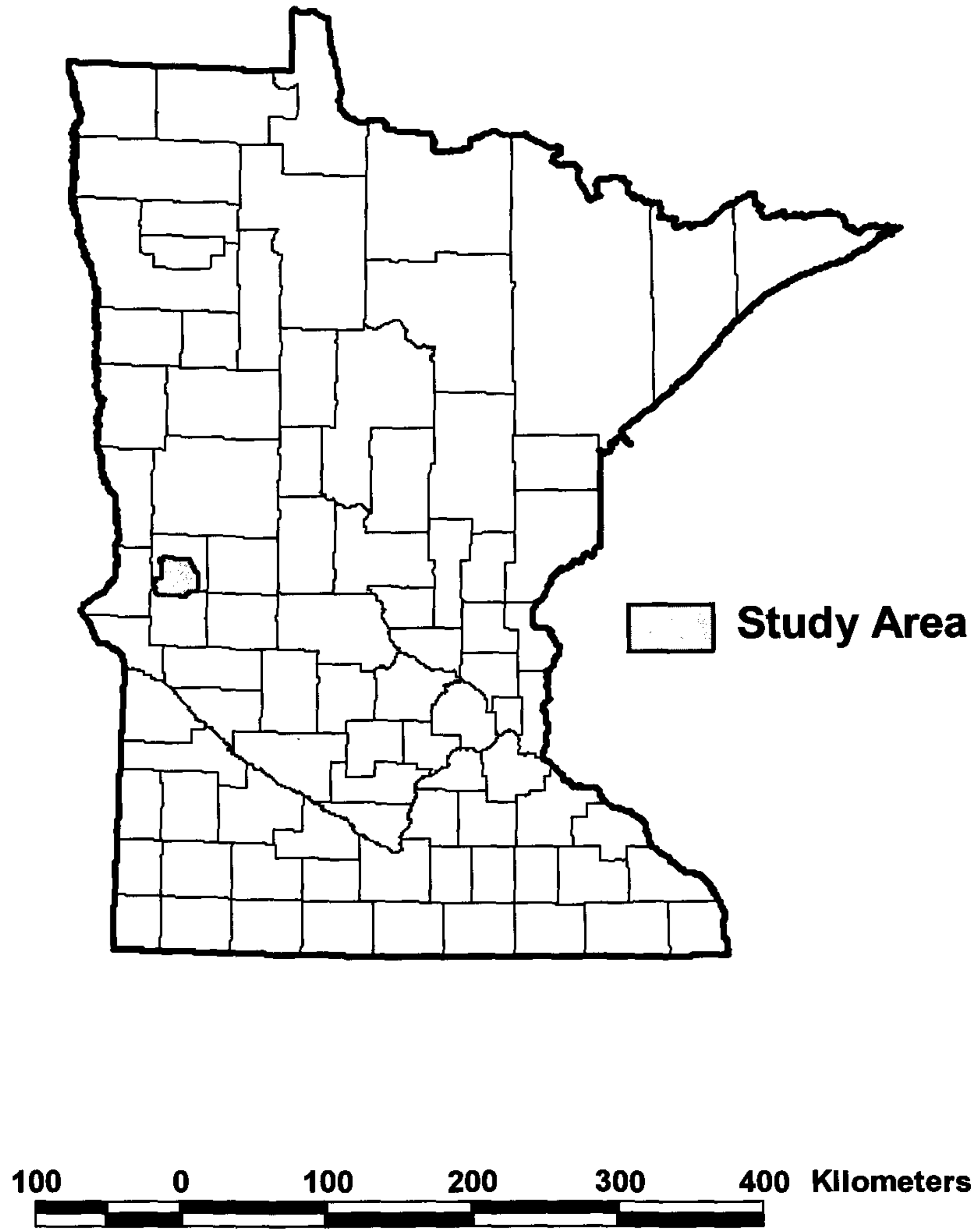


Figure 1. Nest structure study area in Grant and Stevens counties, Minnesota, 1997-1999.

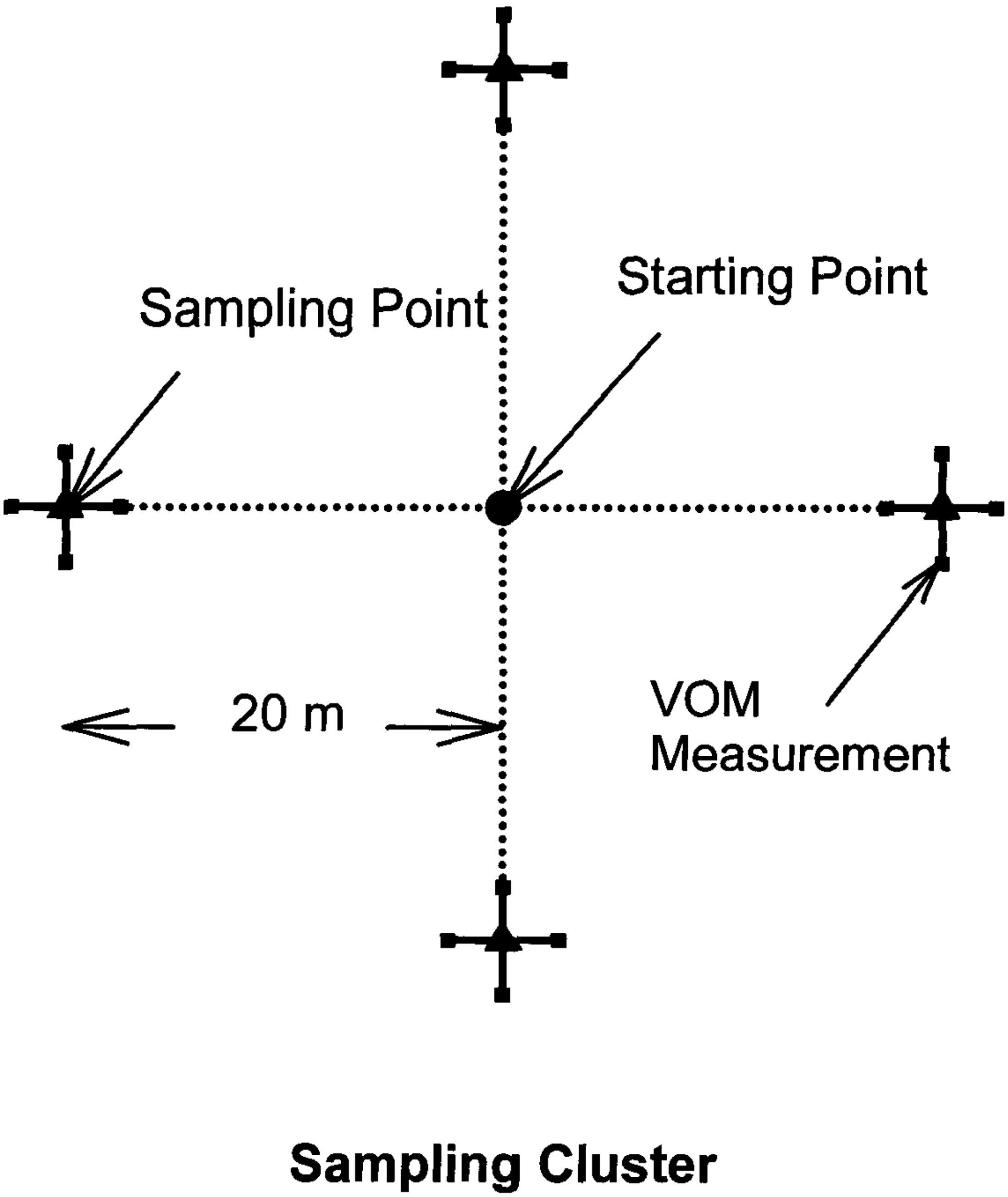


Figure 2. Sampling cluster used to estimate visual obstruction measurements in nest structure study in Grant and Stevens counties, Minnesota, 1997-1999.

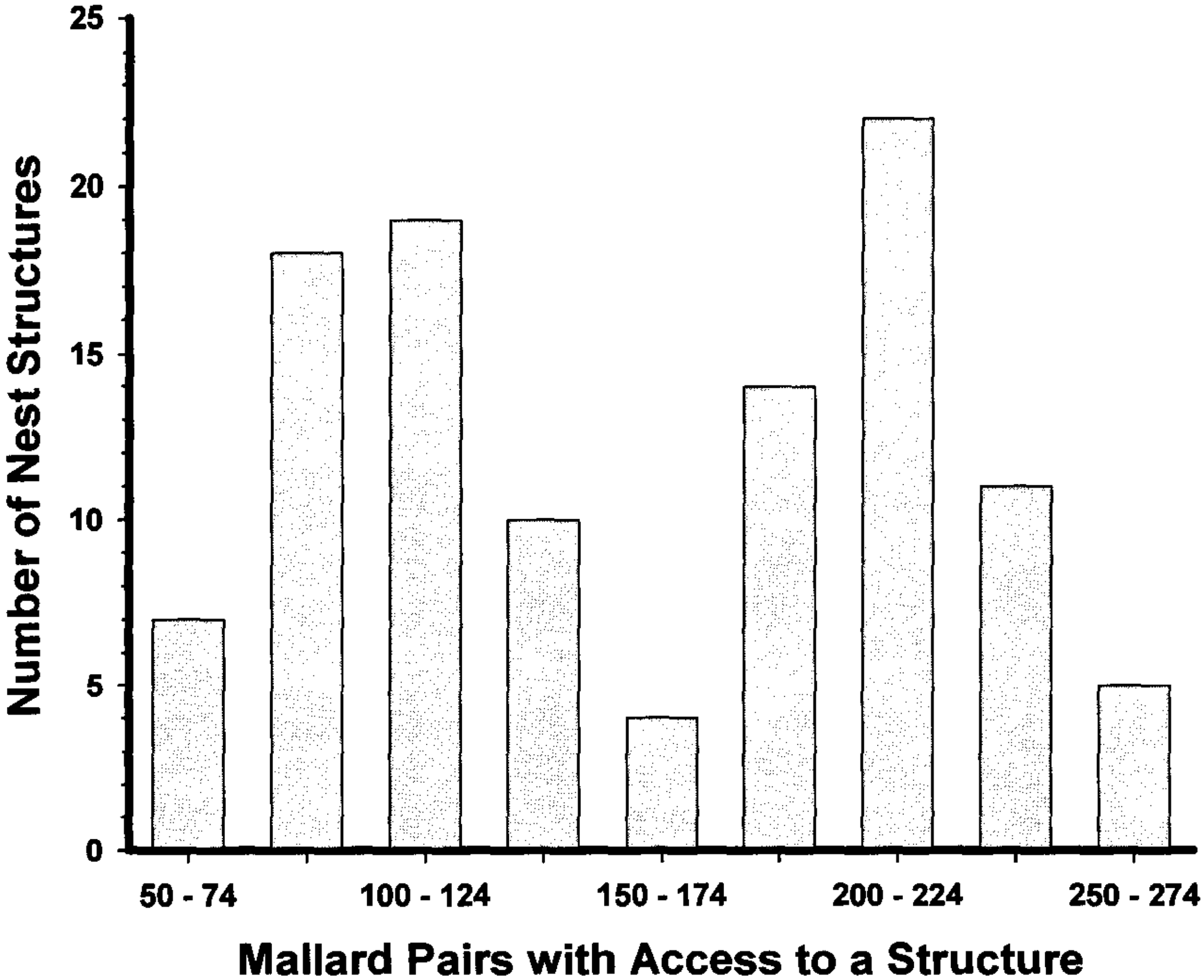


Figure 3. Number of mallard pairs with access to nest structures in Grant and Stevens counties, Minnesota, 1997-1999.

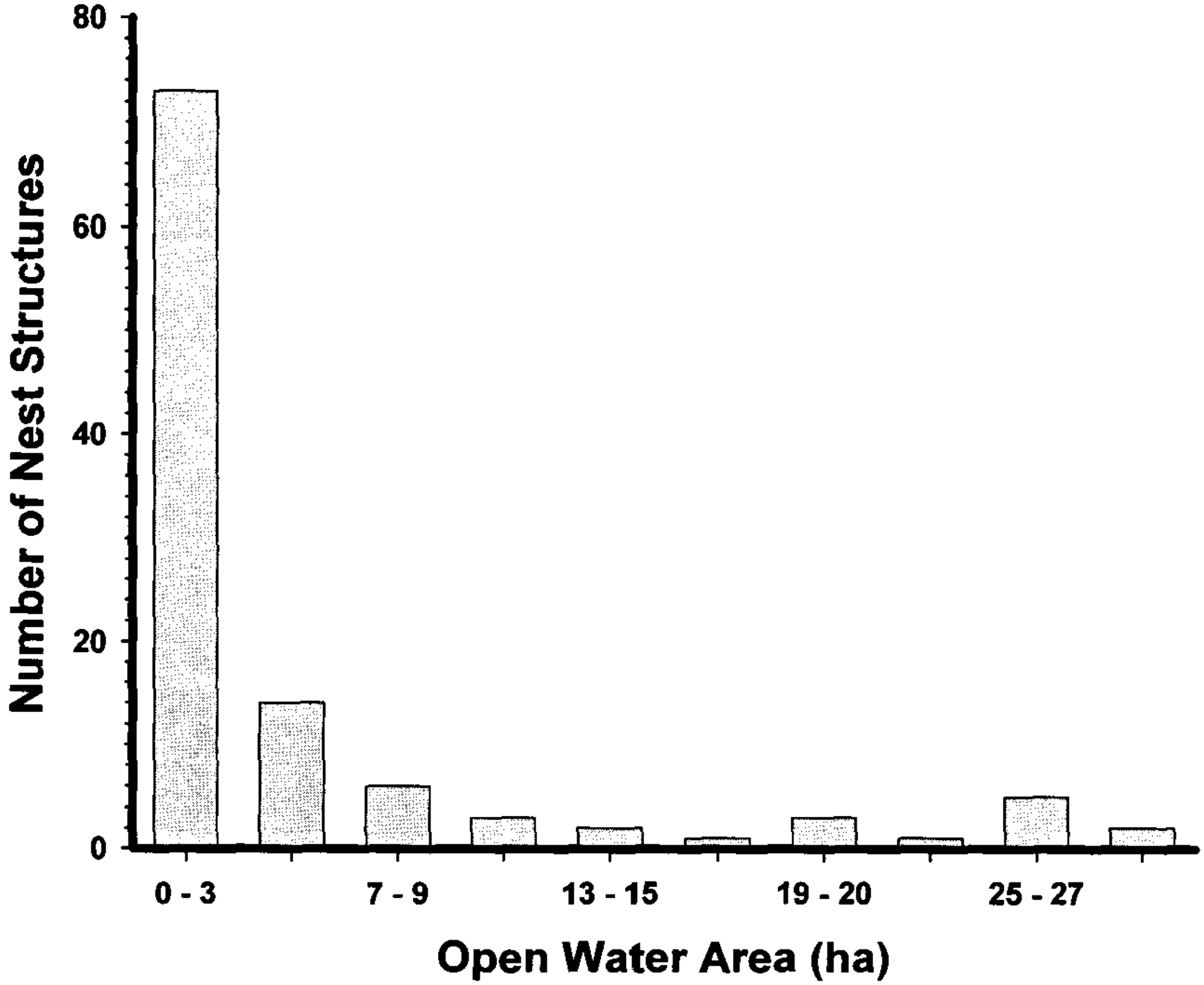


Figure 4. Size of the open-water area associated with nest structures in Grant and Stevens counties, Minnesota, 1997-1999.

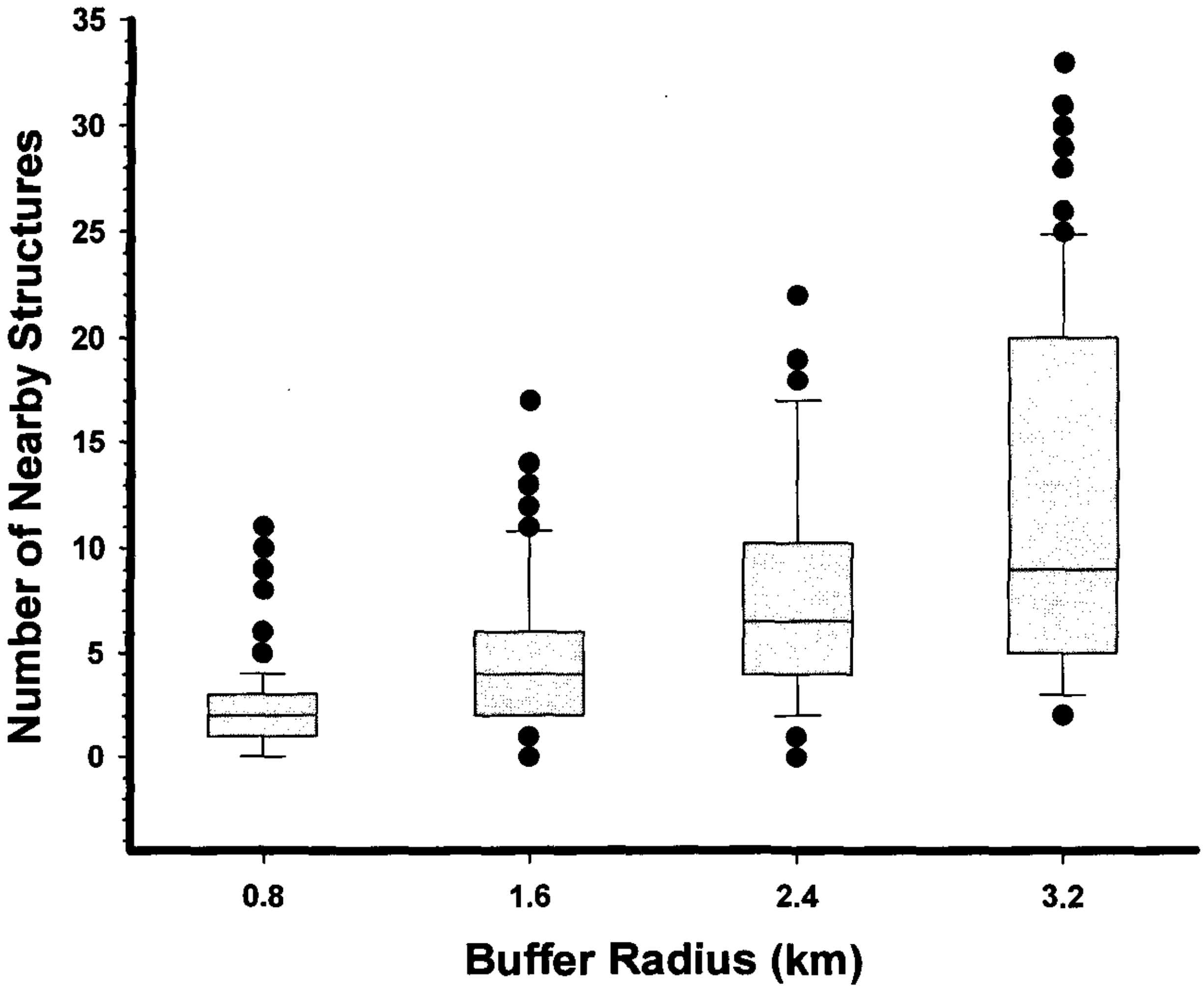


Figure 5. Number of nearby nest structures associated with each nest structure in 4 different sized buffers in Grant and Stevens counties, Minnesota, 1997-1999.

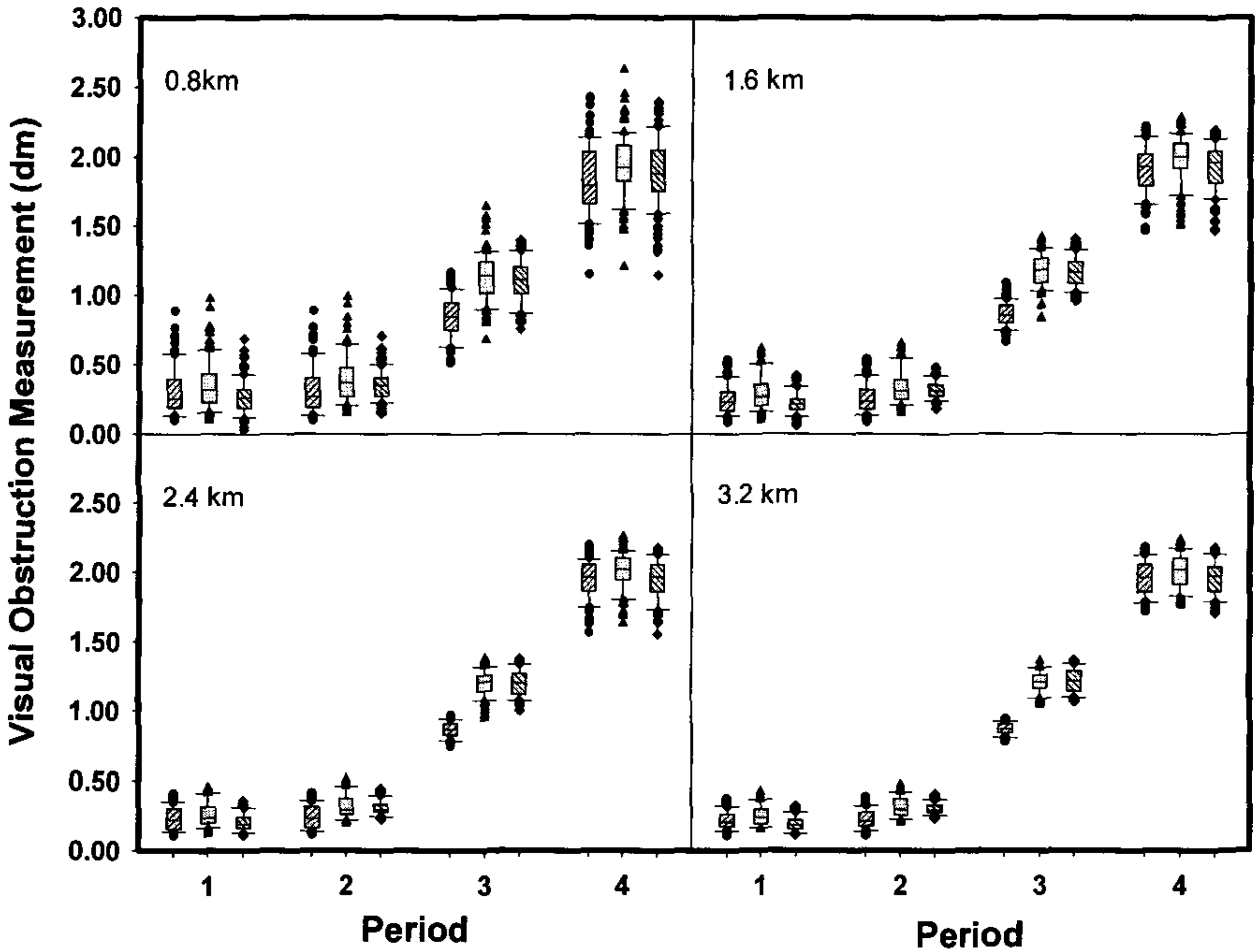


Figure 6. Distribution of aggregate visual obstruction measurements by period and buffer size in Grant and Stevens counties, Minnesota, 1997-1999. Right-hand diagonal fill and circular symbols = 1997. Plain fill and triangular symbols = 1998. Left-hand diagonal fill and diamond symbols = 1999.

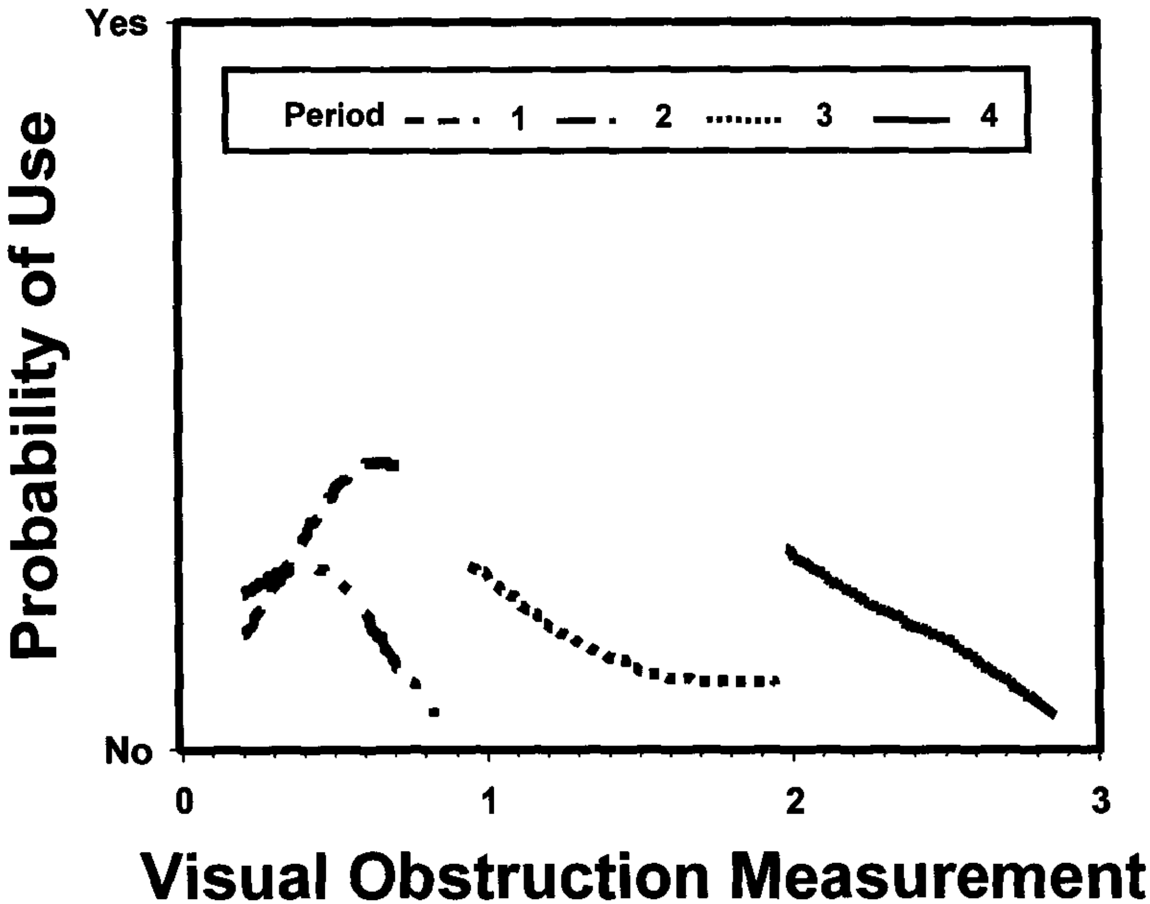


Figure 7. Spline curves illustrating the effect of the aggregate visual obstruction measurement on the probability of nest structure use during the 4 different time periods in Grant and Stevens counties, Minnesota, 1997-1999.

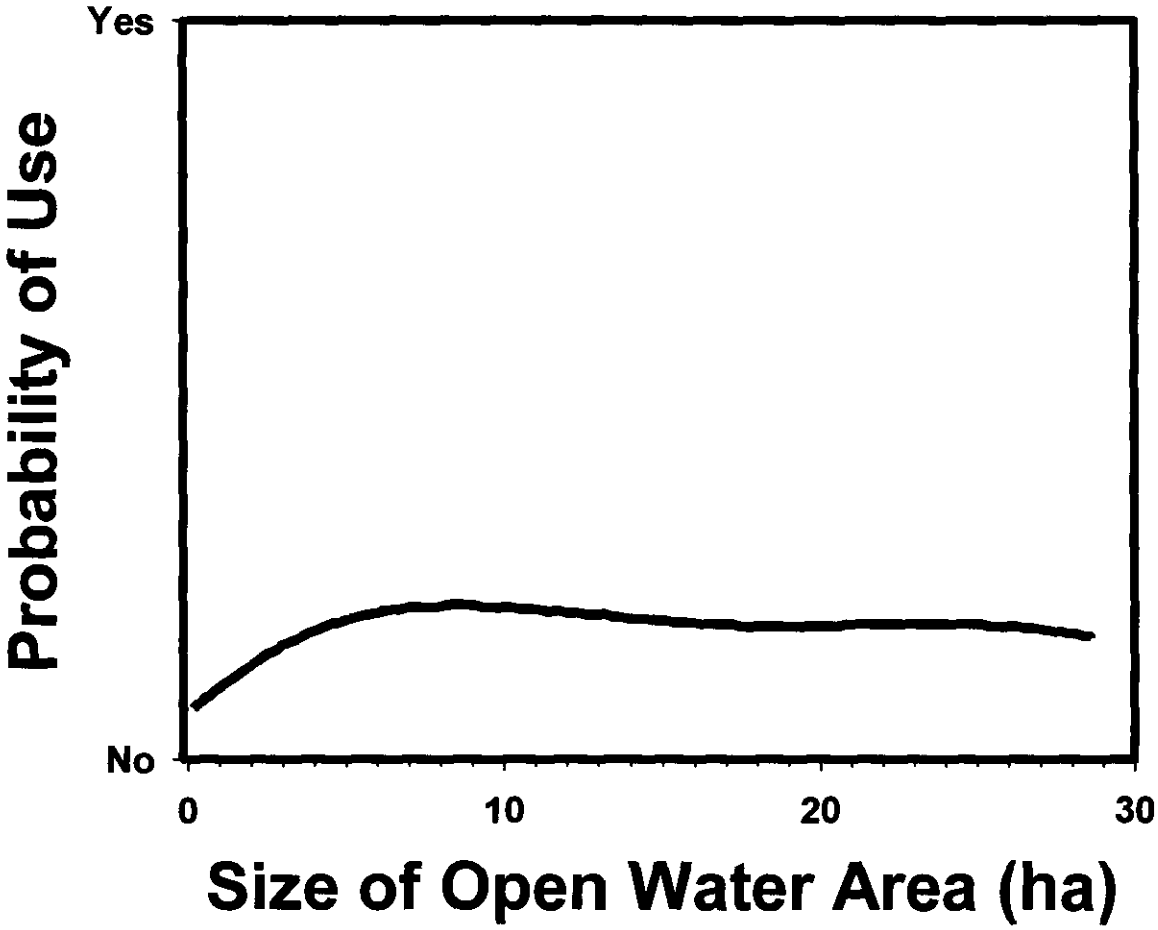


Figure 8. Spline curves illustrating the effect of the size of the open water area on the probability of nest structure use in Grant and Stevens counties, Minnesota, 1997-1999.

BEMIDJI AREA BREEDING PAIR SURVEY – UPDATE 2003

Michael C. Zicus and David P. Rave

ABSTRACT

A breeding duck pair survey of 6 lakes in the Bemidji vicinity that had been conducted from 1959 – 1990 was reinstated in spring 2003. Methods followed those used in the early survey period when lakes were surveyed in mid-May and again in early June. Mallard (*Anas platyrhynchos*), blue-winged teal (*A. discors*), and wood ducks (*Aix* sponsa) showed no significant trends in the number of pairs counted during 1959 – 1990. In contrast, pairs of common goldeneyes (*Bucephala clangula*), American wigeon (*Anas americana*), and ring-necked ducks (*Aythya collaris)* appeared to be declining significantly. In 2003, numbers of pairs of most species were similar to those counted in the later years of the earlier surveys. Common goldeneyes were a notable exception with more pairs counted in 2003 than during any prior year.

INTRODUCTION

Staff in the Minnesota Department of Natural Resources Wetland Wildlife Populations and Research Group has been developing a forest waterfowl research initiative. To better understand the status of breeding waterfowl populations in the Laurentian mixed forest province of Minnesota, the decision was made to reinstate a breeding duck pair survey of 6 lakes in the Bemidji vicinity in spring 2003. This survey was conducted last in 1990. The results of this survey will then be compared with historic counts on these same lakes to begin to better understand the status of forest breeding waterfowl populations.

METHODS

Methods followed those used in the early survey period (L. Johnson, pers. commun.). The survey included Blackduck Lake, Gull Lake, Moose Lake, Dixon Lake, Gilstad Lake, and Medicine Lake (Fig. 1). From 1959-1976, an early breeding pair count was conducted during the second week of May to survey mallard (*Anas platyrhynchos*), wood duck (*Aix* sponsa), and common goldeneye (*Bucephala clangula*) pairs. A second survey during the last week of May or first week of June was directed at blue-winged teal (*Anas discors*), ring-necked duck (*Aythya collaris)*, and American wigeon (*A. americana*) pairs. From 1977 – 1990, only 1 breeding pair survey per year was conducted during the third week of May. Each lake count was conducted by 2 observers in a boat while slowly motoring around the lake perimeter (Appendix A), and data were recorded directly onto a data collection form. Areas of the lake with dense stands of wild rice or other vegetation were scanned with binoculars, and all waterfowl were recorded. Historically, 1 – 3 lakes were counted each day that weather permitted, such that the entire 6-lake survey was completed in 2-4 days.

All ducks, geese, and swans present were counted on each survey date. In addition to pairs comprised of a drake and a hen, lone drake, or each drake in a flock of 5 or less was considered to represent a pair. Drakes in flocks of 6 or more were considered migrants and were not expanded to represent pairs (L. Johnson, pers. commun.).

RESULTS

Fourteen species of waterfowl were seen during the survey from 1959 – 1990. Mallard, blue-winged teal and wood ducks on the 6 lakes showed no significant trends in the number of pairs during this period. In contrast, pairs of common goldeneyes, American wigeon, and ring-necked ducks appeared to be declining significantly.

In spring 2003, the 2 breeding pair surveys were completed from 12-13 May and from 2-5 June. Sixteen species of waterfowl were seen in 2003. Numbers of pairs of the most common species were similar to those counted in the later years of the earlier surveys (Figures 2 – 7). However, common goldeneyes were a notable exception with more pairs counted during the 2003 survey than during any year of the earlier survey.

DISCUSSION

Common goldeneye, ring-necked duck, and American wigeon are unique species within Minnesota's Laurentian mixed forest province. Data from the earlier years of this survey indicated that these species were declining on the survey lakes when the survey ended. As the Wetland Group considers future work in the Laurentian mixed forest province, it is clear that additional breeding waterfowl surveys will be needed before the status of forest breeding waterfowl can be established with any certainty. In the interim, this survey can provide data that can be compared to those collected beginning in 1959. As such, it will be useful to help define priority work regarding the local population status of several forest species.

ACKNOWLEDGMENTS

L. Johnson developed the survey initially and conducted it until 1959. Discussions with him were helpful in documenting the survey protocol so that the survey could be reinstated.

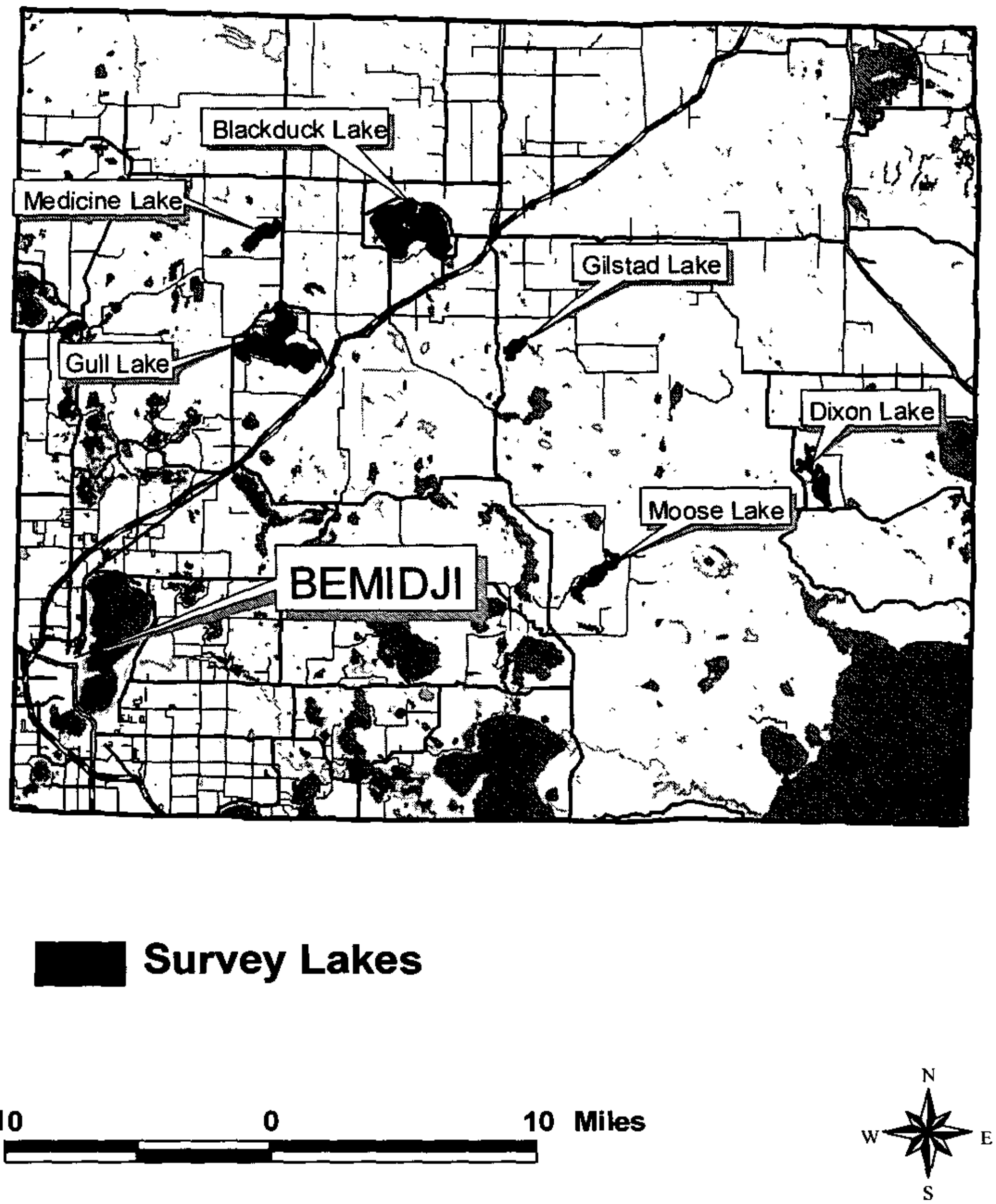


Figure 1. Location of lakes surveyed in the former Bemidji Area Pair Survey.

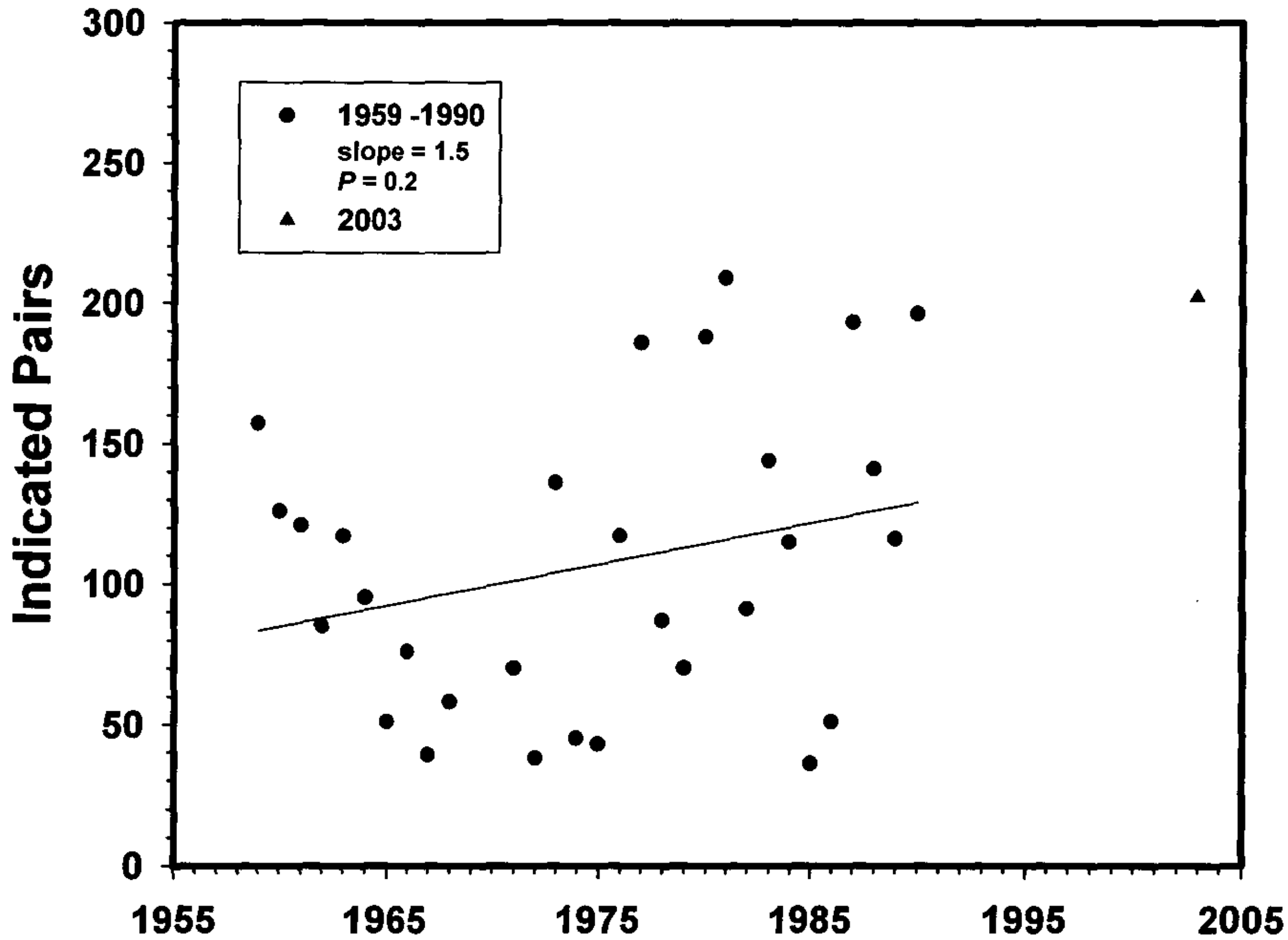


Figure 2. Indicated mallard pairs on Bemidji Area Pair Survey, 1959-2003.

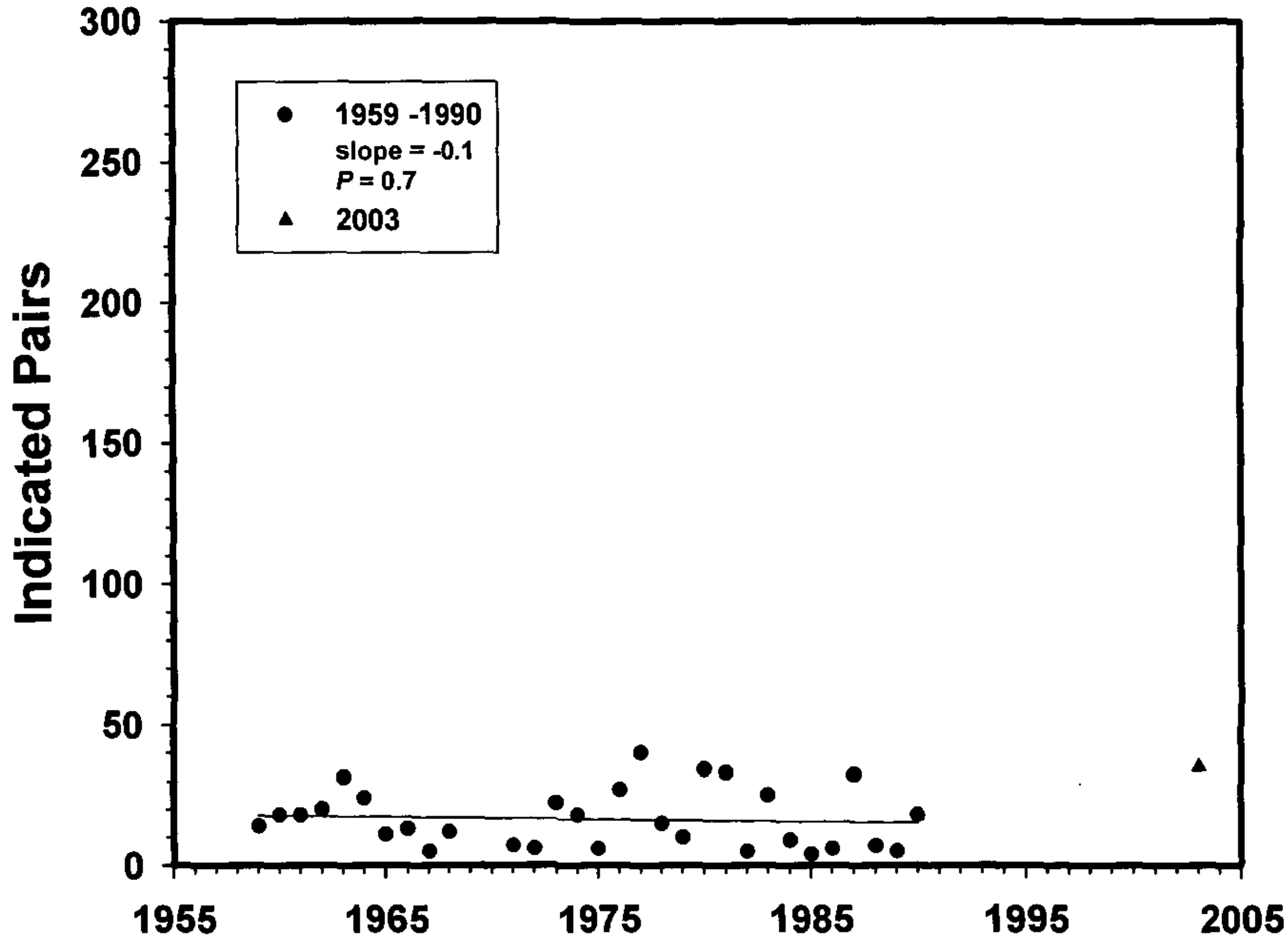


Figure 3. Indicated wood duck pairs on Bemidji Area Pair Survey, 1959-2003.

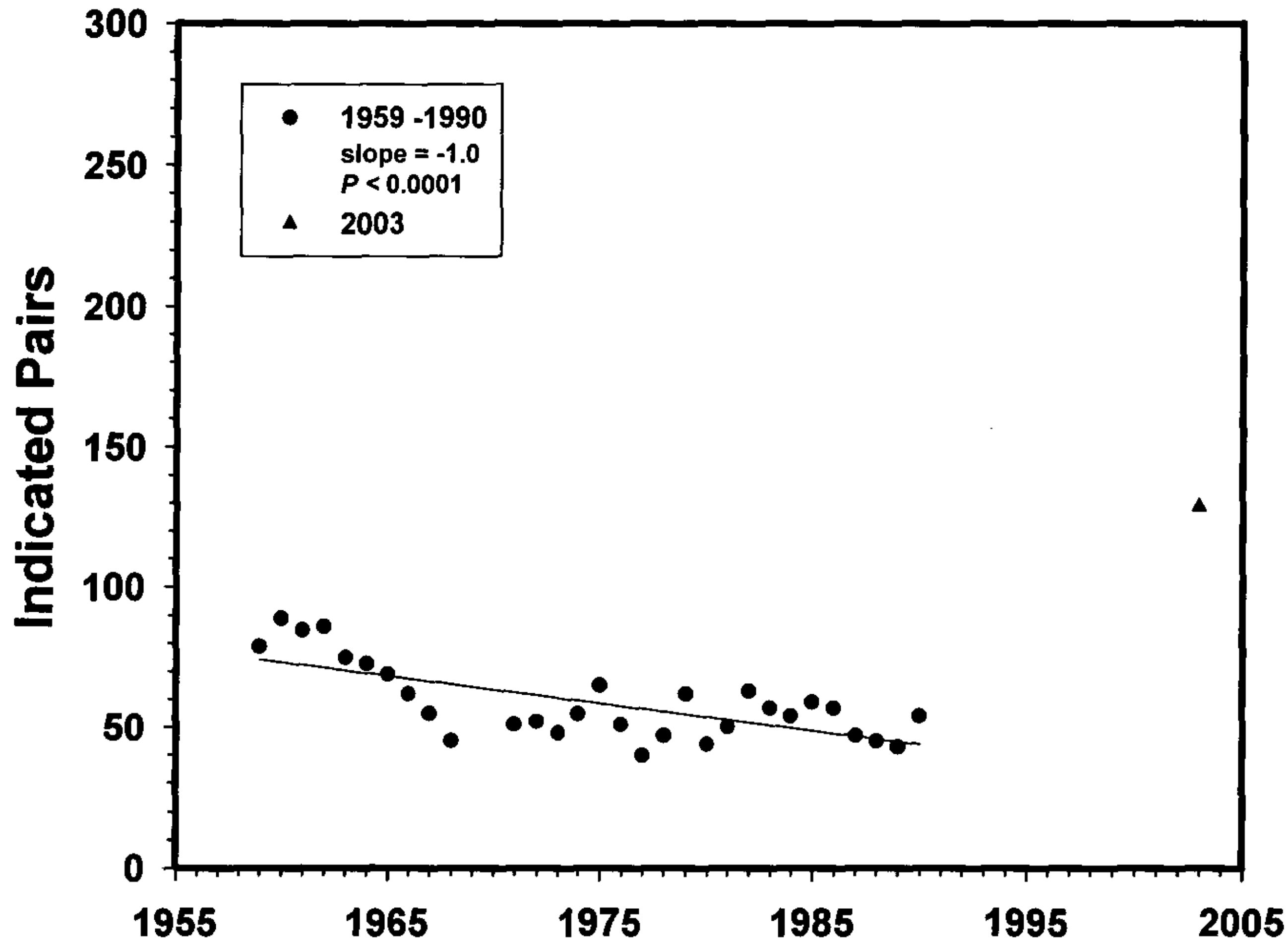


Figure 4. Indicated common goldeneye pairs on Bemidji Area Pair Survey, 1959-2003.

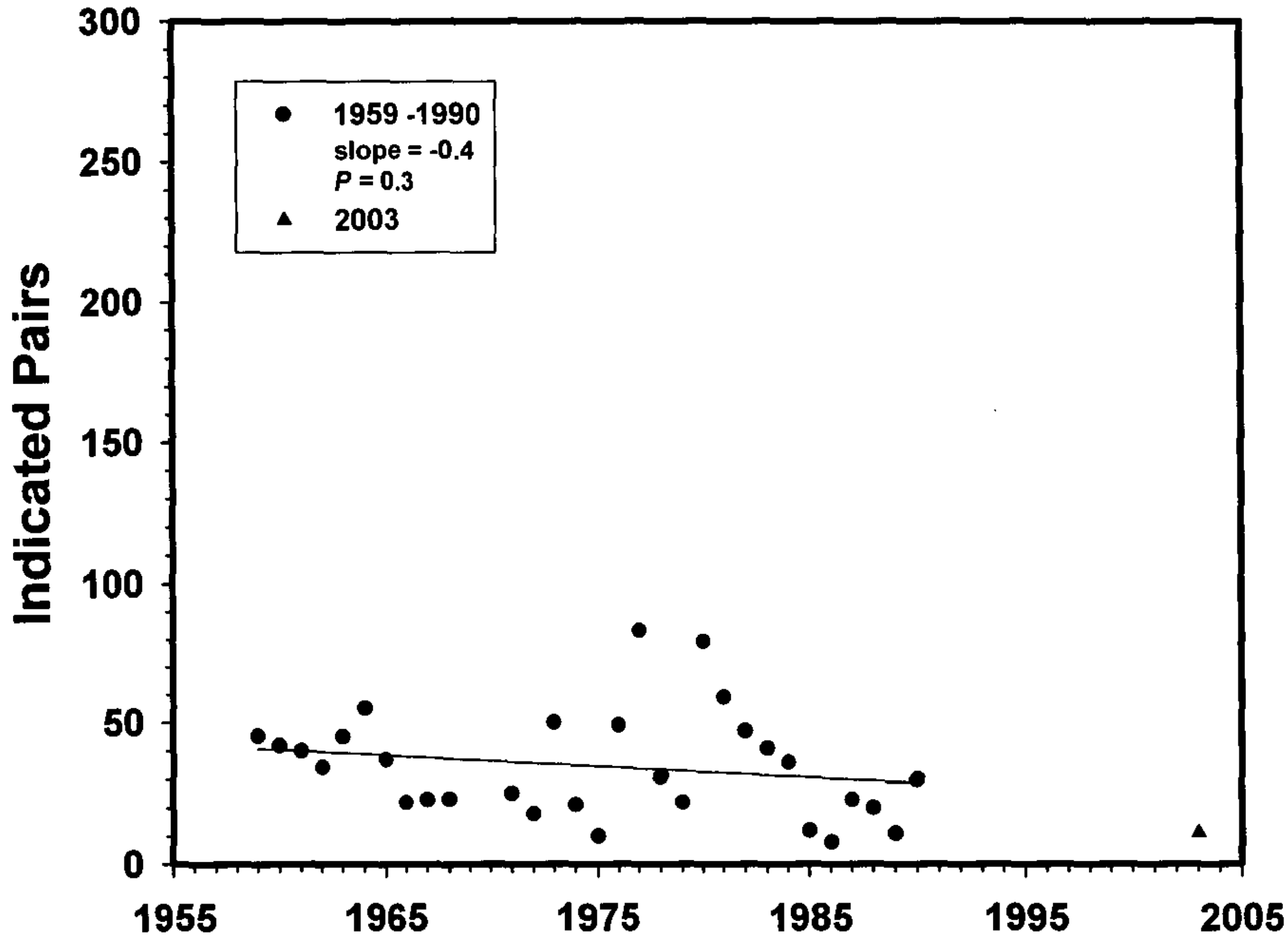


Figure 5. Indicated blue-winged teal pairs on Bemidji Area Pair Survey, 1959-2003.

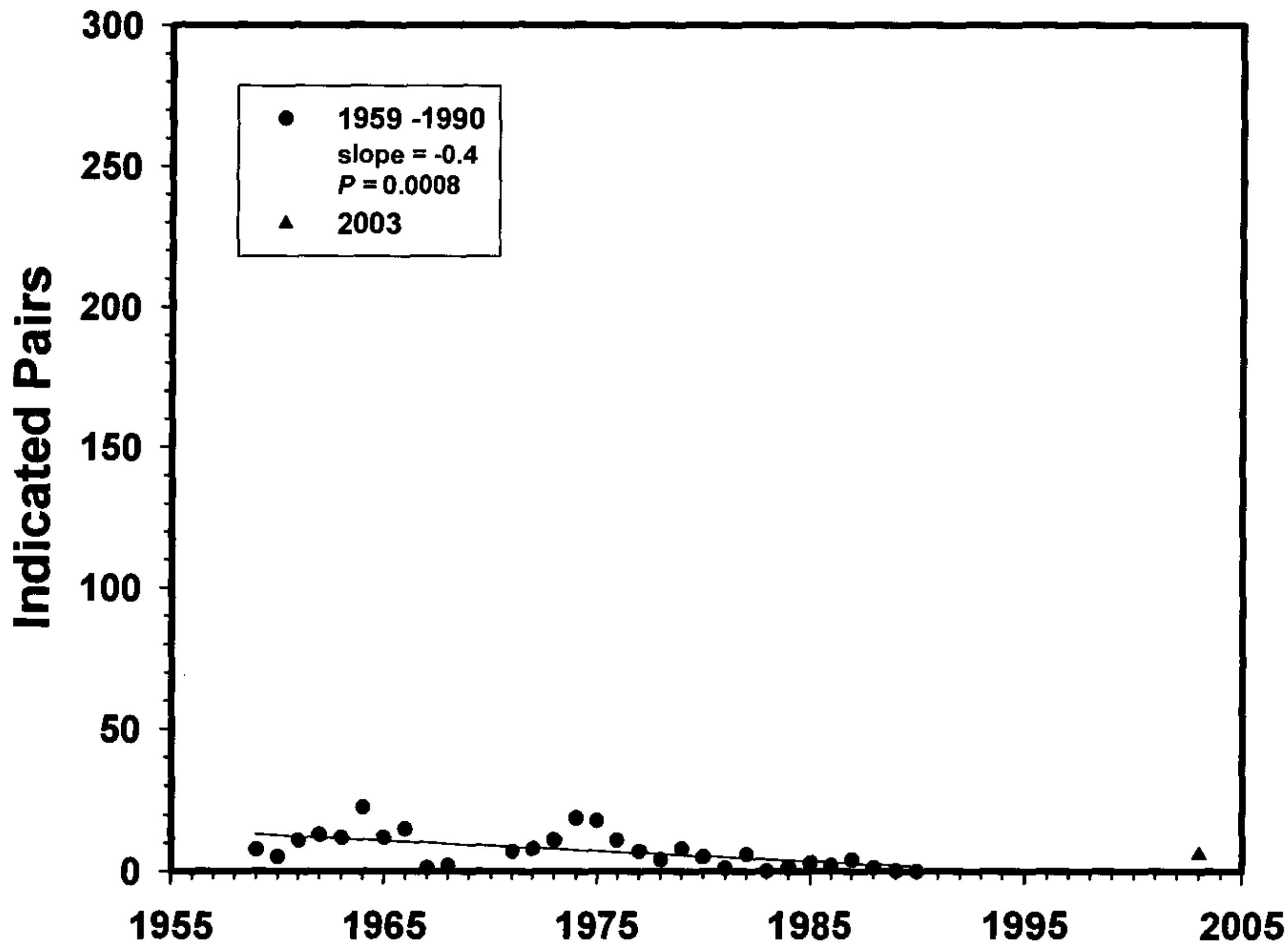


Figure 6. Indicated ring-necked duck pairs on Bemidji Area Pair Survey, 1959-2003.

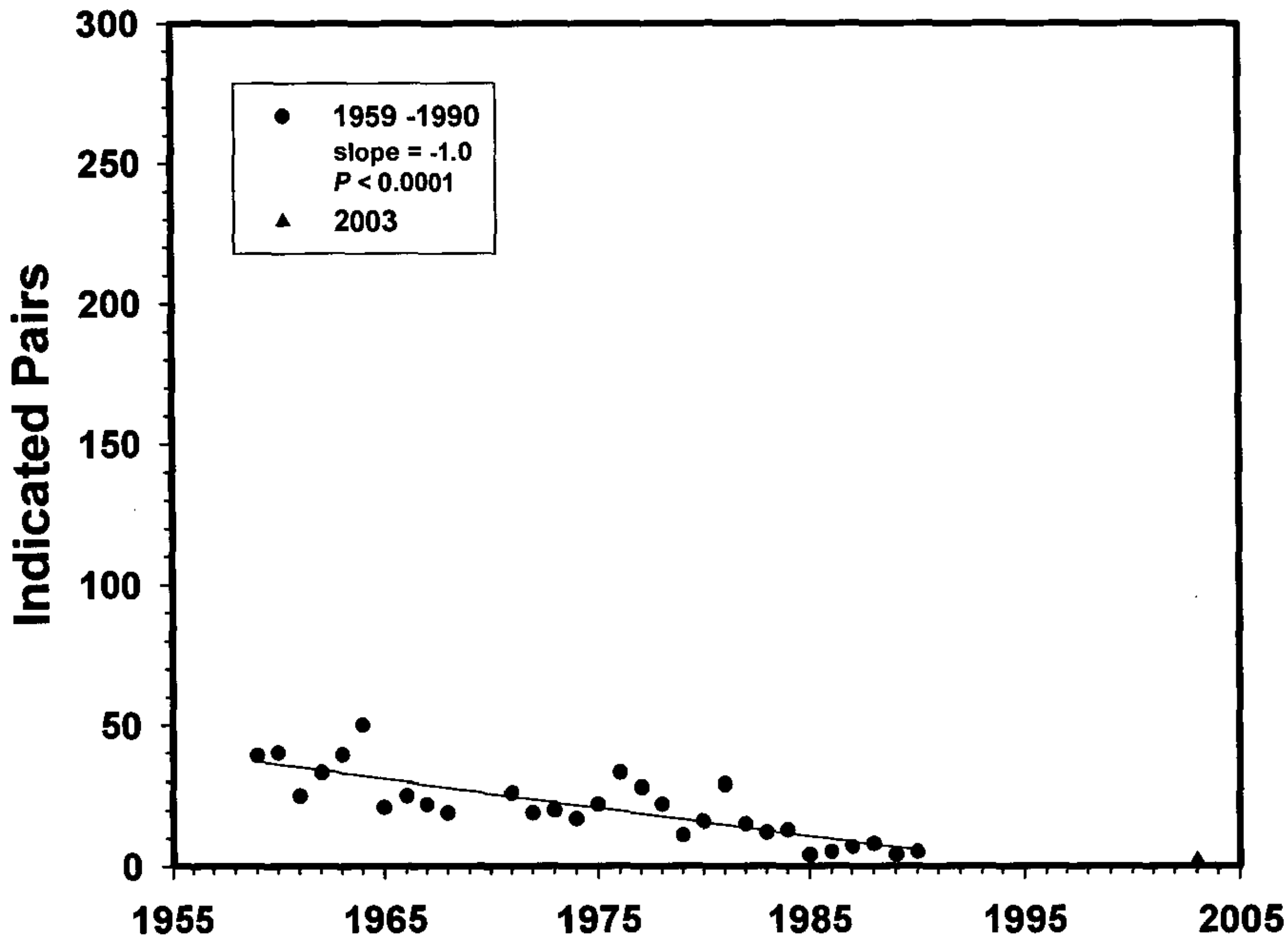
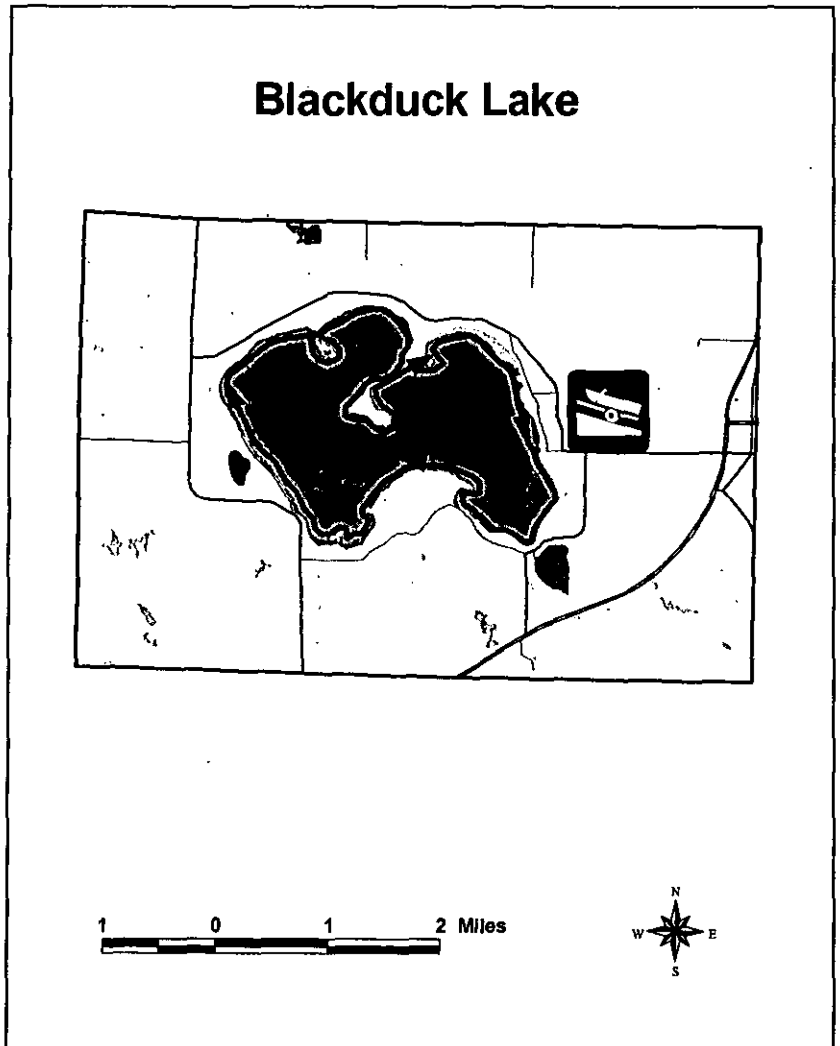
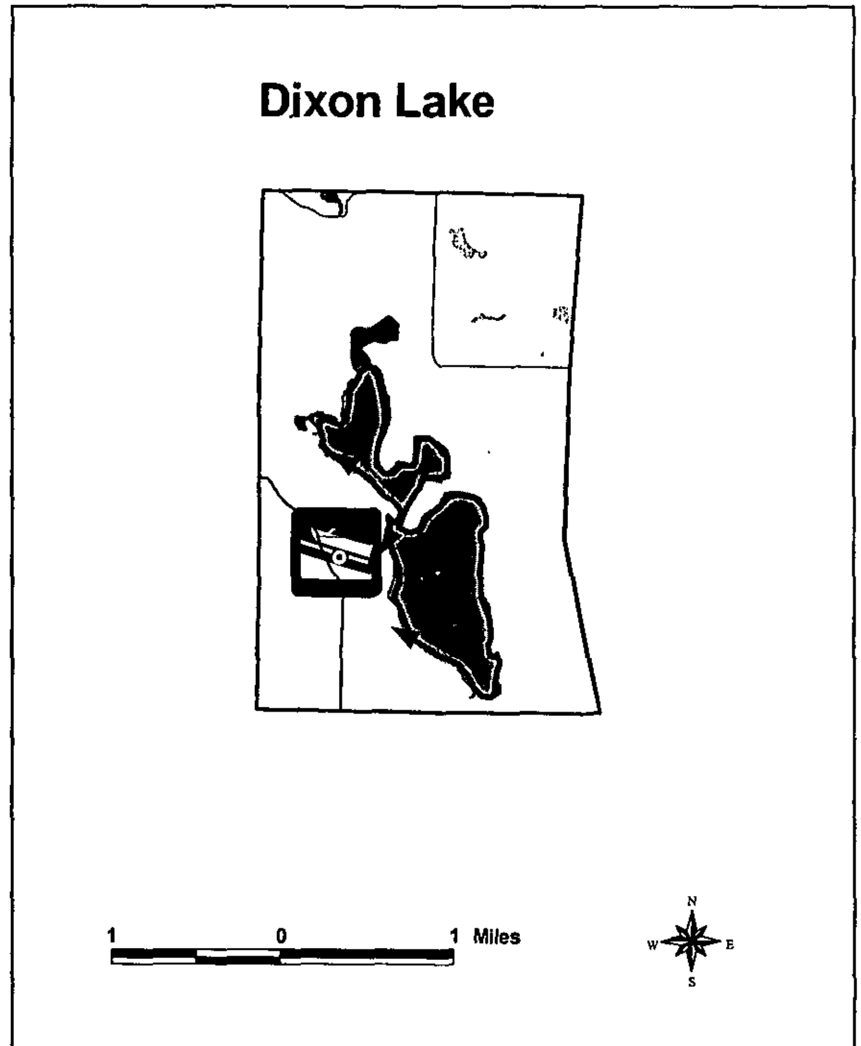


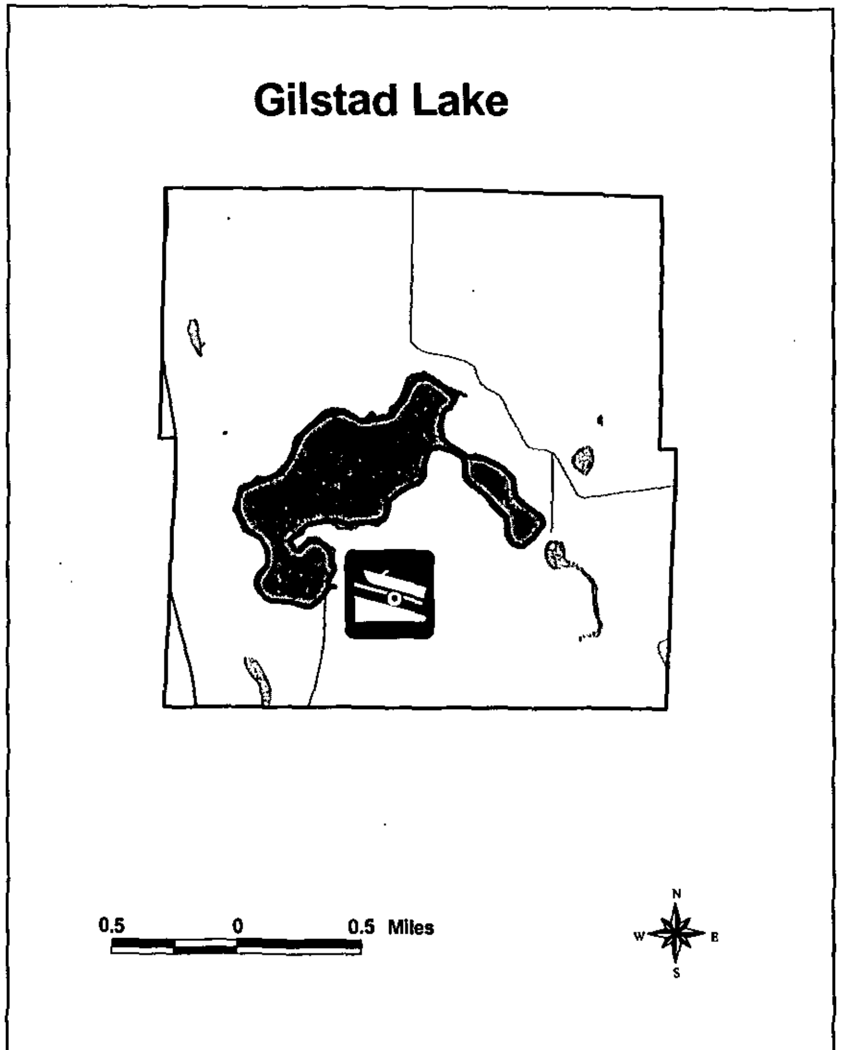
Figure 7. Indicated American wigeon pairs on Bemidji Area Pair Survey, 1959-2003.

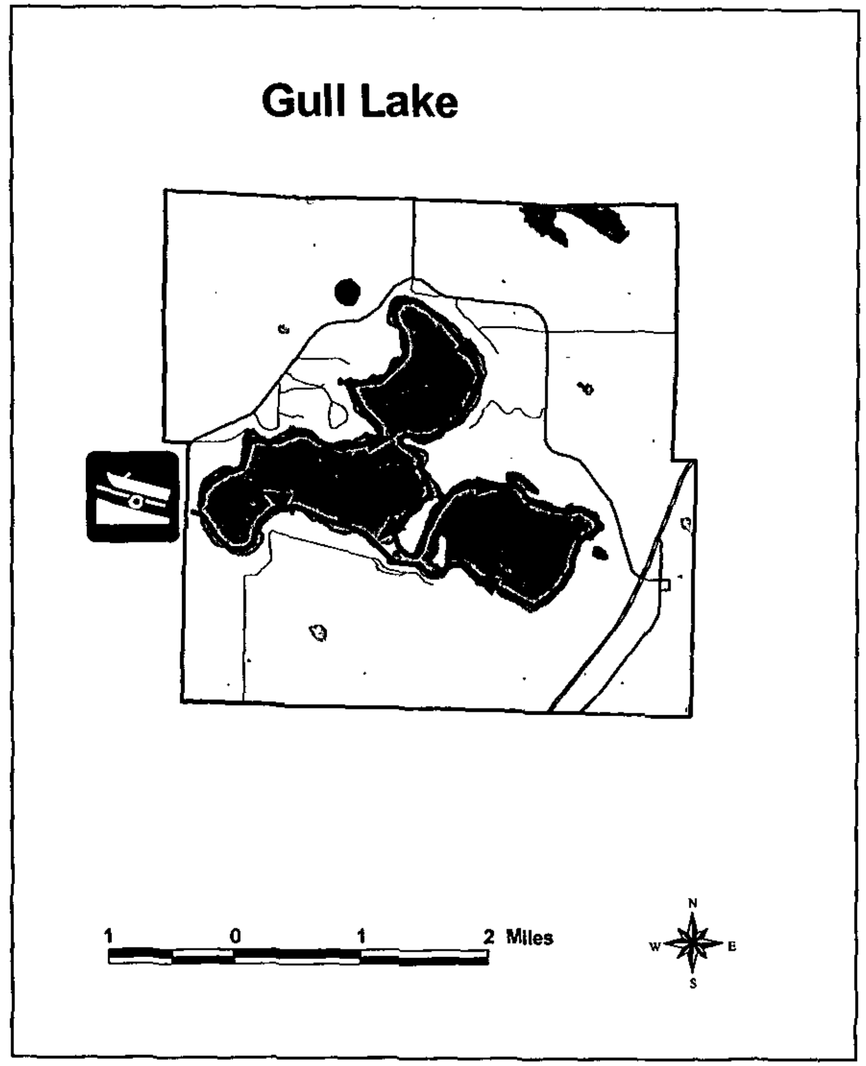
Appendix A. Access points and survey routes on lakes included in the Bemidji Area Pair Survey, 1959-2003.



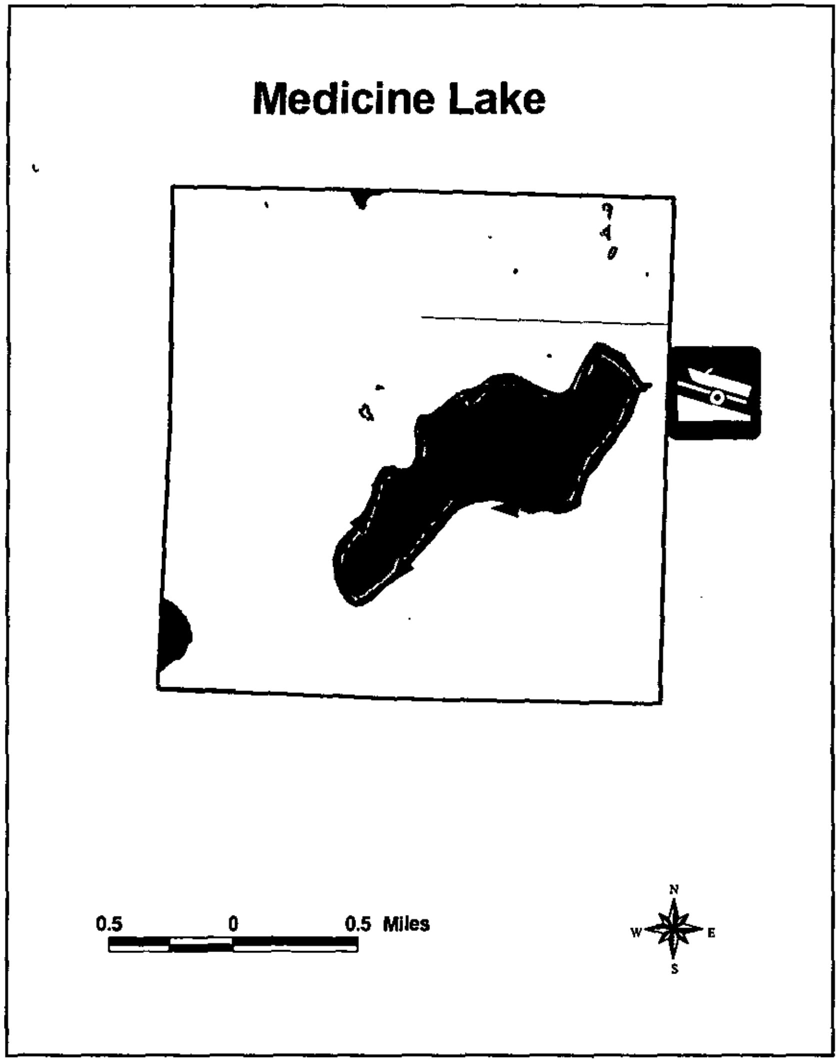


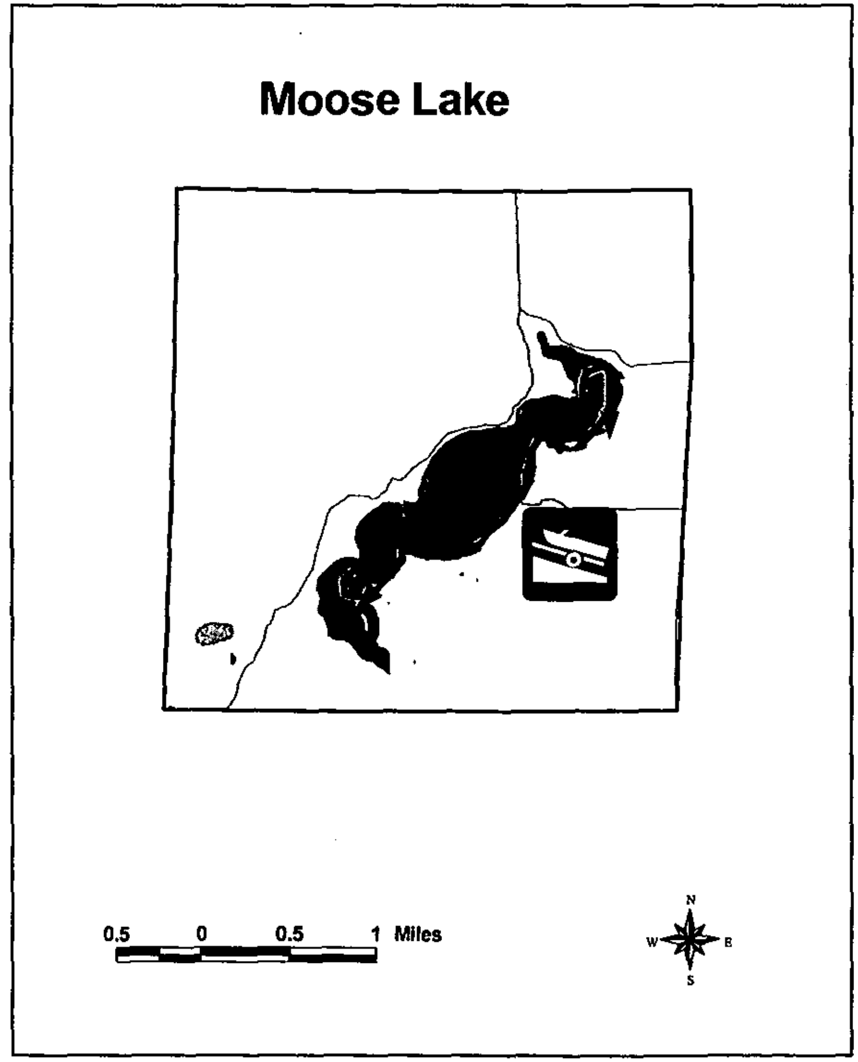
Appendix A (continued).





Appendix A (continued).





Appendix B. Total indicated breeding pairs on lakes included in the Bemidji Area Pair Survey, 1959-2003.

Year	Mallard	Common goldeneye	Blue-winged teal	American wigeon	Wood duck	Ring-necked duck	Hooded merganser	Ruddy duck	Green-winged teal	Northern shoveler
1959	157	79	45	39	14	8	2	3	3	2
1960	126	89	42	40	18	5	1	1	1	1
1961	121	85	40	25	18	11	2	1	0	1
1962	85	86	34	33	20	13	6	0	0	0
1963	117	75	45	39	31	12	8	0	2	0
1964	95	73	55	50	24	23	3	0	3	0
1965	51	69	37	21	11	12	0	0	0	0
1966	76	62	22	25	13	15	3	0	1	1
1967	39	55	23	22	5	1	3	0	4	2
1968	58	45	23	19	12	2	2	0	2	0
1971	70	51	25	26	7	7	3	0	0	1
1972	38	52	18	19	6	8	1	0	0	0
1973	136	48	50	20	22	11	2	1	3	3
1974	45	55	21	17	18	19	6	0	0	0
1975	43	65	10	22	6	18	1	0	0	0
1976	117	51	49	33	27	11	3	0	4	5
1977	186	40	83	28	40	7	1	0	0	0
1978	87	47	31	22	15	4	0	0	0	0
1979	70	62	22	11	10	8	0	0	2	0
1980	188	44	79	16	34	5	1	0	3	8
1981	209	50	59	29	33	1	2	0	0	7
1982	91	63	47	15	5	6	0	1	1	4
1983	144	57	41	12	25	0	0	0	2	3
1984	115	54	36	13	9	1	1	0	0	0
1985	36	59	12	4	4	3	1	0	0	0
1986	51	57	8	5	6	2	2	0	0	0
1987	193	47	23	7	32	4	3	0	2	0
1988	151	45	20	8	7	1	3	0	1	0
1989	116	43	11	4	5	0	2	0	0	0
1990	196	54	30	5	18	0	4	0	2	0
2003	200	122	11	2	33	6	11	0	4	4

Appendix B (continued).

Year	Black duck	Gadwall	Red head	Bufflehead	Northern pintail	Lesser scaup	Red breasted merganser	Common merganser	Canada goose	Trumpeter swan
1959	0	0	0	0	0	0	0	0	0	0
1960	0	0	0	0	0	0	0	0	0	0
1961	0	0	0	0	0	0	0	0	0	0
1962	0	0	0	0	0	0	0	0	0	0
1963	0	0	0	0	0	0	0	0	0	0
1964	0	0	0	0	0	0	0	0	0	0
1965	0	0	0	0	0	0	0	0	0	0
1966	0	0	0	0	0	0	0	0	0	0
1967	1	0	0	0	0	0	0	0	0	0
1968	0	0	0	0	0	0	0	0	0	0
1971	0	0	0	0	0	0	0	0	0	0
1972	0	0	0	0	0	0	0	0	0	0
1973	0	0	0	0	0	0	0	0	0	0
1974	0	0	0	0	0	0	0	0	0	0
1975	0	0	0	0	0	0	0	0	0	0
1976	0	0	0	0	0	0	0	0	0	0
1977	0	0	1	0	0	0	0	0	0	0
1978	0	2	0	0	0	0	0	0	0	0
1979	0	0	0	0	0	0	0	0	0	0
1980	0	1	0	0	0	0	0	0	0	0
1981	0	0	0	0	0	0	0	0	0	0
1982	0	0	0	1	0	0	0	0	0	0
1983	0	0	0	1	0	0	0	0	0	0
1984	0	0	0	1	1	0	0	0	0	0
1985	0	0	0	0	0	0	0	0	0	0
1986	0	0	0	0	0	0	0	0	0	0
1987	0	0	0	0	0	0	0	0	0	0
1988	0	0	1	0	0	0	0	0	0	0
1989	0	2	0	2	0	0	0	0	0	0
1990	0	0	0	0	0	0	0	0	0	0
2003	0	0	0	2	1	2	1	1	4	1

BEMIDJI AREA RING-NECKED DUCK SURVEY

Michael C. Zicus, Todd Eberhardt, Jeff DiMatteo, and Leon L. Johnson.

ABSTRACT

The Bemidji Area Ring-necked Duck (*Aythya collaris*) Survey was initiated in 1969 to assess breeding trends in the Bemidji vicinity. Fifteen lakes/wetlands have been included in the survey at one time or another. From 1969 – 1974, 12 – 14 lakes were surveyed each year while 15 lakes were surveyed each year from 1975 – 1984. Since 1984, 14 lakes have been surveyed each year. In total, 10 lakes have been surveyed consistently since 1969. The survey is conducted during the second or third week of June. In spring 2003, 87 ring-neck pairs were counted which was similar to recent years. However, ring-necked duck pairs counted during this survey have shown a significant long-term decline. The decline is apparent in both the 10-lake data set dating back to 1969 and the 14-lake data set dating back to 1974.

INTRODUCTION

Moyle (1964) described ring-necked ducks (*Aythya collaris*) in Minnesota as nesting primarily in the northern-forested portions of the state with appreciable numbers in the forest-prairie transition zone. At the time, ring-necked ducks were believed to be the second most abundant species (to mallards) breeding in Minnesota's forest zone. However, little was known about population trends of breeding ring-necked ducks in Minnesota, and a survey was developed to monitor trends in breeding ring-necks in north central Minnesota in 1969.

Staff in the Minnesota Department of Natural Resources Wetland Wildlife Populations and Research Group has been developing a forest wetlands and waterfowl initiative. The status of ring-necked ducks has been among the topics considered because the species has been considered an important forest resident and identified in the Minnesota Department of Natural Resources' *A Vision for Wildlife and its Use – Goals and Outcomes 2003 – 2013* as an indicator species for the Forest Province. The protocol and results of the Bemidji Area Ring-necked Duck Survey have not been formally reported before, and this report is intended as documentation for the survey.

METHODS

The survey currently is conducted on 14 lakes/wetlands (Fig. 1) during the second or third week of June depending on seasonal phenology (Kolkin Marsh was dropped from the survey in 1985). Pairs are counted when approximately 50% of the indicated pairs are comprised of lone males. An observer in a boat conducts each lake count while slowly motoring around the lake or wetland perimeter (Appendix A), and data are recorded as observations are made. All ducks, geese, and swans present are counted on each survey date. In addition to pairs comprised of a drake and a hen, each hen, lone drake, or each drake in a flock of 5 or less is considered to represent a pair. Drakes in flocks of 6 or more are considered migrants and are not expanded to represent pairs.

RESULTS

Fifteen wetlands have been included in the survey at one time or another since it began in 1969. From 1969 – 1974, 12 – 14 lakes were surveyed each year while 15 lakes were surveyed each year from 1975 – 1984. Since 1984, 14 lakes have been surveyed. In total, 10 lakes have been surveyed every year since 1969. Past survey data are included in Appendix B.

The 2003 survey was conducted from 10-18 June. Eighty-seven ring-neck pairs were counted which was similar to recent years. However, ring-necked duck pairs counted during this survey have shown a significant long-term decline (Fig. 2). The apparent decline is similar in both the 10-lake data set dating back to 1969 and the 14-lake data set dating back to 1974.

DISCUSSION

Ring-necked ducks are an important species breeding within Minnesota's Laurentian Mixed Forest Province. There is a tendency for the number of pairs counted in the Bemidji Area Survey to increase in dry years and decrease in wetter years; nonetheless, the number of breeding pairs counted has declined by approximately two thirds since inception of the survey. Care must be used when drawing inferences from a survey such as this because the geographic scope is clearly limited. Wetlands included in the survey were considered some of the best ring-necked duck breeding habitat within the surveyed area at the time the survey was initiated and habitat conditions in and around these wetlands appear to have changed little since. However, with increasing development in north central Minnesota, these wetlands will almost certainly experience habitat changes in the future.

The Minnesota Department of Natural Resources' *Restoring Minnesota's Wetland and Waterfowl Hunting Heritage* acknowledged that a sizable population of Minnesota-breeding ducks is the cornerstone to improving fall duck use. Properly designed breeding population surveys are needed to monitor the status of all species of resident forest waterfowl. As the Wetland Group considers future work in the Laurentian mixed forest province, it is clear that additional breeding waterfowl surveys will be needed before the status of ring-necked ducks and other forest breeding waterfowl can be established with any certainty. In the interim, this survey can continue to provide data that can be compared to those collected beginning in 1969. As such, it will be useful to help define priority work regarding the local population status of ring-necked ducks.

ACKNOWLEDGMENTS

D. Rave prepared the figures for this report and provided useful comments.

LITERATURE CITED

Moyle, J. B. (ed.). 1964. Waterfowl in Minnesota. Technical Bulletin No. 7. Minnesota Department of Conservation.

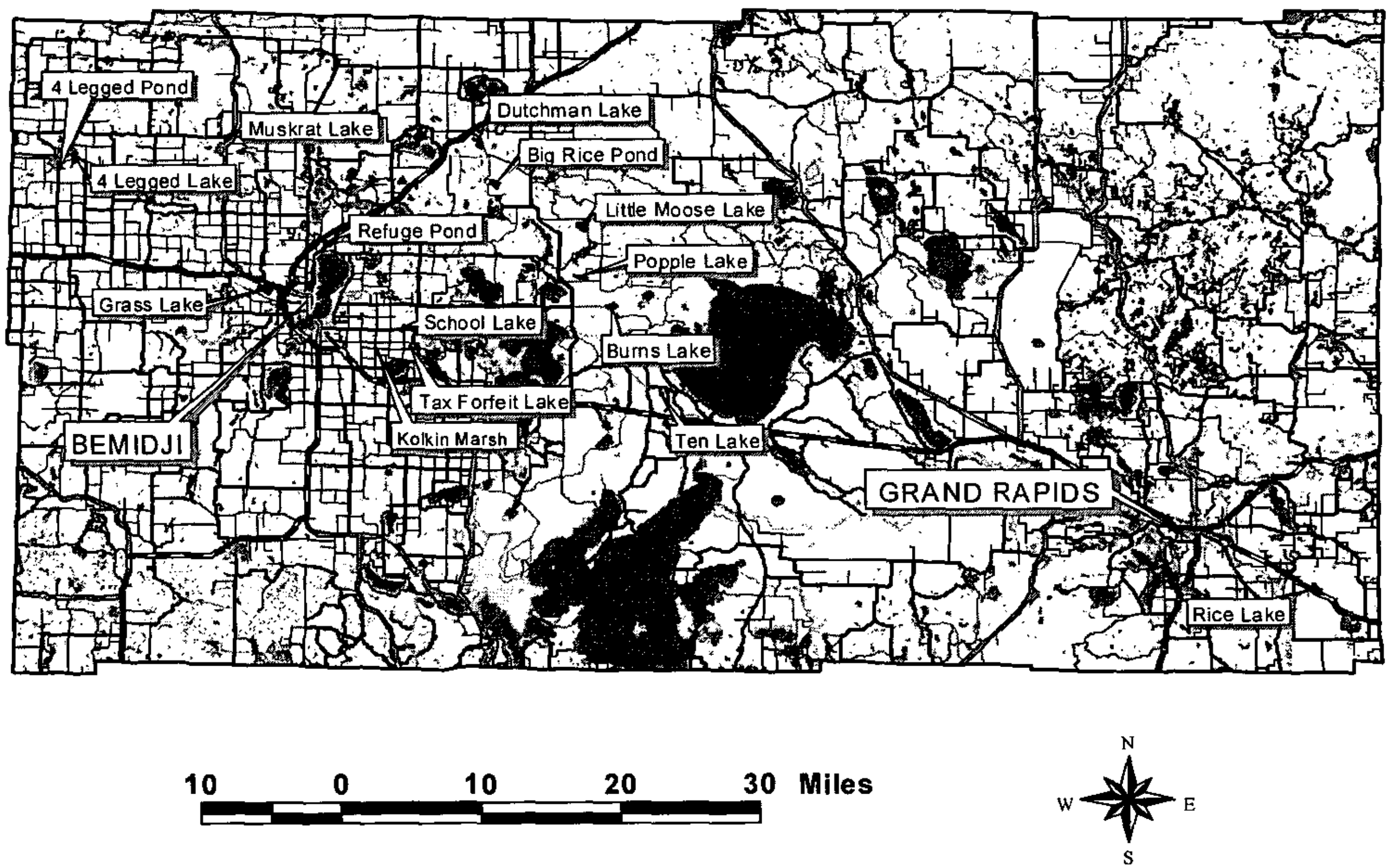


Figure 1. Location of lakes/wetlands surveyed in the Bemidji Area Ring-necked Duck Survey.

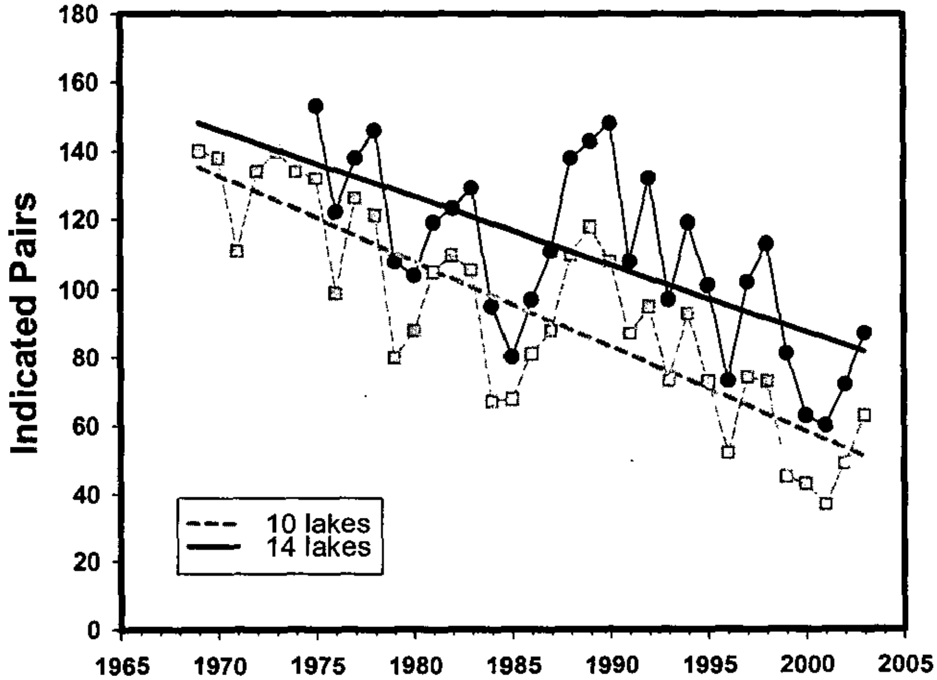
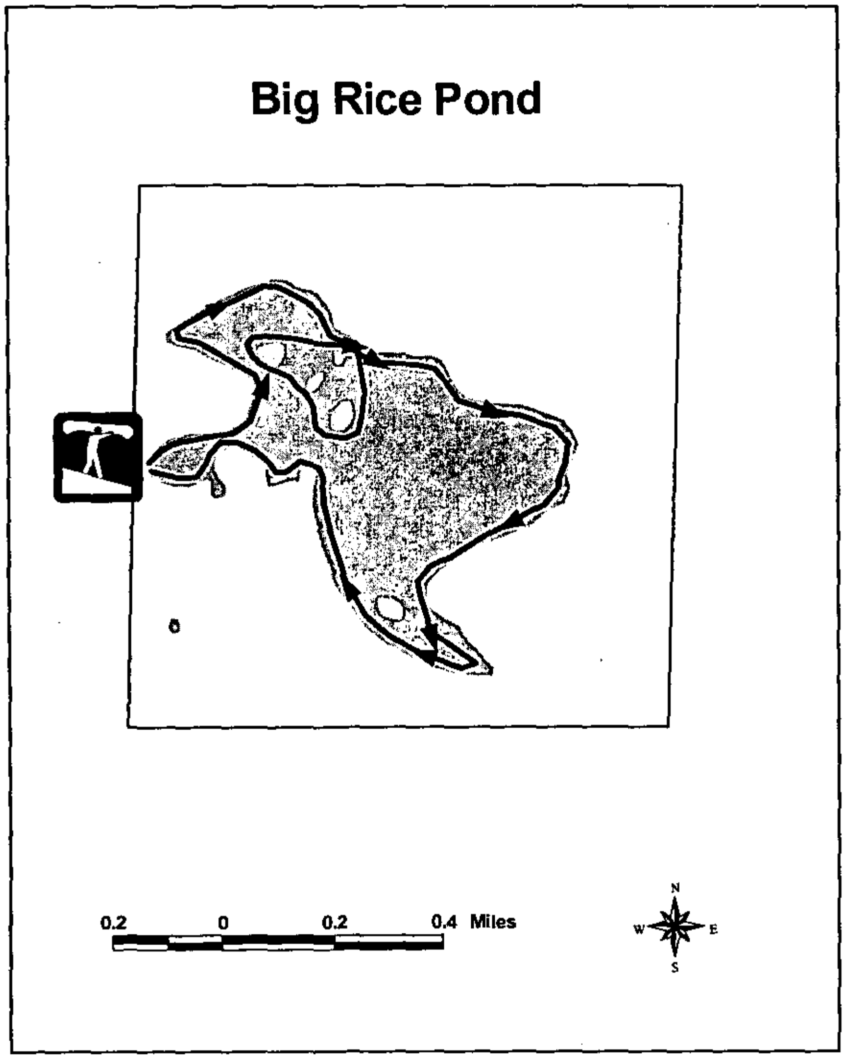
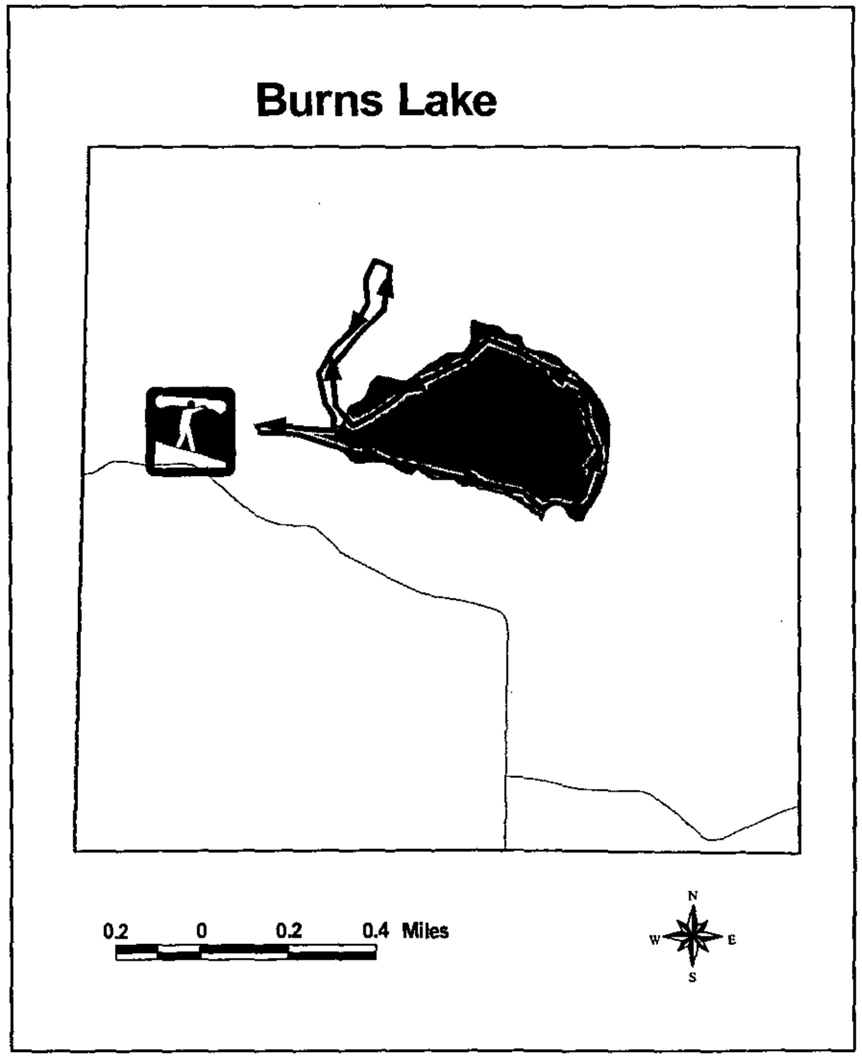


Figure 2. Indicated ring-necked duck pairs on the Bemidji Area Ring-Necked Duck Survey, 1969-2003.

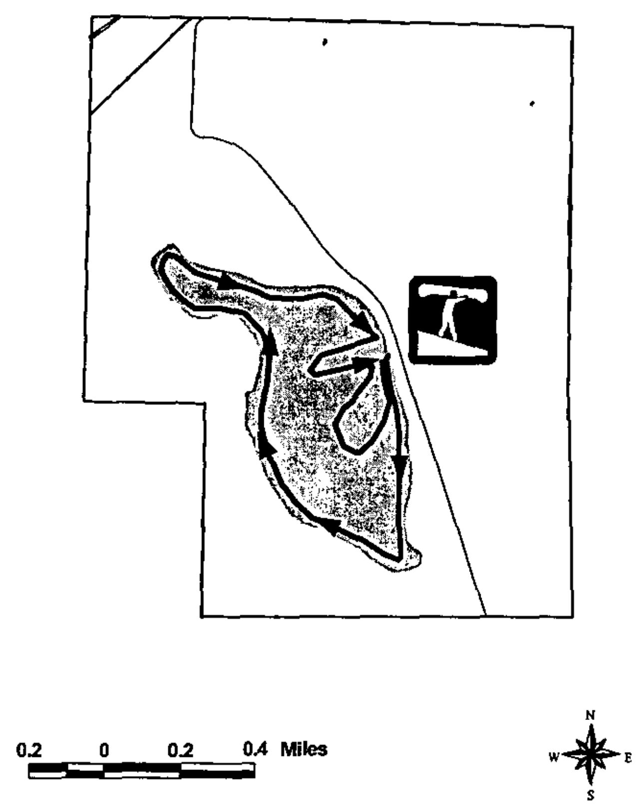
Appendix A. Access points and survey routes on lakes included in the Ring-necked Duck Survey, 1969-2003.



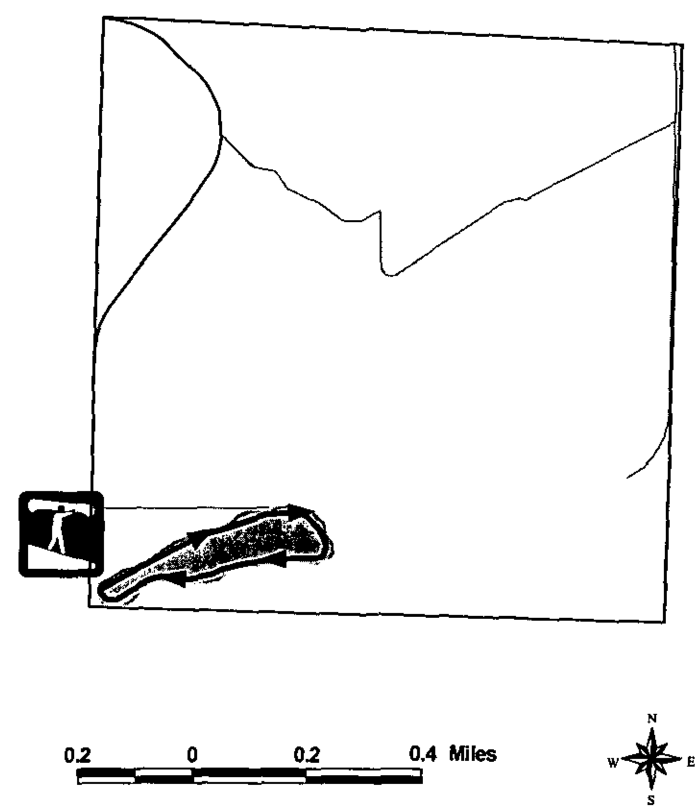


Appendix A (continued).

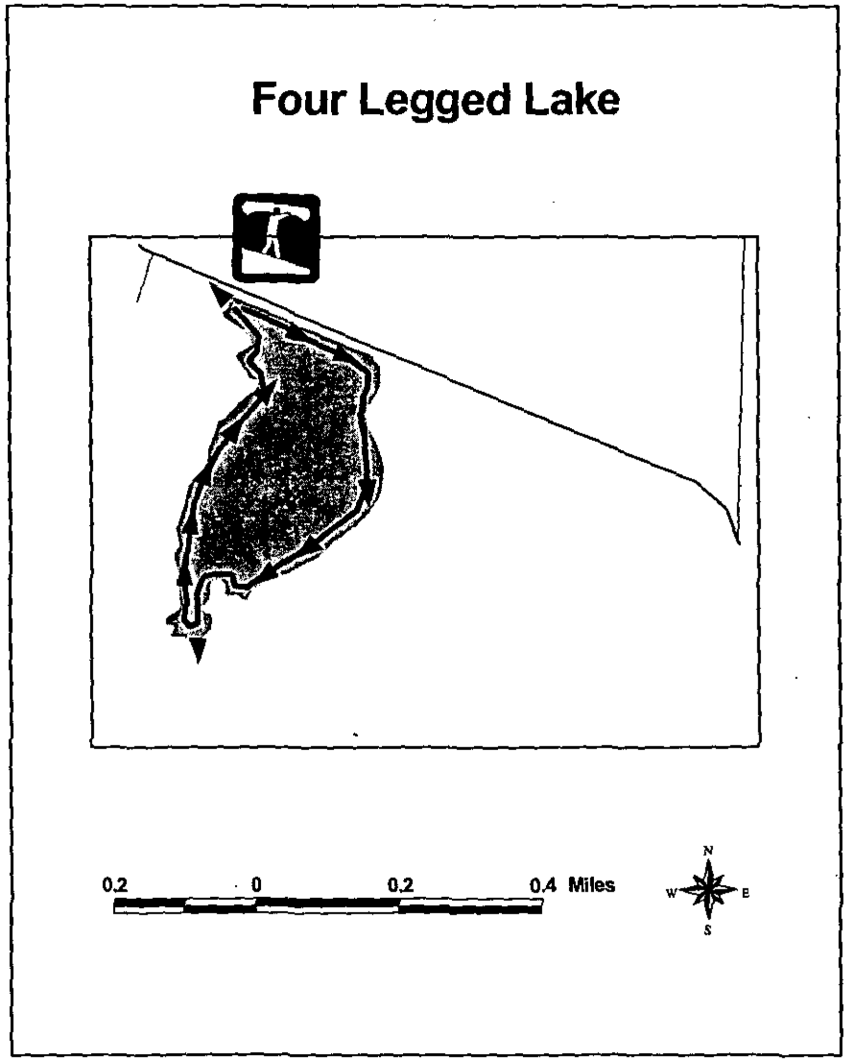
Dutchman Lake

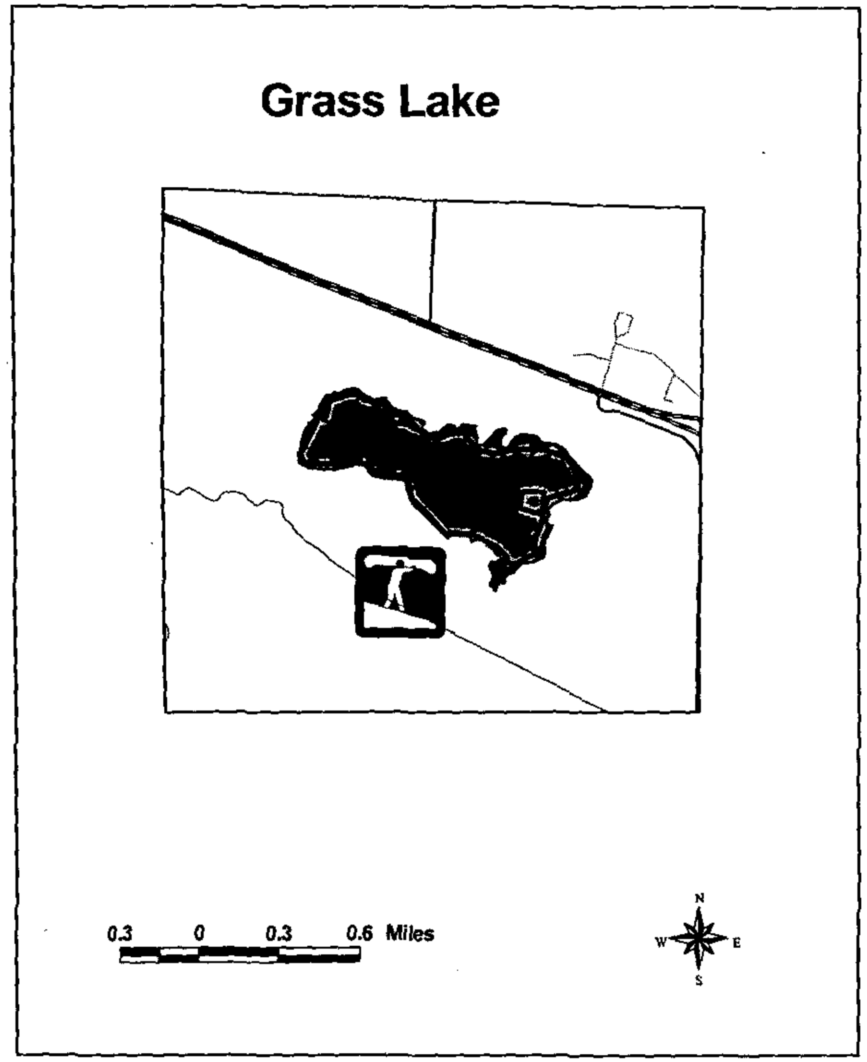


Four Legged Pond

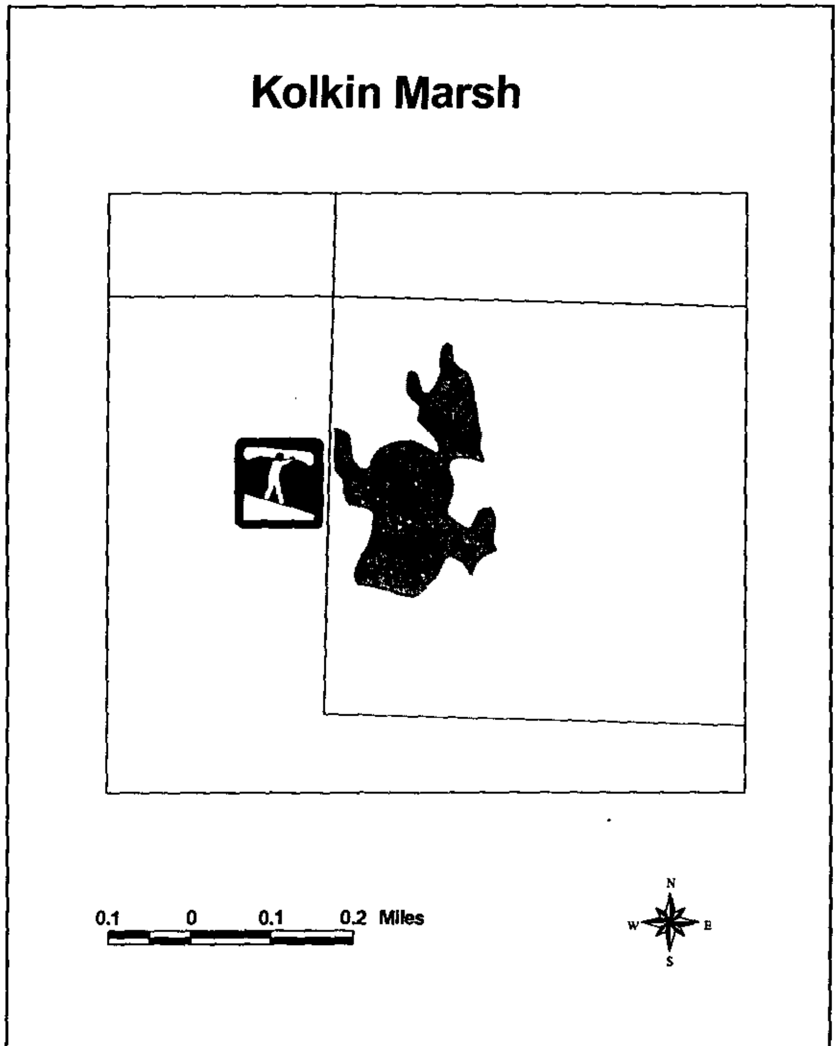


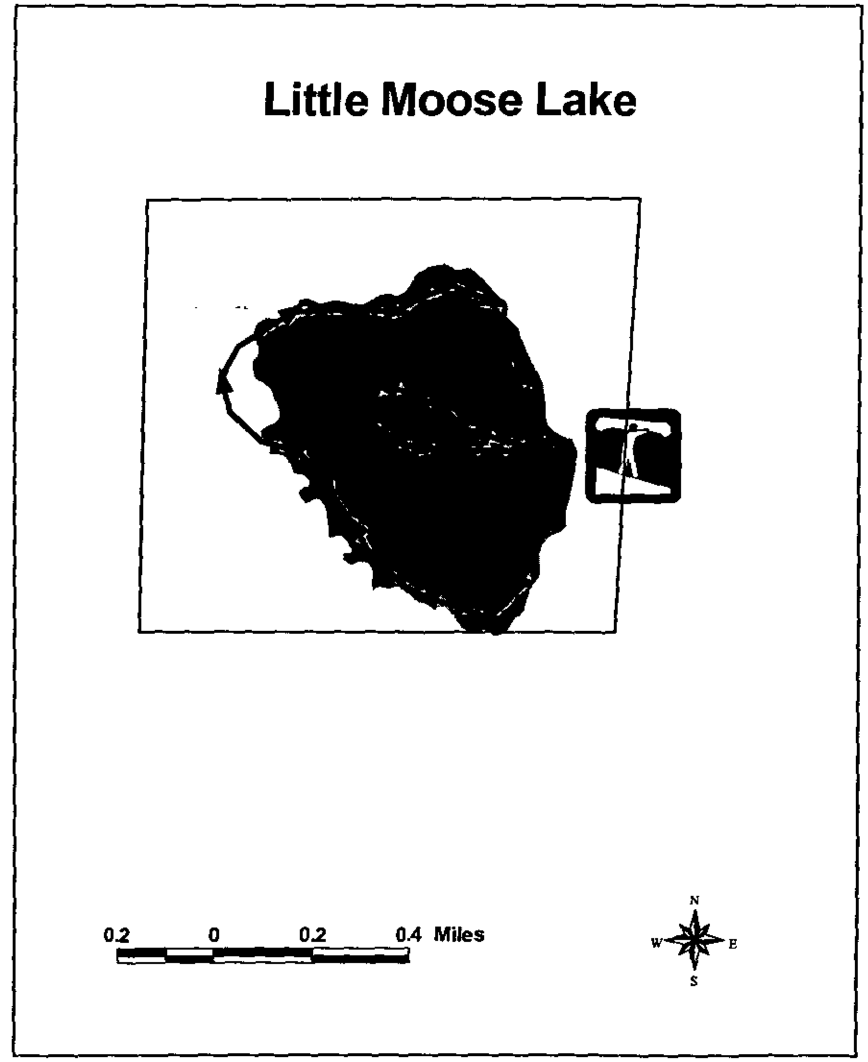
Appendix A (continued).



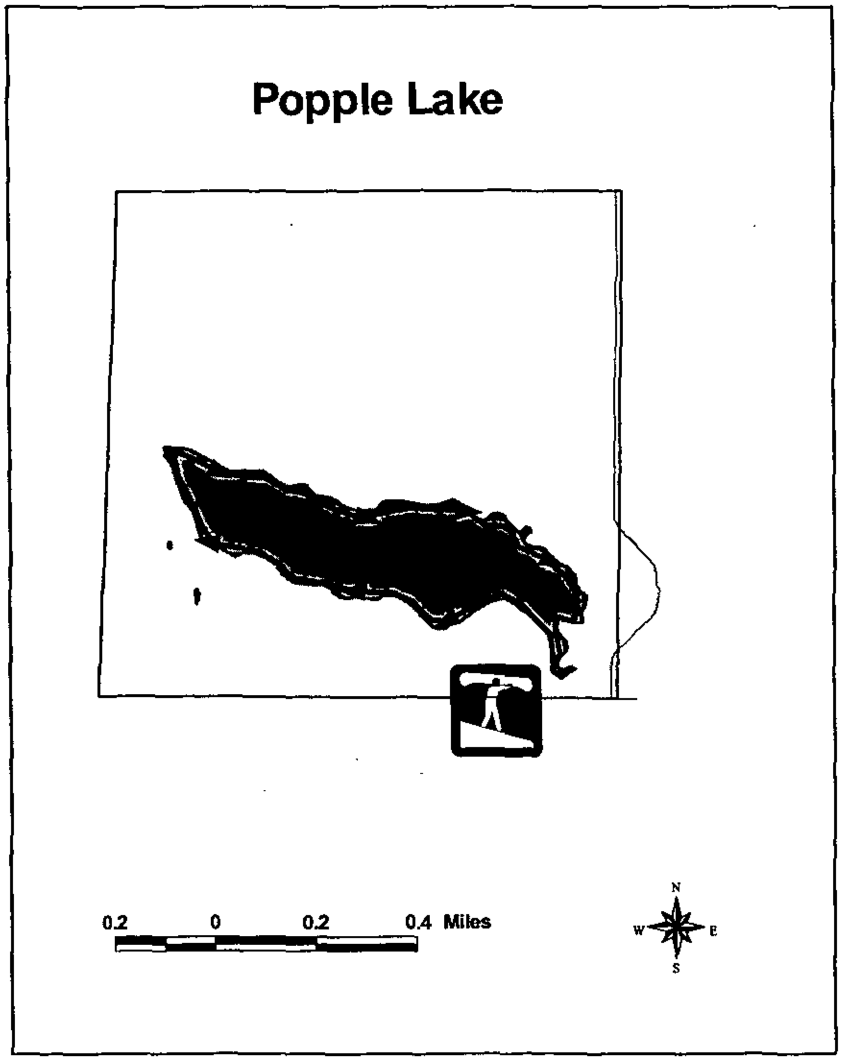


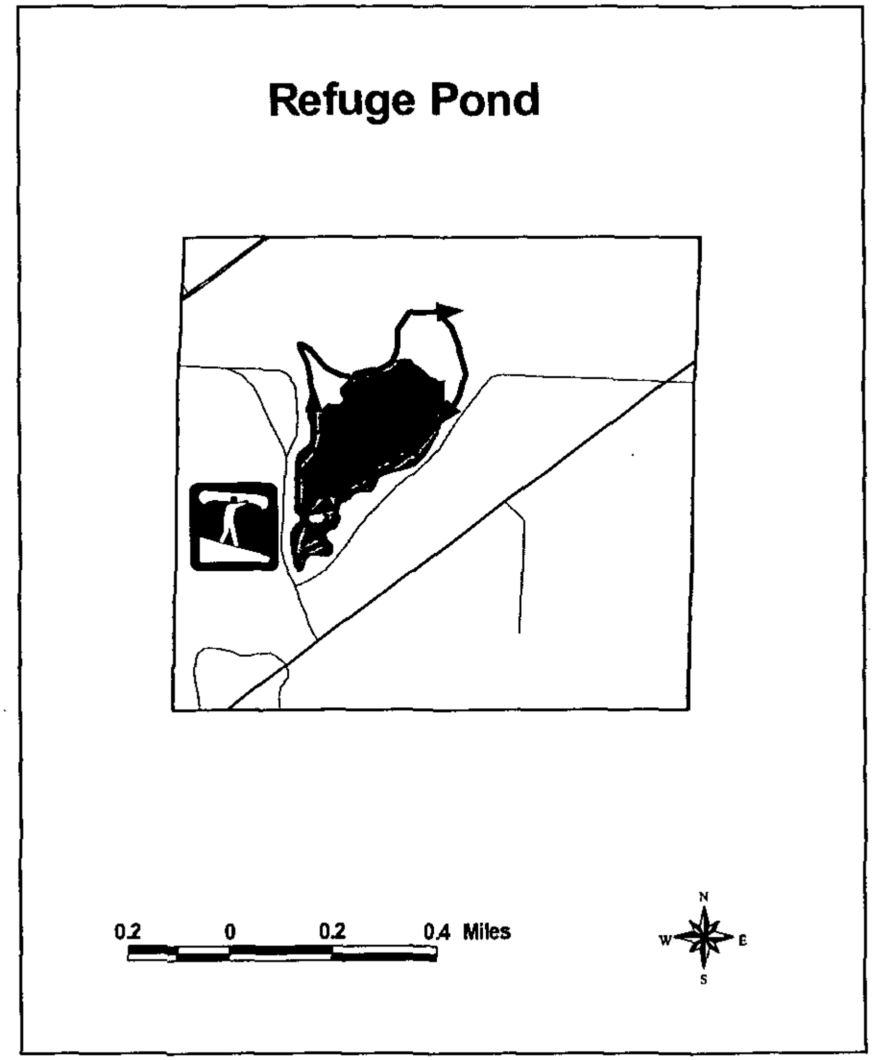
Appendix A (continued).



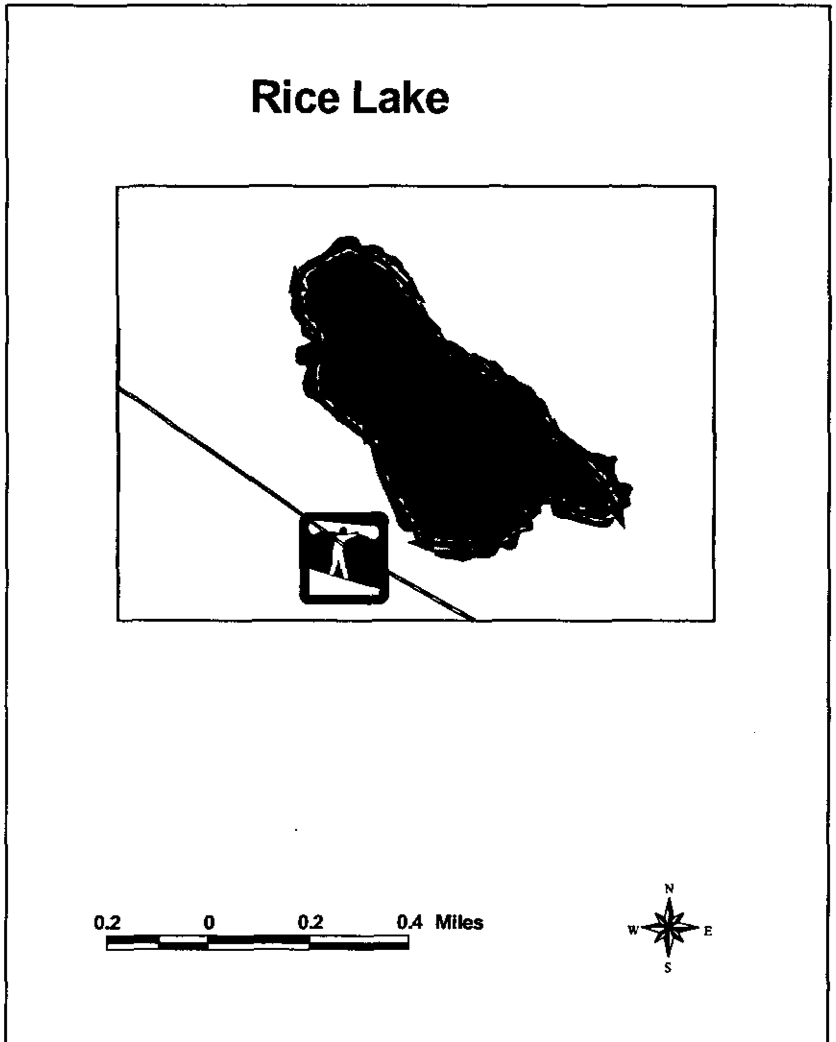


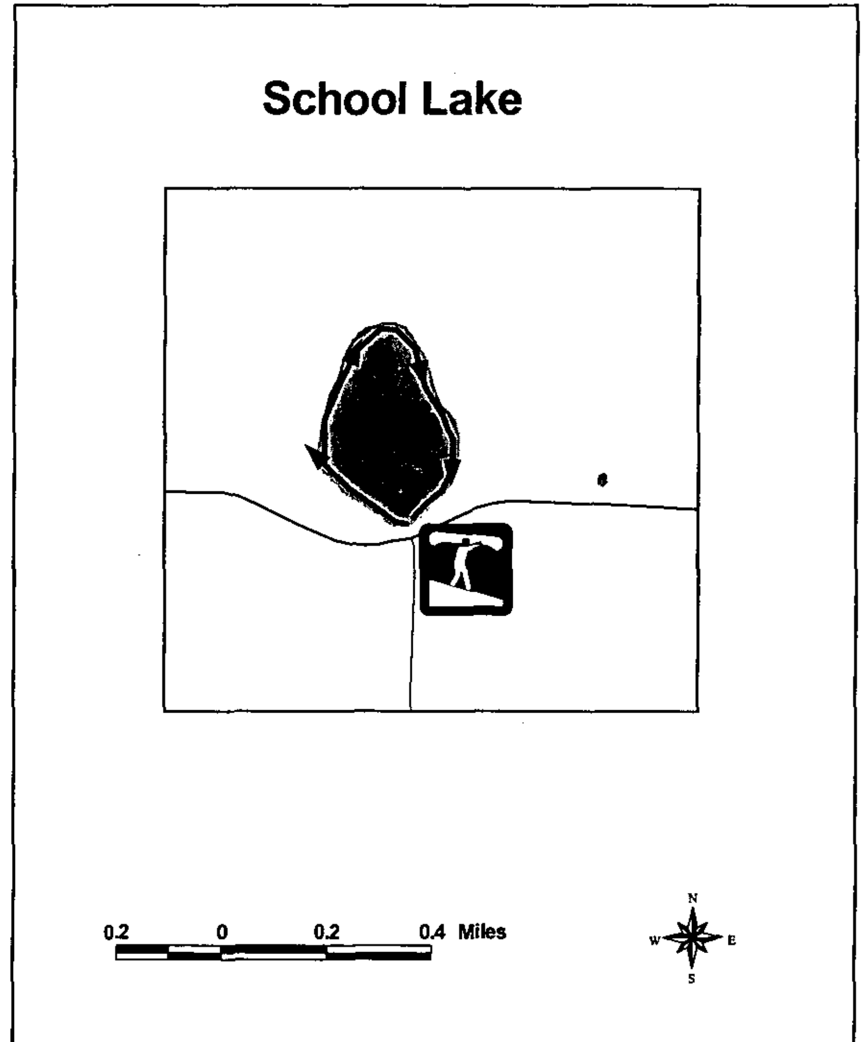
Appendix A (continued).



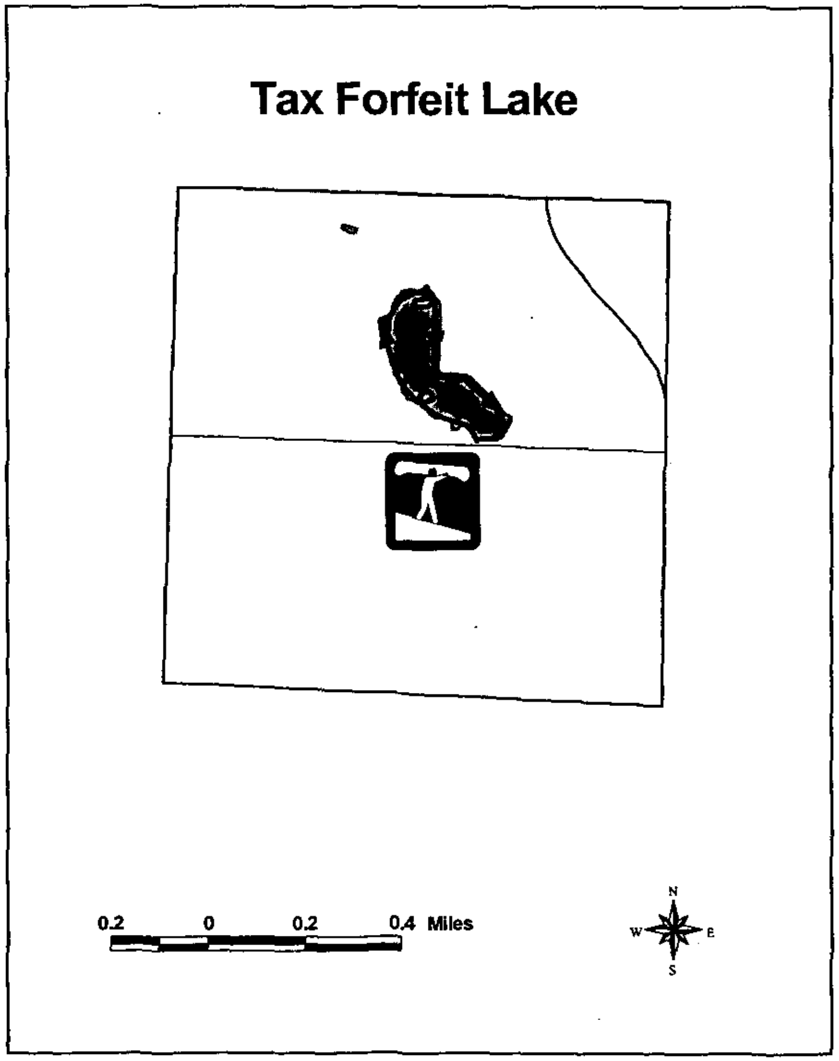


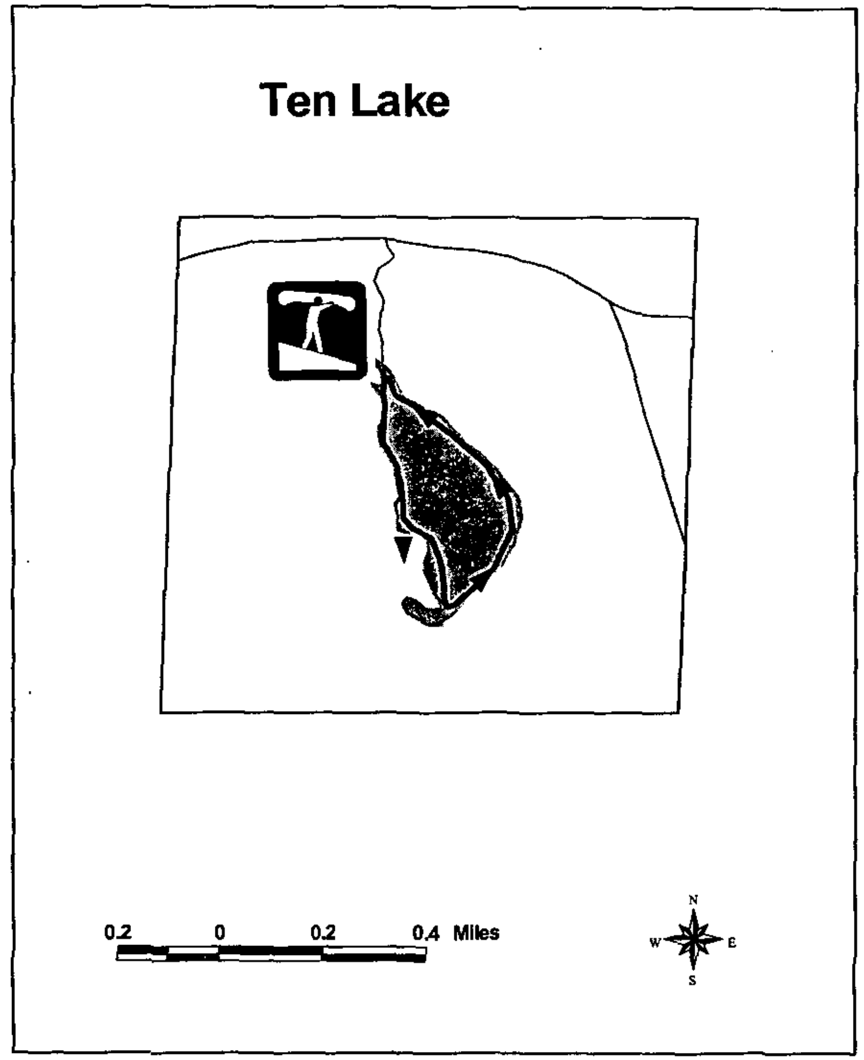
Appendix A (continued).





Appendix A (continued).





Appendix B. Ring-necked duck breeding pair counts for the lakes/wetlands included in the Bemidji Area Ring-necked Duck Survey, 1969-2003 (part 1).

Year	Big Rice Pond	Burns Lake	Dutchman Lake	Four-legged Lake	Four-legged Pond	Grass Lake	Little Moose Lake	Muskrat Lake
1969	15		14	10	7	30	18	
1970	17	7	9	13	10	30	24	
1971	14	6	9	6	7	21	18	
1972	8	8	10	9	15	33	5	
1973	11	12	12	11	8	32	5	
1974	12	6	9	8	10	20	9	
1975	13	3	14	5	15	19	16	9
1976	14	2	7	9	5	15	1	16
1977	10	2	16	5	0	16	22	5
1978	7	0	15	12	3	17	18	12
1979	4	9	4	7	10	11	11	4
1980	1	0	3	6	7	12	16	7
1981	13	1	7	9	0	20	19	6
1982	6	3	4	13	0	18	20	2
1983	7	1	12	9	1	13	16	14
1984	7	3	6	9	2	6	8	15
1985	4	1	5	12	0	10	4	4
1986	3	2	7	12	4	10	8	7
1987	5	2	14	12	3	17	12	10
1988	12	8	16	20	4	21	13	6
1989	12	3	15	27	4	21	9	10
1990	11	7	10	29	1	25	5	14
1991	6	8	16	14	0	20	4	3
1992	3	7	14	19	2	19	8	21
1993	11	6	9	14	2	8	1	15
1994	6	3	12	14	2	17	11	16
1995	6	11	8	7	3	17	5	11
1996	7	6	2	5	3	12	3	8

Appendix B, part 1 (continued).

Year	Big Rice Pond	Burns Lake	Dutchman Lake	Four-legged Lake	Four-legged Pond	Grass Lake	Little Moose Lake	Muskrat Lake
1997	7	4	5	2	4	11	27	14
1998	9	10	13	3	3	6	14	11
1999	11	14	3	3	3	8	8	5
2000	5	9	3	1	0	10	2	4
2001	10	6	6	1	0	4	7	5
2002	16	11	7	5	4	4	8	8
2003	9	13	14	9	7	8	7	2

Appendix B. Ring-necked duck breeding pair counts for the lakes/wetlands included in the Bemidji Area Ring-necked Duck Survey, 1969-2003 (part 2).

Year	Popple Lake	Refuge Pond	Rice Lake	School Lake	Ten Lake	Tax Forfeit Lake	Kolkin Marsh
1969	16	9	18	3	6		8
1970	5	13	15	2			5
1971	7	13	9	7	7	1	3
1972	10	12	22	10	14	8	3
1973	14	14	19	14	4	8	6
1974	14	23	18	11		3	6
1975	5	14	24	7	9	8	11
1976	6	16	20	6	5	1	7
1977	12	15	19	11	5	5	0
1978	7	10	29	3	13	4	0
1979	10	6	9	8	15	2	3
1980	14	12	14	3	9	6	1
1981	9	13	15	0	7	5	1
1982	14	11	20	4	8	2	2
1983	4	9	32	3	8	0	2
1984	0	8	19	2	10	0	3
1985	0	8	23	2	7	0	
1986	0	7	28	2	7	0	
1987	0	7	17	1	11	1	
1988	2	9	12	1	14	4	
1989	1	11	15	3	12	1	
1990	3	12	8	4	19	2	
1991	0	9	15	3	10	4	
1992	5	13	10	2	9	5	
1993	2	12	11	3	3	10	
1994	4	9	15	3	7	3	
1995	2	6	19	0	6	5	
1996	0	2	16	2	7	0	

Appendix B, part 2 (continued).

Year	Popple Lake	Refuge Pond	Rice Lake	School Lake	Ten Lake	Tax Forfeit Lake	Kolkin Marsh
1997	0	6	12	0	10	0	
1998	0	2	23	0	19	0	
1999	0	2	7	0	17	0	
2000	0	1	21	0	7	1	
2001	0	1	5	3	12	0	
2002	0	2	3	0	4	0	
2003	0	1	8	0	9	1	

DOES MALLARD CLUTCH SIZE VARY WITH LANDSCAPE COMPOSITION: A DIFFERENT VIEW*

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

We report on the relationship between Mallard clutch size and cropland area in the landscape in western Minnesota during 1997 – 1999. We measured clutch size in two types of nest structures and fit a mixed-effects model to the data to examine the relationship. Our model also included covariates to control for the effects of year, nest initiation date, estimated pair numbers, and nest structure type. Unique landscapes associated with each nest (n = 134) ranged from 46.4 – 84.8% cropland. Clutch size was unrelated to cropland area, nest structure type, and estimated number of pairs with access to structures. Mean clutch size declined with nest initiation date early in the nesting season, but increased somewhat for nests initiated after 30 May. Clutch size also differed among years. Mean clutch size, adjusted for nest initiation date, was 11.0 $\pm$ 0.19 SE for 1997, 10.5 $\pm$ 0.19 SE for 1998, and 11.0 $\pm$ 0.19 SE for 1999. Conclusions regarding the significance of the year effect and the degree of nonlinearity due to nest initiation date were sensitive to potential clutch size outliers, but cropland area had no effect on clutch size regardless of the way we constrained clutch size. Nest parasitism by philopatric females laying in certain structures might explain the observed increase in clutch size in late nest initiations. *Received 29 June 2003, accepted 24 August 2003.*

*Abstract of paper in Wilson Bulletin 115(4):409-413.

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

Michael C. Zicus and David P. Rave

ABSTRACT

We collected 45 common goldeneye (*Bucephala clangula*) and 42 hooded merganser (*Lophodytes cucullatus*) eggs from northern Minnesota for contaminant assays and to determine eggshell thicknesses. Contaminant assays have not begun yet whereas eggshells have been measured. Mean eggshell thickness was 0.403 $\pm$ 0.004 SE mm and 0.602 $\pm$ 0.009 SE mm for common goldeneye and hooded merganser eggs respectively, which was 9.5 and 6.0% greater than in 1981 but still 7.4 and 4.1% less than that measured prior to the use of DDT (Zicus et al. 1988). Ratcliffe indexes increased proportionately less than did eggshell thickness for goldeneyes and remained 4.7% less than the pre-1900 value. The index was unchanged from 1981 and remained 6.1% less than the pre-DDT value for mergansers. These eggshell thickness/density metrics suggest a 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 cause them to be vulnerable to these contaminants. Egg samples from northeastern Minnesota are needed as is funding for the chemical assays.

INTRODUCTION

Staff in the Minnesota Department of Natural Resources Wetland Wildlife Populations and Research Group has been developing a forest wetlands and waterfowl initiative. The welfare of common goldeneyes (*Bucephala clangula*) has been among the topics considered because Minnesota common goldeneye populations might be declining in some locations (Fig. 1), and the species has been identified in the Minnesota Department of Natural Resources' *A Vision for Wildlife and its Use – Goals and Outcomes 2003 – 2013* as indicator species for the Forest Province.

Zicus et al. (1988) assayed eggs of common goldeneye and hooded mergansers (*Lophodytes cucullatus*) collected in 1981 for contaminants, and compared eggshell thicknesses with historic values. Organochlorine pesticides and polychlorinated biphenyls (PCBs) were modest in both species, but PCB levels in goldeneye eggs were greater than in merganser eggs. In comparison, geometric mean mercury (Hg) levels in merganser eggs were greater than in goldeneye eggs and were considered high enough to be a concern. 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 (Fig. 2). 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. Eggshell thinning and related shell breakage could be affecting some Minnesota common goldeneye populations negatively. Our objective was to repeat the earlier work to determine the extent to which contaminant loads and eggshell thickness of these species might have changed since 1981.

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, Minnesota Department of Natural Resources, unpubl. data). One randomly selected, unincubated egg was collected from common goldeneye and hooded merganser nests primarily within the Laurentian Mixed Forest Province of Minnesota (Fig. 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

In spring 2003, we collected 45 common goldeneye and 42 hooded merganser eggs from northern Minnesota. Cooperators assisted with the egg collections and most samples came from north central Minnesota. No eggs were collected from northeastern Minnesota. Mean eggshell thickness measured at the equator was 0.403 + 0.004 SE mm and 0.602 + 0.009 SE mm for common goldeneye and hooded merganser eggs respectively. These values are 9.5 and 6.0% greater than those measured in 1981 (Table 1), but still 7.4 and 4.1% less than those measured prior to the use of DDT.

Ratcliffe indexes (Table 2), 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. The goldeneye index was 4.9% greater than in 1981, but still 4.7% less than for a sample of eggs collected prior to 1900. In contrast, there was no change in the hooded merganser index from 1981, which was 6.1% less than that of eggs collected prior to the use of DDT.

DISCUSSION

Organochlorine pesticides and PCBs in the environment 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 also has declined, it is still being deposited in aquatic ecosystems of northern Minnesota, is a significant concern in many locations, and has been identified as such in the new federal Clear Skies Initiative (http://www.epa.gov/air/clearskies/basic.html). 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, Minnesota Department of Natural Resources, personal commun.). 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. Thus, we believe assays for PBDEs would be prudent because goldeneye and merganser food habits cause them to be vulnerable to these contaminants.

Mean eggshell thickness for both mergansers and goldeneyes increased significantly between 1981 and 2003. This suggests a decreased exposure to compounds related to eggshell thinning during this period. The breeding range of common goldeneyes and hooded mergansers in Minnesota extends further east than the geographic extent of the 2003 collections. The conditions of our initial federal permit allow us to collect an additional 5 common goldeneye and 8 hooded merganser eggs. Also, our permit has been amended to allow collection of 25 additional eggs for each species from northeast Minnesota in spring 2004.

Funding is needed before we can proceed with the chemical assays. Analyses for Hg, PCBs, and organochlorine pesticides are estimated to cost approximately $250/sample (M. Briggs, Minnesota Department of Natural Resources, pers. commun.). We would need to assay >30 eggs of each species to achieve minimal precision for comparison to our earlier work. Analyses for PBDEs would be very costly and we continue to explore ways to fund these assays.

ACKNOWLEDGMENTS

We'd like to thank Mark Briggs for providing the collection jars for the chemical assays. Jeffrey Dittrich, Ross Heir, Jeff Lawrence, Mike Loss, Gretchen Mehmel, Frank Swendsen, Dan Thul, and John Williams collected eggs for us, and the Dixon Lake Association allowed us to collect eggs from their nest boxes.

LITERATURE CITED

Faber, R. A., and J. J. Hickey. 1973. Eggshell thinning, chlorinated hydrocarbons, and mercury in inland aquatic birds, 1969 and 1970. Pesticide Monitoring Journal 7:27 – 36.

Ratcliffe, D. A. 1976. Decrease in eggshell weight in certain birds of prey. Nature 215:208 – 210.

White, D. H., and E. Cromartie. 1977. Residues of environmental pollutants and shell thinning in merganser eggs. Wilson Bulletin 89:532 – 542.

Zicus, M. C., M. A. Briggs, and R. M. Pace III. 1988. DDE, PCB, and mercury residues in Minnesota Common goldeneye and Hooded merganser eggs. Canadian Journal of Zoology 66:1871 – 1876.

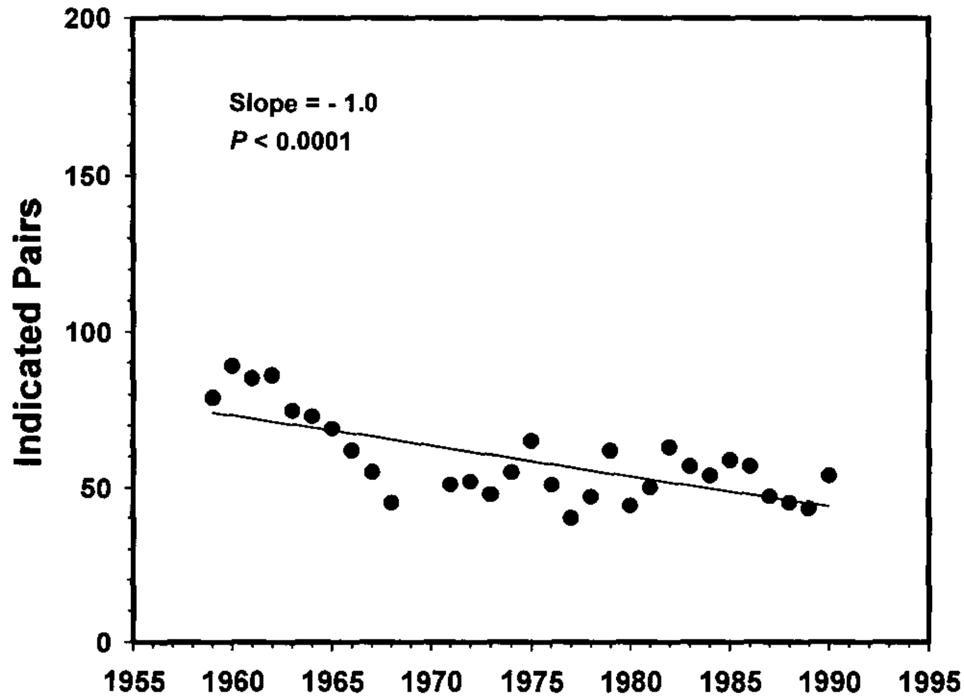


Figure 1. Indicated breeding common goldeneye pairs counted on the Bemidji Area Pair Survey declined during the period 1959 - 1990.

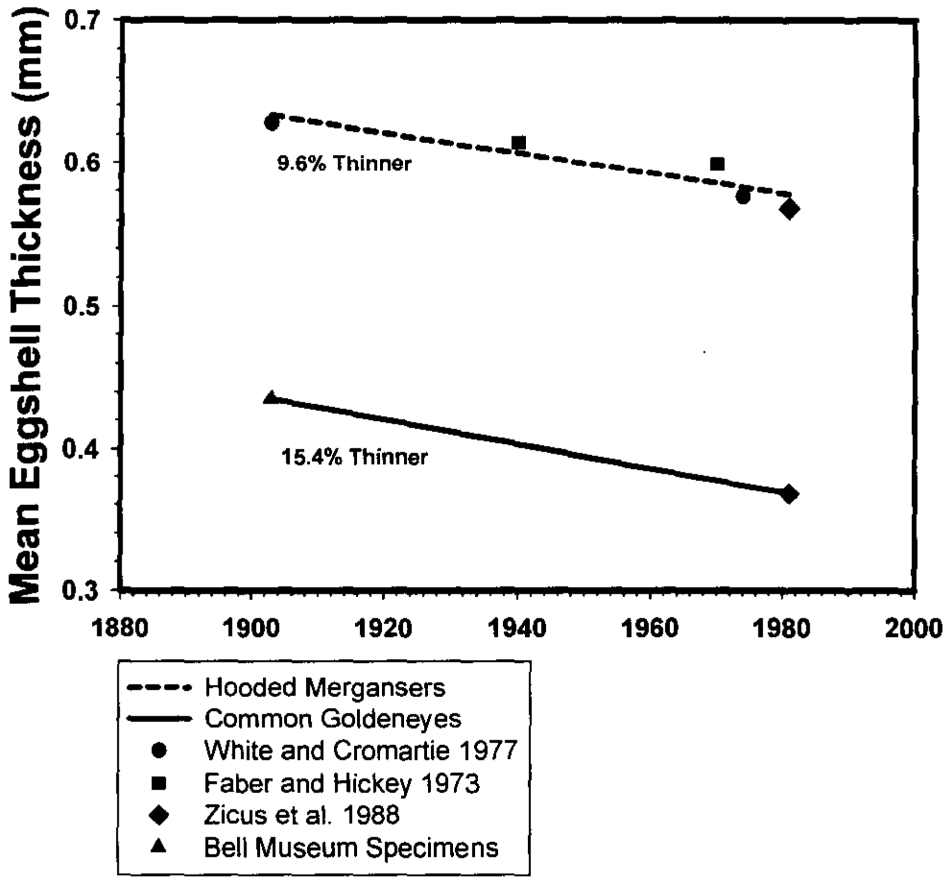


Figure 2. Mean eggshell thickness for common goldeneye and hooded mergansers declined between 1900 and 1981.

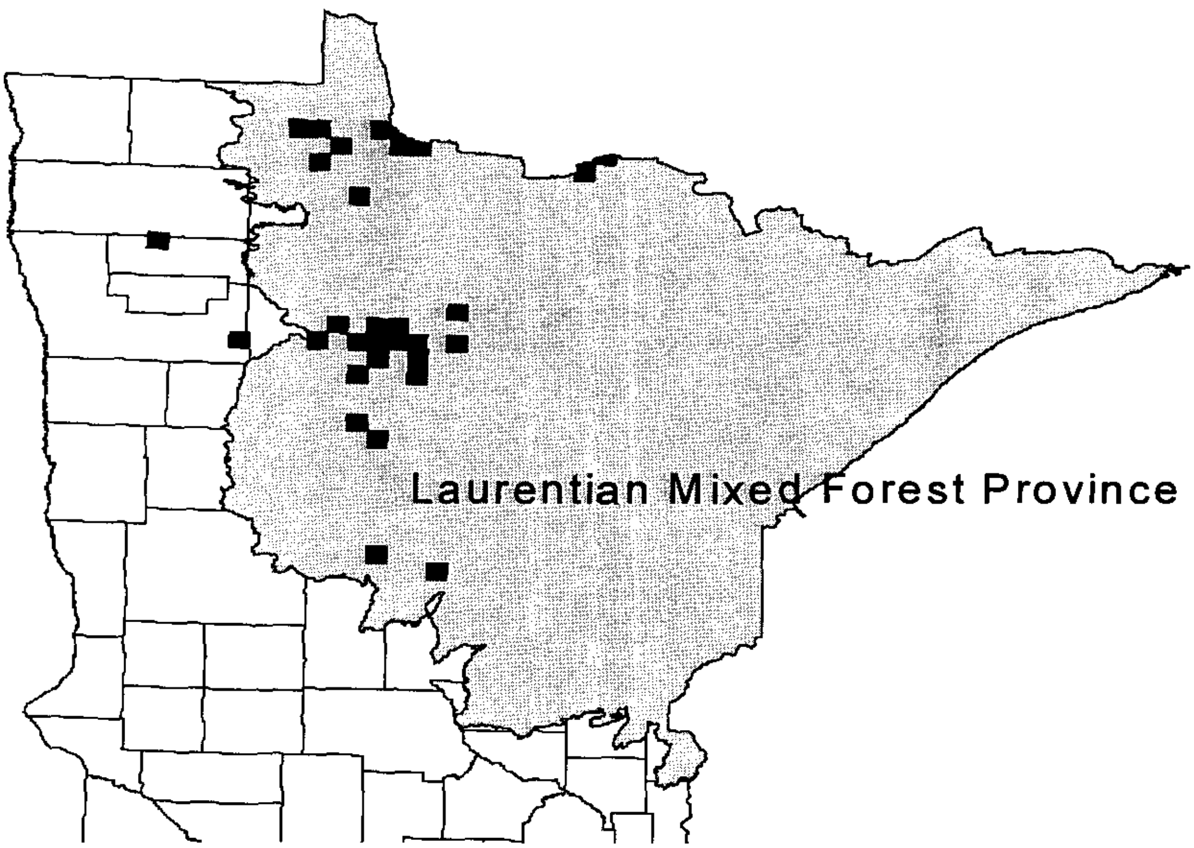


Figure 3. Minnesota townships where common goldeneye and hooded merganser eggs were collected in 2003.

Table 1. Common goldeneye and hooded merganser eggshell thicknesses measured in Minnesota in 2003 were greater than those measured in 1981 but still less than those measured ~1900.

Species		~1900	1981	2003
Common goldeneye	(mean)	0.435[a]	0.368[a]	0.403
	(95% CI)	0.423 – 0.447	0.360 – 0.376	0.396 – 0.410
Hooded merganser	(mean)	0.628[b]	0.568[a]	0.602
	(95% CI)	0.579 – 0.677	0.554 – 0.582	0.584 – 0.620

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

Table 2. Ratcliffe indexes for common goldeneye eggshells measured in Minnesota in 2003 were greater than those measured in 1981 but still less than those measured ~1900 whereas hooded merganser indexes measured in 1981 and 2003 were similar and remained less than those measured prior to 1947.

Species		~1900	1981	2003
Common goldeneye	(mean)	2.648[a]	2.405[a]	2.524
	(95% CI)	2.472 – 2.824	2.360 – 2.450	2.477 – 2.571
Hooded merganser	(mean)	4.000[b]	3.757[a]	3.755
	(95% CI)	3.890 – 4.110	3.692 – 3.822	3.651 – 3.859

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

HOODED MERGANSER NEST ATTENDANCE PATTERNS

Michael C. Zicus and David P. Rave

ABSTRACT

We initiated a study of hooded merganser (*Lophodytes cucullatus*) nest attendance in 2002 by deploying 11 wooden nest boxes in northern Hubbard County. Nest boxes replaced non-functional and missing boxes that were frequently used by hooded mergansers in an earlier study. We monitored these and 3 pre-existing boxes for hooded merganser use in 2002-2003. Hooded merganser nests were selected for monitoring opportunistically once nests were initiated, and attendance data were recorded remotely. Four nests were monitored each year for a total of 124 days in 2002 and 115 days in 2003. Data from additional nests would markedly improve the sample.

INTRODUCTION

The hooded merganser (*Lophodytes cucullatus*) is a relatively abundant cavity nesting species in Minnesota; however, its general ecology in the state is poorly understood. This project is a continuation of a cavity nesting waterfowl study begun in 1980. The objective of this project is to examine patterns of hooded merganser nest attendance to assess the variability associated with hens and incubation stage. Specifically, we are interested in the daily timing and number of recesses taken by a typical female, the average recess duration, and the total daily recess time over the course of incubation. Knowledge of these parameters will further our understanding of the bioenergetics of this little studied species.

METHODS

We deployed 11 wooden nest boxes along the shoreline of lakes and wetlands in northern Hubbard County (Paul Bunyan State Forest) in late winter 2002. These nest boxes replaced non-functional and missing boxes that were frequently used by hooded mergansers in an earlier study (Zicus 1990). We monitored these boxes and 3 pre-existing boxes for hooded merganser use in 2002-2003. Merganser nests were selected for monitoring opportunistically once nests were initiated. Nest attendance data were recorded remotely (Cooper and Afton 1981) nest was selected following methods used by Zicus et al. (1995) for common goldeneyes (*Bucephala clangula*). After installation, monitors were checked twice weekly to assure that they were functioning properly and to change chart paper and batteries.

Hooded merganser hens typically lay an egg every other day until the clutch is finished (Zicus 1990). We considered the incubation period to begin after the clutch was completed at 0:01 hr on the day hens began consistently taking one or more daytime recesses followed by an overnight session at the nest. Hooded merganser ducklings are usually brooded in the nest for about 24 hours prior to departure (M. Zicus, unpubl. data), so we defined the end of incubation and the beginning of the hatching period as 24:00 hr 2 days prior to the day ducklings departed the nest.

RESULTS

We monitored 4 nests for a total of 124 days in 2002 and 4 nests for a total of 115 days in 2003 (Table 1). Data from 10 days in 2002 and 15 days in 2003 were lost because of monitoring equipment malfunctions or disturbance of hens by our activities.

These data should be viewed cautiously, as they have not been weighted for the influences of incubation day or hen. Our 2 nest monitors were set up opportunistically as nests were found and later moved to different nests after the first nests hatched. As a result the data are unbalanced with different incubation days monitored for different hens. However, data inspection suggested no striking trends with respect to day of incubation for the number of recesses taken each day (Fig. 1) or the total amount of time in daily recesses (Fig. 2).

DISCUSSION

The lack of knowledge of hooded mergansers and other poorly studied species has been described as an "Achilles heel" (B. Batt, Ducks Unlimited, pers. commun.). Although the hooded merganser is not an important bird in the hunter's bag, Minnesota hunters have taken ~13% of the Mississippi flyway hooded merganser harvest in recent years (i.e., 10-17% during 1999-2001). The biology of this secretive species is relatively unknown, thus information on all aspects of the life history of hooded mergansers is needed.

Studies of incubation patterns provide insights into reproductive strategies and bioenergetics of nesting females. Generally, small-bodied waterfowl spend a greater proportion of incubation time in recesses than do larger waterfowl. Our unweighted data suggest that hooded mergansers spent approximately 81–84% of incubation time on the nest, which would be at the low end of values reported for species studied to date. Hooded mergansers in Ontario have been reported to incubate 85% of the time (Mallory et al. 1993), whereas blue-winged and green-winged teal (*Anas discors, A. cyanoptera*), which are approximately 35% smaller than hooded mergansers, have been reported to incubate 79–83% of the time (Afton and Paulus 1992).

Longer recess time is generally required for smaller-bodied hens to maintain body mass during incubation. Hooded mergansers on our study area have been reported to lose 5–6% of their body mass during incubation, which is among the lowest observed (Zicus 1997). Surprisingly, common goldeneyes, which are about 25% larger than hooded mergansers, lost approximately 11% of their body mass despite spending 80–85% of their time incubating (Zicus and Hennes 1995).

A striking aspect of nest attendance data is often the high variability within and among females (Zicus and Hennes 1995) making these data difficult to analyze. The adaptive consequences of incubation patterns can only be understood through the study of many species in different locations. Data from additional hooded merganser nests would markedly improve our sample.

ACKNOWLEDGMENTS

We'd like to thank Steve Caron for providing the nest boxes that we erected in the Paul Bunyan State Forest.

LITERATURE CITED

Afton, A. D., and S. L. Paulus. 1992. Incubation and brood care, p. 62-108. *In* B. D. J. Batt, A. D. Afton, M. G. Anderson, C. D. Ankney, D. H. Johnson, J. A. Kadlec, and G. L. Krapu [eds.], Ecology and management of breeding waterfowl. Univ. of Minnesota Press, Minneapolis.

Cooper, J. A., and A. D. Afton. 1981. A multiple sensor system for monitoring avian nesting behavior. Wilson Bulletin 93:325-333.

Mallory, M. L., H. G. Lumsden, and R. A. Walton. 1993. Nesting habits of hooded mergansers *Mergus cucullatus* in northeastern Ontario. Wildfowl 44:101-107.

Zicus, M. C. 1990. Nesting biology of hooded mergansers using nest boxes. Journal of Wildlife Management 54:637-643.

Zicus, M. C., and S. K. Hennes. 1995. Common goldeneye nest attendance patterns. Condor 97:461-472.

Zicus, M. C. 1997. Female hooded merganser body mass during incubation. Condor 90:220-224.

Table 1. Nest attendance data during 3 periods (laying, incubating, and hatching) for hooded mergansers nests in wooden nest boxes in northern Hubbard County MN, 2002-2003.

	Year					
	2002			2003		
Period	Laying	Incubating	Hatching	Laying	Incubating	Hatching
Number of nests	2	4	3	2	4	3
Days monitored[a]	18	67	6	17	59	5
Recesses/day[b]	na[c]	3.6	2.0	na[c]	3.0	3.0
Minutes/day on nest[b]	128.7	1209.3	980.0	200.9	1161.7	968.6
Minutes/day off nest[b]	1311.3	230.7	132.8	1239.1	278.3	324.0

[a]Days monitored on which the monitor did not malfunction and the hen was not flushed by the investigator.
[b]Means not adjusted for incubation day and hen effects.
[c]Not applicable.

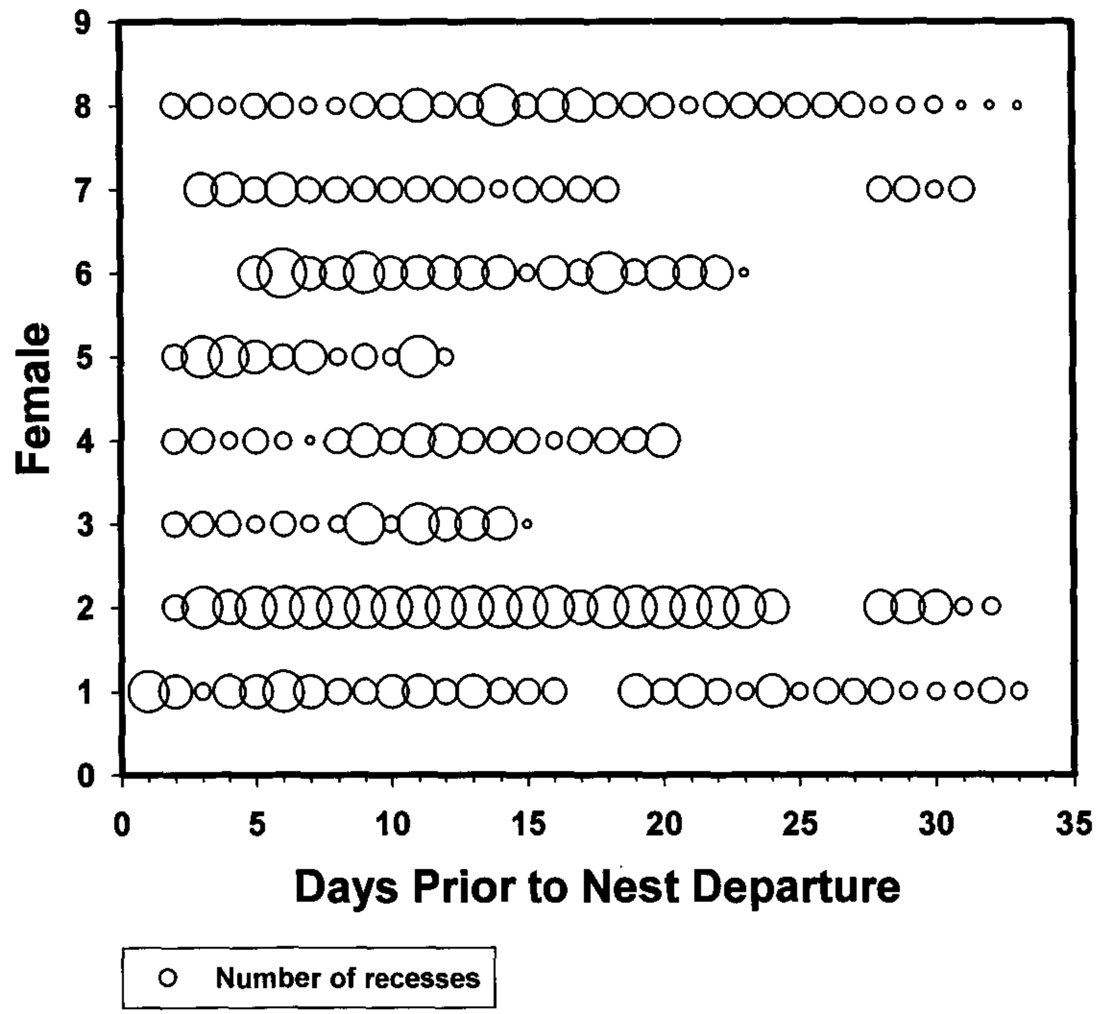


Figure 1. Bubble plots depicting the daily number of incubation recesses for hooded mergansers in north central Minnesota, 2002-2003. Circle size is proportional to the number of recesses, and absence of a circle indicates missing data.

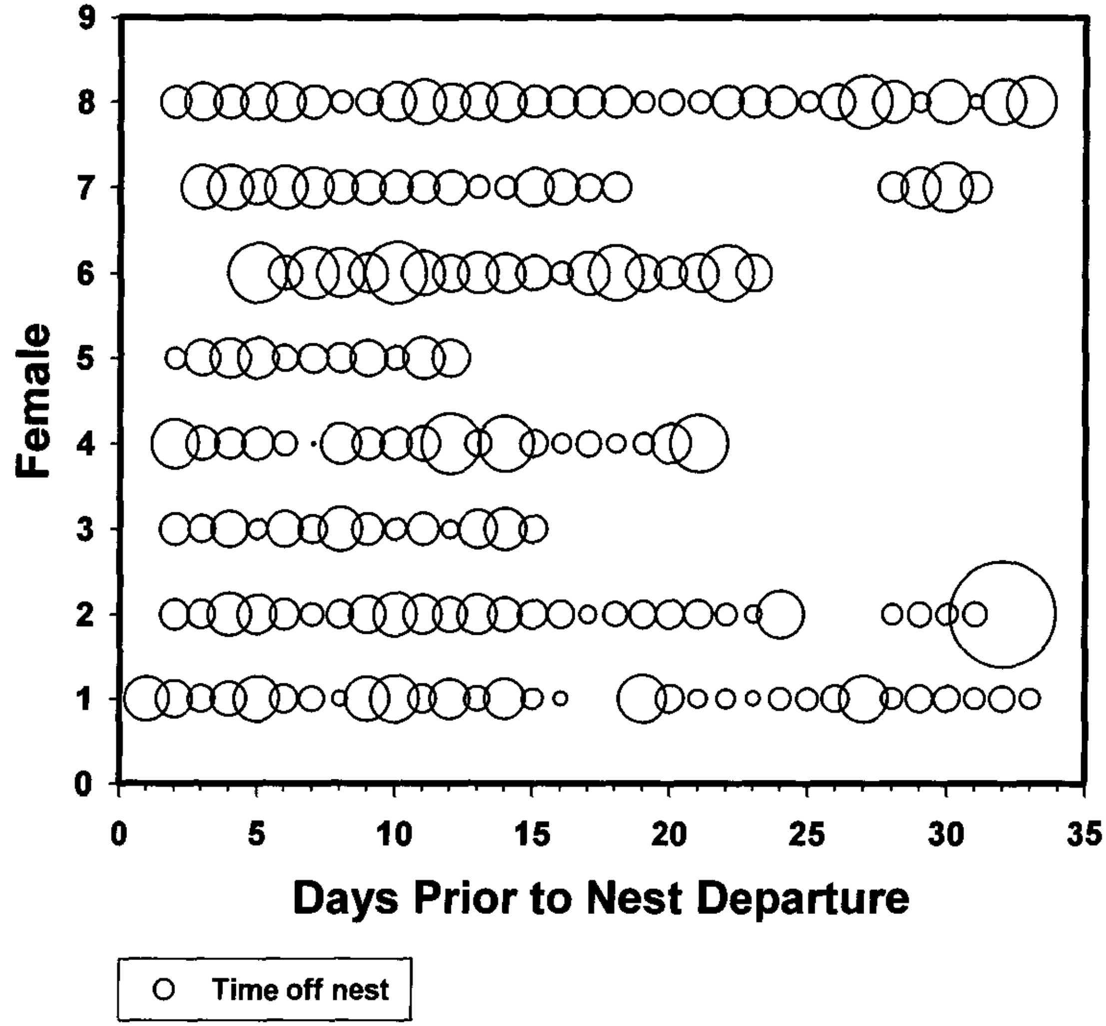


Figure 2. Bubble plots depicting the total time daily in incubation recesses for hooded mergansers in north central Minnesota, 2002-2003. Circle size is proportional to the amount of time, and absence of a circle indicates missing data.

PROPOSED RING-NECKED DUCK BREEDING PAIR SURVEY

Michael C. Zicus, David P. Rave, and Robert Wright

ABSTRACT

Ring-necked ducks (*Aythya collaris*) have been identified by the Minnesota Department of Natural Resources' *A Vision for Wildlife and its Use – Goals and Outcomes 2003 – 2013* 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 for the species are inadequate. We intend to survey breeding ring-necked ducks through Minnesota's primary ring-necked duck breeding range using a helicopter because visibility of ring-necked ducks from a fixed-wing airplane is poor in most ring-neck breeding habitats. MN-GAP habitat model data was used to quantify nesting cover in section-sized survey plots. Four survey strata have been defined based on the amount of ring-necked duck nesting cover included in each survey plot. For the pilot survey, we apportioned 200 section-sized survey plots between high and low habitat strata within 6 survey-area Ecological Classification System sections. Plots in a third habitat stratum will not be sampled because we believe the probability of ring-neck pairs occurring on these plots is very low, and plots in a fourth strata will not be sampled because we believe no breeding ring-necked ducks occur on these plots. We will assess the validity of these assumptions during the survey. Breeding pairs will be recorded by quarter section within the survey plots to evaluate the stratification approach we've used and assess the efficiency of different sized plots. Preliminary observations during the spring 2004 Canada goose survey suggest that it will be feasible to count ring-necked ducks on section-sized plots.

INTRODUCTION

Staff in the Minnesota Department of Natural Resources 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 identified by the Minnesota Department of Natural Resources' *A Vision for Wildlife and its Use – Goals and Outcomes 2003 – 2013* as an indicator species for the Forest Province.

Little is known about the current distribution and relative 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 while acknowledging "local" breeding to the Iowa border. In comparison, the Minnesota Department of Natural Resources' *Gap Analysis Project* (MN-GAP) defined ring-neck breeding range as including any Ecological Classification System (ECS) subsection where ring-necked duck reproduction had been documented or 87% of the state (Fig. 1).

Continentally, numbers of breeding ring-necks have been increasing, but this might not be the case in Minnesota (Fig. 2). Current Minnesota waterfowl breeding pair surveys are inadequate for monitoring resident ring-necked ducks. A *Ring-necked Duck Pair Survey* has been conducted in the Bemidji vicinity since 1969 by the Wetland Wildlife Populations and Research Group and includes lakes that were believed to be some of the best ring-necked duck lakes in north-central Minnesota when the survey was designed. However, the geographic extent of the survey is limited to the Bemidji vicinity. In contrast, the *May Breeding Pair Survey*, which is directed primarily at mallards and also conducted by the Wetland Wildlife Populations and Research Group, has a wider coverage but does not survey much of the northern and eastern portion of the ring-neck breeding range. Further, this survey is too early to provide useful information because ring-necked ducks arrive on breeding areas and begin nesting later than mallards (Hohman and Eberhardt 1998).

The Minnesota Department of Natural Resources' *Restoring Minnesota's Wetland and Waterfowl Hunting Heritage* acknowledged that a sizable population of Minnesota-breeding ducks is the cornerstone to improving fall duck use. Properly designed breeding population surveys are needed to monitor the status of all species of resident forest waterfowl. However, the biology of different forest species precludes the ability to survey all species with a single survey.

METHODS

The survey could be designed to provide 3 types of information: 1) a population estimate, 2) trend data, or 3) distributional data. An optimal operational survey would likely be designed slightly differently depending on the information of greatest interest. At this time, we have not decided which type of information we consider most important. However, this decision is not particularly important for the pilot survey. We will use a helicopter survey because visibility of ring-necked ducks from a fixed-wing airplane is poor in most ring-neck breeding habitats. Survey design is similar to that used for Minnesota's resident Canada geese where 150 sample plots are counted statewide for $12,000 – $15,000 (S. Maxson, J. Lawrence, pers. comm.). We decided that we could afford to sample 200 section-sized plots in the pilot survey because we are sampling a smaller fraction of the state and there will be less travel time among plots.

We will survey Minnesota's primary breeding range using ECS sections and amount of ring-necked duck nesting cover to stratify the sampling. We used the MN-GAP habitat model data to quantify nesting cover in Minnesota. We defined 4 survey strata based on the amount of ring-necked duck nesting cover included in each section-sized survey plot (Table 1). We combined sedge meadow and broadleaf sedge/cattail (types 14 and 15) into a single class because of concerns we have about classification accuracy in the MN-GAP data, and we will stratify on this combined habitat class.

Much is unknown regarding the usefulness of the MN-GAP data as a stratification variable or the most efficient survey plot size. For the pilot survey, we apportioned 200 plots among habitat strata 1 – 2 within the ECS sections in the proposed survey area. Plots were apportioned among ECS sections in proportion to the relative amount of sedge meadow and broadleaf sedge/cattail habitat within each ECS section. Within an ECS section, plots assigned to the section were apportioned between stratum 1 (high) and 2 (low) based on the proportion of potential survey plot in each stratum. Actual survey plots were selected randomly from the potential plots in each stratum.

Plots in habitat stratum 3 will not be sampled because we believe the probability of ring-neck pairs occurring on these plots is very low. Plots in stratum 4 also will not be sampled because we believe no breeding ring-necked ducks occur on these plots. We will assess the validity of these assumptions during the pilot survey. Breeding pairs will be recorded by quarter section within the survey plots. This strategy will allow us to evaluate the stratification approach we've used and assess the value of different sized plots.

The pilot survey will be conducted beginning the second week of June.

RESULTS

We restricted the pilot survey to an area that we consider Minnesota's primary ring-necked duck breeding range (Fig. 3). MN-GAP did not define the primary breeding range, but previous work (Moyle 1964, Hohman and Eberhardt 1998) indicated that it was restricted to northern Minnesota and did not extend much beyond the forest transition zone. Unpublished data from the U.S. Fish and Wildlife Service 4-square Mile Survey were used to confirm which ECS subsections in the MN-GAP breeding range likely represented peripheral areas. 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 pair/year during a 10-year period and if pairs were not counted on the plots in at least 5 of the 10 years. The Minnesota River Prairie subsection qualified as primary breeding range under these criteria, but it was excluded nonetheless. 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. We also removed the Boundary Waters Canoe Area and metropolitan counties from the survey area because of flight restrictions and other logistical considerations.

Preliminary observations during the spring 2004 Canada goose survey, where quarter-section-sized plots 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.

Survey plots were concentrated somewhat in the central and western parts of the survey area (Fig. 4). 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.

Amount of sedge meadow and broadleaf sedge/cattail cover associated with an open water area included in the sample plots was highly skewed (Fig. 5). Plots in the high cover stratum contained from 3.23 – 86.88 ha of cover while those in the low habitat strata contained 0.03 – 3.17 ha of ring-necked duck nesting habitat as defined from the MN-GAP data.

ACKNOWLEDGMENTS

We thank Brian Hargrave and Nancy Dietz for providing the initial MN-GAP data. Discussions with John Fieberg and John Giudice were helpful regarding survey design.

LITERATURE CITED

Hohman, W. L., and R. T. Eberhardt. 1998. Ring-necked duck (*Aythya collaris*). *In* The Birds of North America, No. 329, (A. Poole and F. Gill, eds.). The Birds of North America, Inc. Philadelphia, PA.

Moyle, J. B. (ed.). 1964. Waterfowl in Minnesota. Technical Bulletin No. 7. Minnesota Department of Conservation.

Table 1. Definition of ring-necked duck breeding pair survey strata.

Stratum	Definition
1	Survey plots that have ≥ the median amount (3.18 ha) of MN-GAP class 14 and 15 nesting cover.
2	Survey plots that have < the median amount (3.18 ha) of MN-GAP class 14 and 15 nesting cover.
3	Survey plots that have no MN-GAP class 14 and 15 nesting cover but that include open water that is <250 m from a shoreline.
4	Survey plots that have no MN-GAP class 14 and 15 nesting cover and that include no open water.

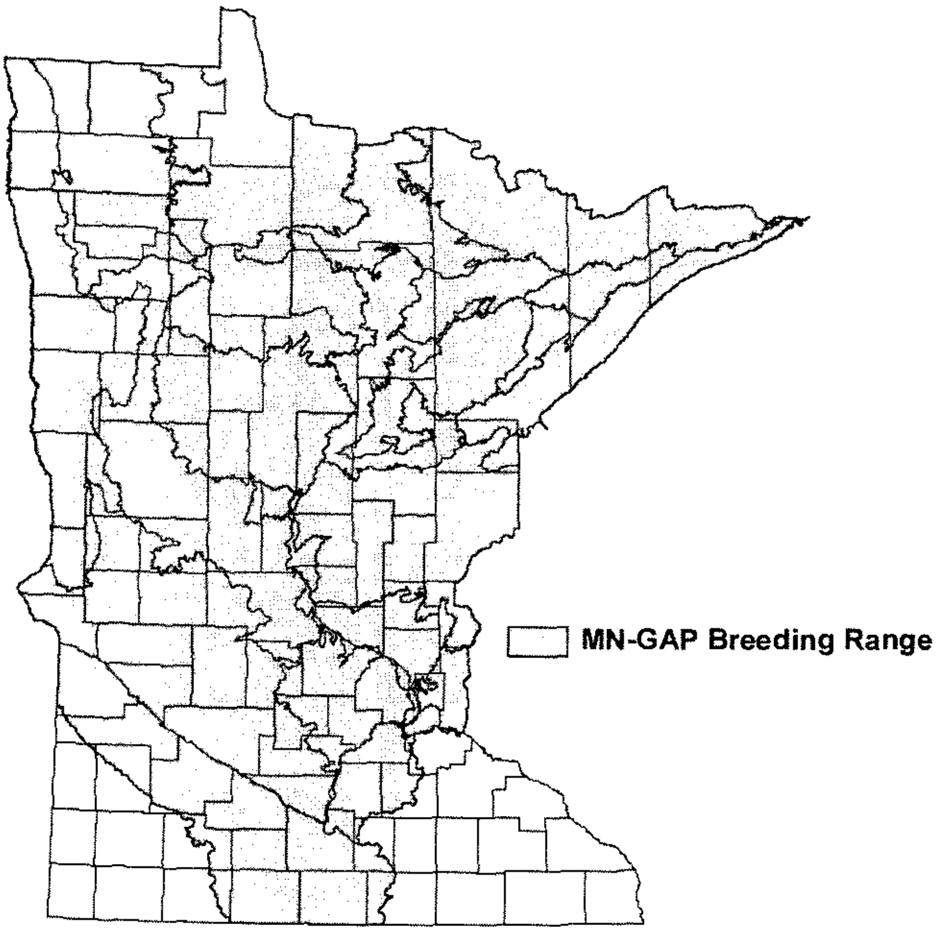


Figure 1. Minnesota ring-necked duck breeding range (Minnesota Department of Natural Resources' *Gap Analysis Project*).

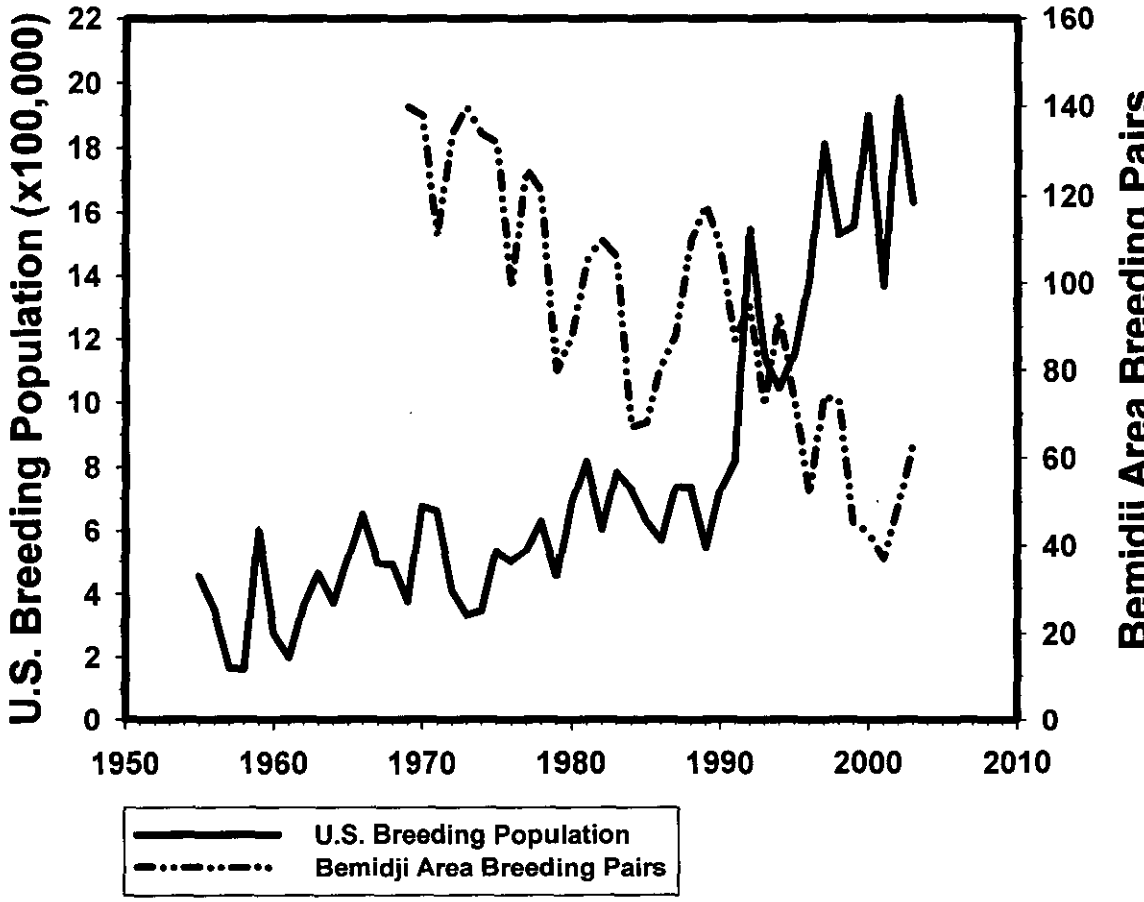


Figure 2. Ring-necked duck breeding population trends as reflected by the U.S. Fish and Wildlife *Breeding Pair Survey* and the Minnesota Department of Natural Resources' *Bemidji Area Ring-necked Duck Survey*.

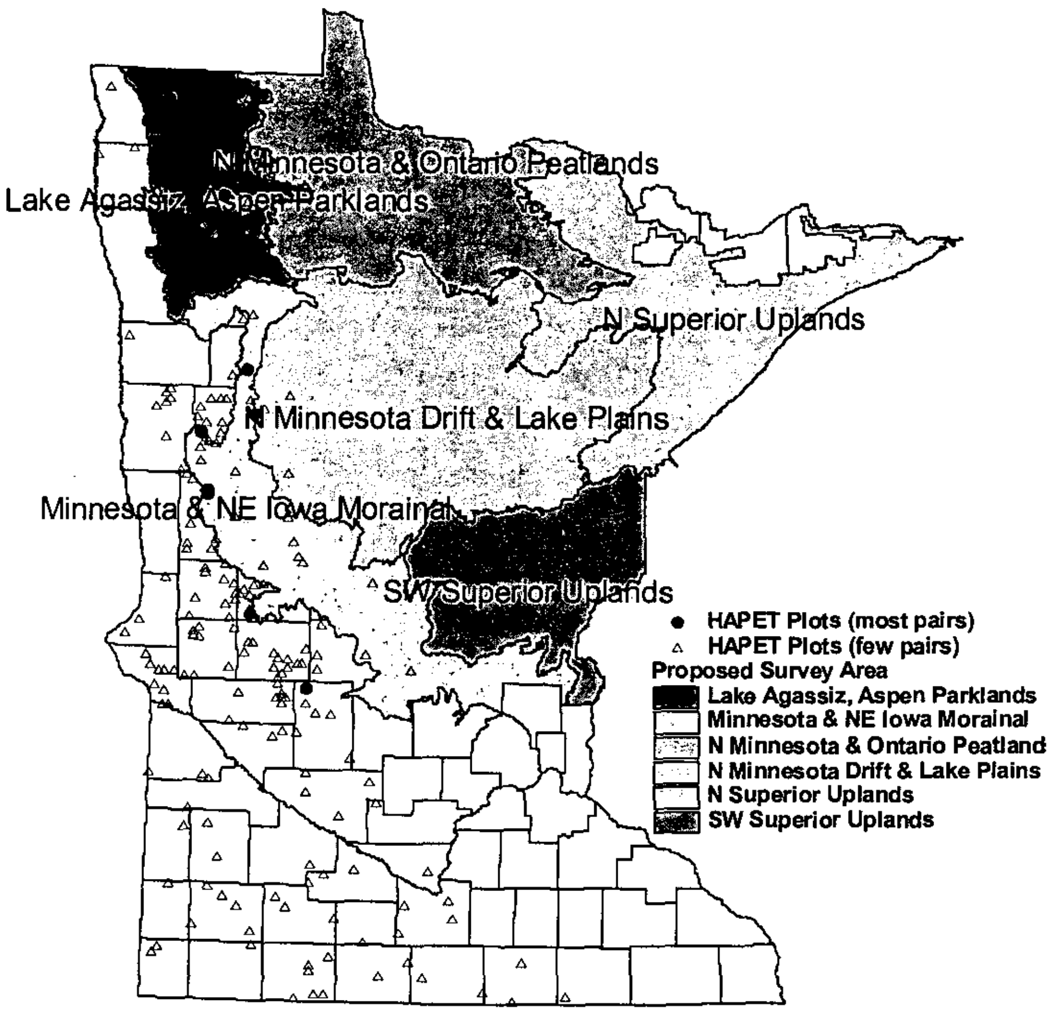


Figure 3. Proposed survey area for Minnesota ring-necked duck breeding pair survey.

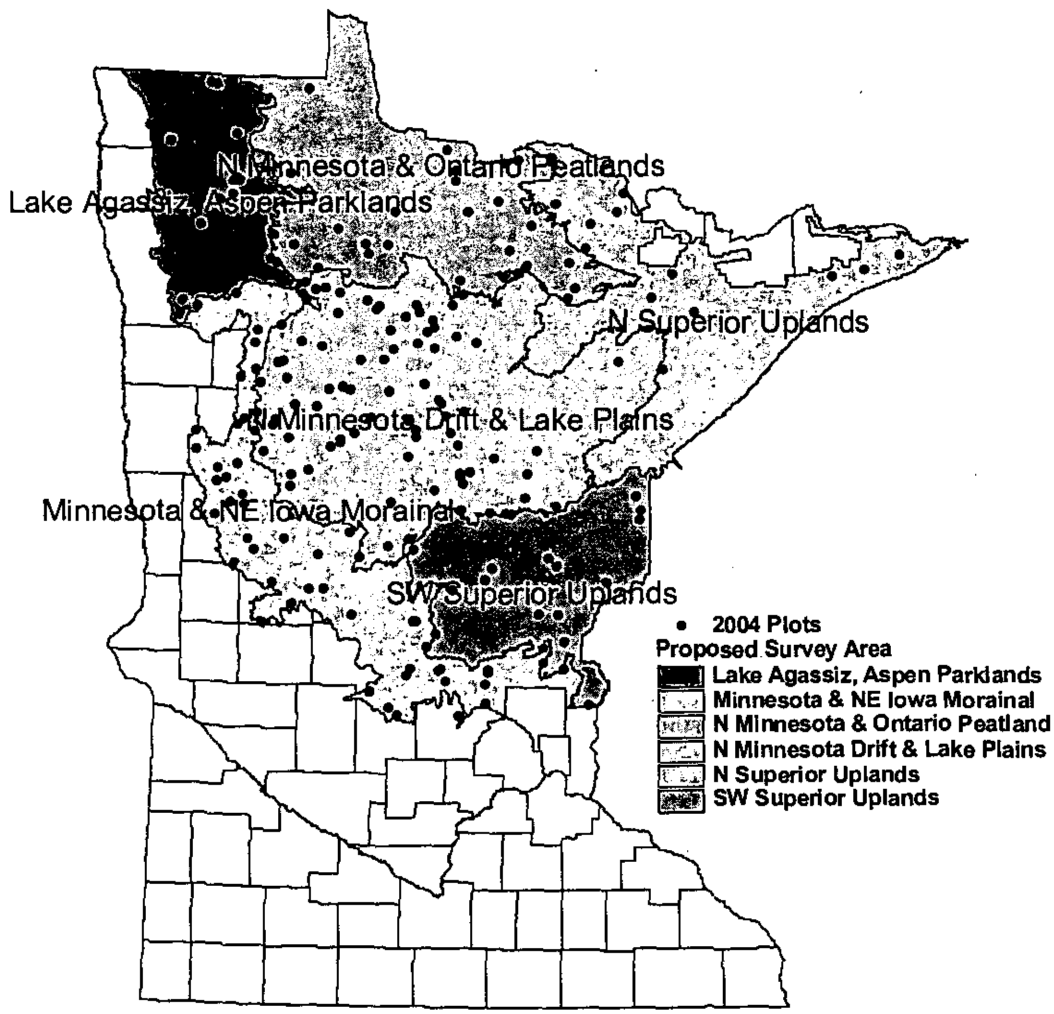


Figure 4. Plots selected for Minnesota ring-necked duck breeding pair survey, 2004.

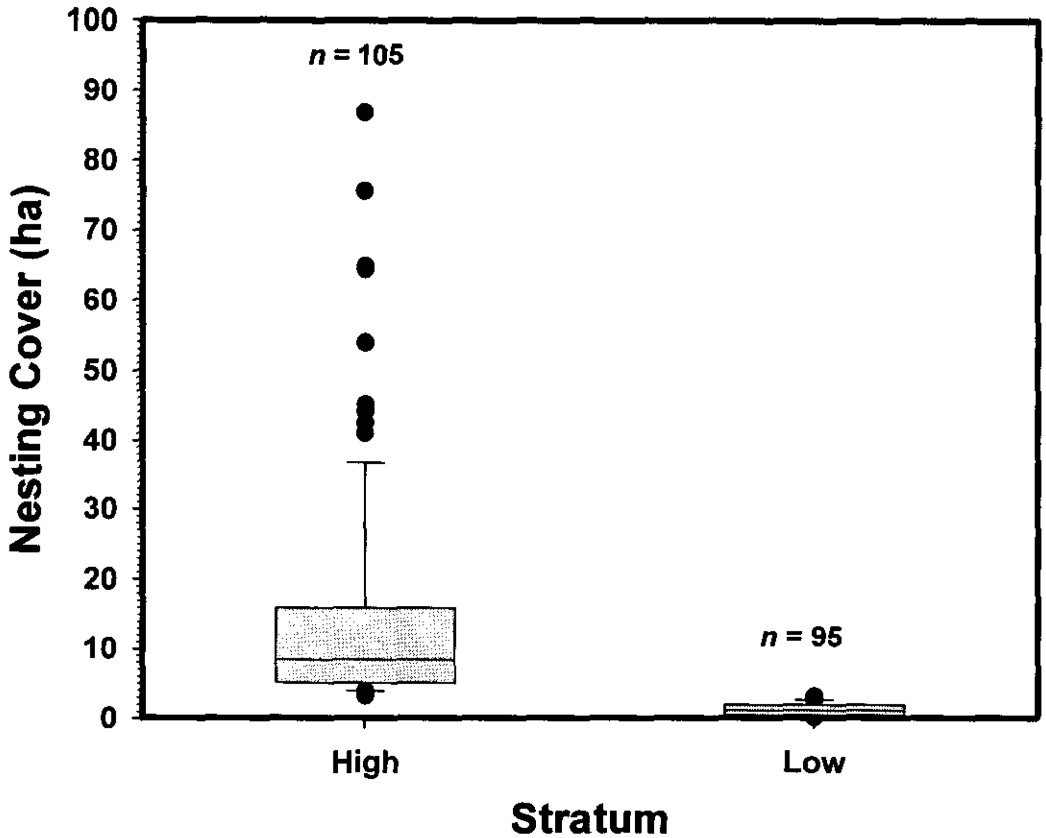


Figure 5. Box and whisker plots of the amount (ha) of nesting cover (sedge meadow and broadleaf sedge/cattail cover associated with open water) contained in the 200 plots to be sampled in the ring-necked duck pair survey in Minnesota's primary breeding range, 2004.

FACTORS INFLUENCING INCUBATION EGG MASS LOSS FOR THREE SPECIES OF WATERFOWL*

Michael C. Zicus, David P. Rave, and Michael R. Riggs[1]

Many bird eggs lose ~15% of their fresh mass before pipping, but individual species have been reported to lose 10–23%. Most published estimates have been imprecise due to small sample sizes. Moreover, published estimates of within- or among-species variance components of mass loss are virtually unknown. We modeled the influence of nest type, clutch size, and egg size on daily mass loss of Mallard (*Anas platyrhynchos*), Common Goldeneye (*Bucephala clangula*), and Hooded Merganser (*Lophodytes cucullatus*) eggs and compared fractional mass loss among species. This knowledge is essential to better understanding the adaptations of birds to their environments. Mallard eggs in artificial nest cylinders lost more mass than those in ground nests, but were unaffected by nest initiation date. Average-sized eggs in Mallard ground nests, Mallard cylinder nests, and Common Goldeneye and Hooded Merganser nests in nest boxes lost 7.9 g (15.2%), 10.8 g (20.3%), 10.3 g (15.5%), and 9.2 g (15.8%) of fresh mass, respectively. For all species, daily mass loss increased as incubation progressed and was affected by an interaction between egg size and incubation time, but was not influenced by clutch size. Depending on species, smallest eggs lost 1.0–4.0% more of their fresh mass than did the largest. Egg-mass variability was partitioned into years, nests within years, and eggs within nests and years. Variability was evenly distributed among the variance components in Mallard ground nests; however, among-eggs within-nest variance predominated in nests in nest cylinders. In contrast, among-nests variation was the dominant source for goldeneyes and mergansers. Nest-site selection and egg size likely involve tradeoffs among optimum egg-mass loss and nest and hatchling survival. *Received 2 September 2003, accepted 2 April 2004.*

*Abstract of paper in press in 2004 Condor.

---

[1]Statistical Research Division, Research Triangle Institute, 3040 Cornwallis Road, Research Triangle Park, NC 27709

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

Mark A. Hanson[1], James O. Church[2], Anthony T. Miller[2,3], Brian J. Palik[4], and Malcolm G. Butler[2]

INTRODUCTION

Seasonal wetlands (sensu Stewart and Kantrud 1971) are abundant in forested landscapes and support unique biological communities. Until recently, these sites were poorly understood and often overlooked by forest managers who were largely unaware of potential implications of timber harvest and related activities in wetlands and adjacent uplands. Seasonal wetlands are common in Minnesota's Laurentian Mixed Forest (Almendinger and Hanson 1998). 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 forested regions of Minnesota, seasonal wetlands usually fill during spring from surface runoff due to snow-melt, and then dry owing to evapotranspiration during 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 linkage to adjacent forested uplands. This is well illustrated by the fact that seasonal wetlands 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 inhibits sunlight available at the wetland surface. Relationships between light availability, primary production, and major vegetation forms are not well known, but canopy closure is a major influence on vegetation dominance in small wetlands. Removal of canopy via timber harvest has potential to influence biological communities in adjacent wetlands owing to increased sunlight, higher water temperatures, and reduced coarse woody debris and leaf litter inputs associated with these activities.

---

[1]Wetland Wildlife Populations & Research Group, Minnesota Department of Natural Resources, 102 23rd St. NE, Bemidji, MN, 56601.
[2]Department of Biological Sciences, Stevens Hall, North Dakota State University, Fargo, ND 58105.
[3]Present address: Third Rock Consultants, LLC, 2514 Regency Road, Suite 104, Lexington, KY, USA 40503.
[4]North Central Research Station, USDA Forest Service,1831 Highway 169 East, Grand Rapids, MN 55744.

Aquatic invertebrates are often the most abundant fauna in seasonal wetlands (Brooks 2000) and serve as important links between products of primary production and higher consumers (Murkin and Batt 1987). Various species of birds, amphibians, and small mammals depend on the aquatic invertebrates which seasonal wetlands provide. Aquatic invertebrate communities in these habitats exhibit life cycles constrained by needs to 1) minimize influences of desiccation, 2) reproduce rapidly, and 3) avoid extirpation due to vertebrate and invertebrate predators (Wellborn et al. 1996, Wiggins et al. 1980). In general, invertebrate species richness probably increases with hydroperiod length (Brooks 2004), but this is mitigated somewhat by complex influences of predation (Hanson et al. In Review). More broadly, 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, common 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 make no similar recommendations for many small, seasonally-flooded wetlands. This may be unfortunate given strength of functional linkages (Palik et al. 2001, Colburn 2001) between small wetlands and adjacent upland landscapes, at least at local spatial scales. However, guidelines encourage retention of five % cover in patches following clear-cut timber harvest and suggest that these "five % patches" should be focused adjacent to seasonal wetlands (Minnesota Forest Resources Council 1999). Whether these "five % suggestions" are logistically feasible or work to preserve ecological integrity of seasonal wetlands is largely unknown. Finally, some existing evidence supports the notion that timber harvest influences natural hydroperiods, at least for some wetland types (Dube and Plamondon. 1995, Roy et al. 1997). If this is the case, we expect consequences for resident invertebrate communities whose life-cycle strategies reflect narrow tolerances to influences of flooding, desiccation, and predation.

Research reported here was performed in association with investigators from USFS (North Central Research Station, Grand Rapids, MN), Natural Resources Research Institute (Duluth, MN), and University of Minnesota (St. Paul, MN). Collectively, this group is assessing efficacy of harvest buffer strips on ecological integrity of seasonal wetlands. Here, we summarize results of our research evaluating responses of wetland invertebrate communities to adjacent timber harvest and harvest buffers. Our project has several objectives. First, we used data collected during one pretreatment year (2000) to assess natural variability of resident invertebrate communities. Second, we identified and evaluated potential sources of variability in major taxa of aquatic invertebrates in study wetlands during the first three years following timber harvest. Finally, we performed a general test to assess the efficacy of harvest buffers by examining potential responses of wetland invertebrate communities to timber harvest across the four treatment groups. This is a partial summary based on preliminary analyses; results and interpretation may change as additional data are gathered and interpreted.

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 are 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 wherein 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 % 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.

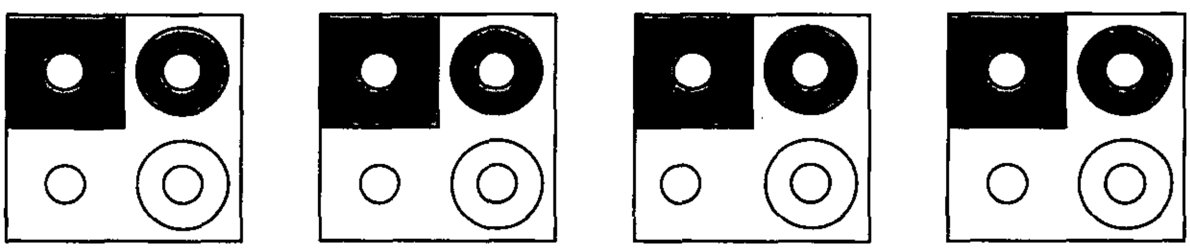


Figure 1. Drawings depict four experimental harvest/buffer configurations. Clockwise from upper left, these were control (no harvest), full buffer (no harvest within 50 feet of study wetlands), thinned buffer (50 percent thinning within buffer), and no buffer (clear-cut to wetland margins). Each group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

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 until five sets of samples had been gathered or until the initial wetland desiccation. 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).

Staff from North Central Forest Experiment Station (NCFES) provided physical and chemical data for use in our analyses. These data (hereafter environmental data) included hydroperiod, concentrations of major dissolved nutrients and other selected dissolved ions, alkalinity, specific conductivity, water transparency and color, and canopy coverage over study wetlands. For logistical reasons, not all parameters were assessed in all sites during all study years. Results presented here are based on data currently available to us.

Statistical analysis

Wetlands were considered the units of observation for all our analyses. We used indirect (principle components analysis, PCA) and direct (redundancy analysis, RDA) gradient analyses to assess variability in aquatic invertebrate communities of wetland study sites and to relate observed patterns to environmental gradients. PCA based on pre-treatment data (2000) was used to assess natural variability among invertebrate communities and to illustrate temporal changes in these populations. We used direct gradient analysis (ter Braak 1995) to identify environmental sources of variability in wetland invertebrate communities and to measure magnitude of these influences. RDA was used to identify environmental variables (via forward selection) that contributed significantly to observed patterns in aquatic invertebrates. Following the initial RDA, the total variance explained by all significant model terms was then partitioned to assess the relative contributions of the significant environmental variables (Borcard et al. 1992). Construction of RDA models and variance partitioning were done using CANOCO 4.0 (Ter Braak and Smilauer 1998).

Finally, we used discriminant analysis (DA) and multi-response permutation procedures (MRPP) (McCune and Grace 2002) to evaluate whether wetland invertebrate communities were influenced by adjacent timber harvest, thus indicating if residual buffers were mitigating against influences of these harvests. Following DA and MRPP, we used indicator species analysis (Dufrene and Legendre 1997) to identify potential relationships between individual invertebrate taxa and buffer treatments. Our indicator species analysis was conducted using 1000 permutations. For each wetland, we summed numbers of invertebrates captured in five SATs to produce totals of major taxa collected during each biweekly sampling. These totals were averaged annually, resulting in mean number of organisms per wetland during each study year. Thus, in general, our analyses were based on 64 wetland-year combinations (16 wetlands sampled during 4 years) of major invertebrate taxa and environmental data provided by NCFES. DA was conducted using SAS JMP 5.0.1a; MRPP was performed using PC-ORD version 4.0 (McCune and Mefford 1999).

RESULTS

Pre-treatment variation

Invertebrate communities in our study wetlands were dominated by Culicidae, Dytiscidae, Hydrophilidae, *Daphnia* spp., *Eubranchipus* sp., Ostracoda, Copepoda, and Chironomidae. Wetland invertebrate communities varied dramatically among sampling dates during 2000 (Figure 2). PCA of these four sampling dates during 2000 illustrated

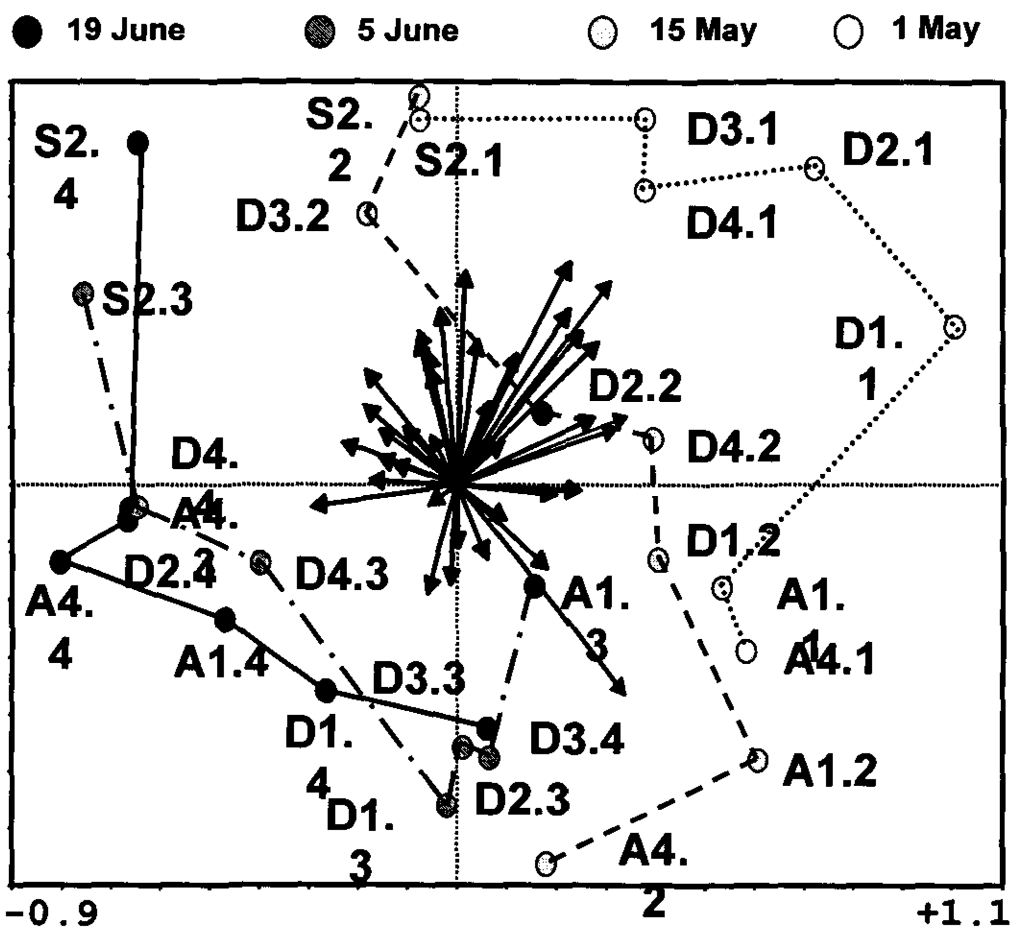


Figure 2. Principle components analysis depicting invertebrate community characteristics in 7 study wetlands sampled during period of 1 May – 19 June 2000 (pre-harvest). Wetland study sites connected with the same line were sampled on the same date. Other wetlands were sampled during 2000, but only these 7 sites were sampled on all four 4 dates during the period indicated above. Changing position (wetland site scores) over time indicated change in numbers and composition of invertebrates. Arrows indicate direction of association with invertebrate taxa (not shown).

variability in wetland site scores through an annual growing season. Very generally, wetlands were dominated by Diptera (mosquitos) and, sometimes, by Anostraca (fairy shrimp, *Eubranchipus*) on earlier sampling dates. We observed a transition toward zooplankton, Hemiptera, and Coleoptera later in the growing season.

We were able to gather invertebrate data from all 16 wetlands during only one sampling period in 2000 (week of 15 May). Environmental variables which were significantly associated with invertebrate community structure were hydroperiod and water-column nitrate concentrations. Hydroperiod and nitrate explained 34 and 24 percent of the variance in invertebrate communities, respectively (Figure 3a).

Post-treatment Patterns

Hydroperiod was the only environmental variable among those we assessed that showed a consistent association with invertebrate community characteristics (in other words, only hydroperiod explained significance variance in invertebrates during each year of our study). For example, results of RDA from 13 May 2001 indicated that hydroperiod and light absorbance (attenuation) were significantly associated with invertebrate community composition. Variance partitioning here indicated that hydroperiod and water-column light absorbance (attenuation) explained 45 and 20 percent of the variance respectively (Figure 3b). RDA results from 2002 indicated that the invertebrate community was constrained significantly by hydroperiod and total nitrogen. Here, wetland hydroperiod and nitrogen explained 21 and 13 percent of the variance, respectively (Figure 3c).

As expected, we collected Anostraca (*Eubranchipus*) from many of our study wetlands, but their occurrences were sporadic and short-lived. We saw some evidence that variability in life-cycle chronology (or abundance) of *Eubranchipus* was associated with buffer treatments (Figure 4). This may have been related to apparent changes in ice-out chronology also observed during 2001, the only year when ice-out dates were recorded for these study sites (Figure 5).

Community Response to Treatment

Discrimininant analysis (DA) showed moderate separation among harvest treatments (Figure 6). Results of our means separation tests (the MRPP) indicated that there was a significant timber harvest "effect" and also reflected extremely high variation within treatments ($t=-3.188$, $p=0.004$, $R=0.022$). Orthogonal comparisons among treatments (MRPP) indicated that wetlands within clear-cut treatments were distinguishable ($p<0.05$) from the other three treatment groups. Invertebrate communities in full buffer wetlands were only marginally different from those in the partial buffer sites, and neither full nor partial buffer sites differed distinctly from controls (Table 1). DA models misclassified 16 of the 64 site-year combinations, likely due to extreme variability in these communities. Five control wetlands were misclassified as full buffer, and four were misclassified as partial buffer. Three full buffer wetlands were misclassified, one as a clear-cut site and two as partial buffers. Two partial buffer wetlands were misclassified, one as a control and the other as a full buffer. One clear-cut wetland was misclassified as a control. Our indicator species analysis showed that Hemiptera were positively associated with clear-cut sites and *Eubranchipus* sp. was positively associated with the full buffer treatment (Table 2). This simply means that hemipterans and *Eubranchipus* occurred more often than would be expected by chance in wetlands associated with clear-cuts and control sites, respectively.

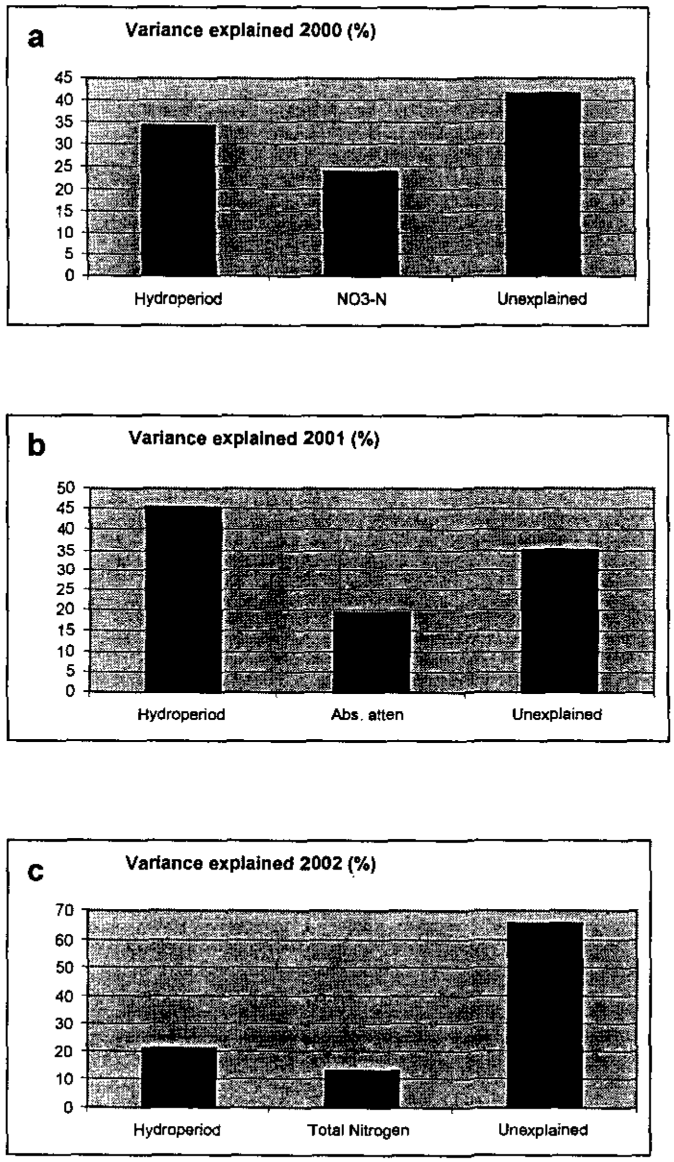


Figure 3. Results of variance partitioning using partial redundancy analysis (details explained in text). Histograms depict proportions of variance in aquatic invertebrate communities associated with significant environmental variables measured at each wetland (selected using forward selection procedures). Values presented here are based on mean numbers of aquatic invertebrates collected in all 16 study wetlands during 2000 (a), 2001 (b), and 2002 (c).

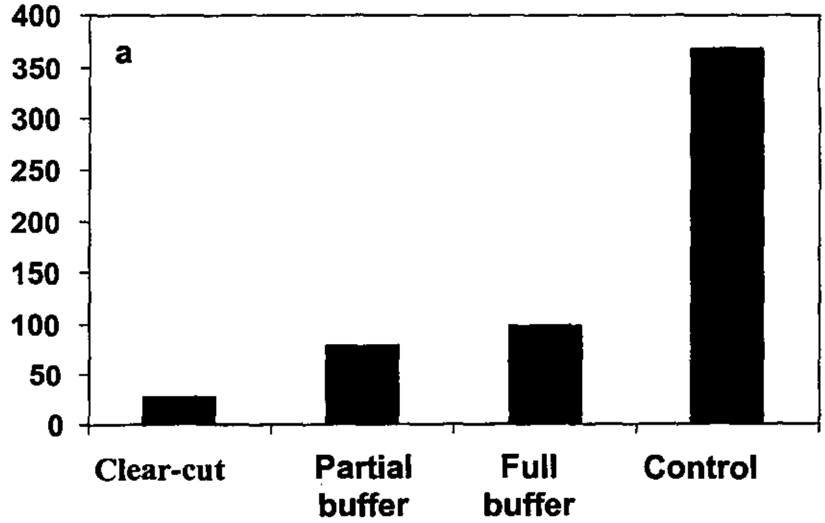


Figure 4. Mean annual number of *Eubranchipus* sp. sampled per wetland in the four treatments during 2001, one year following timber harvest. Variability was extremely high among sites, thus differences were not statistically significant.

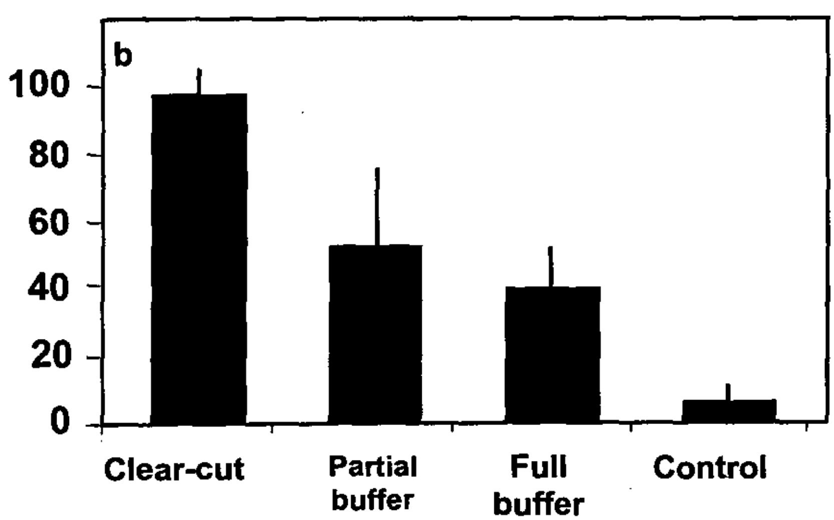


Figure 5. Mean percent open water by harvest/buffer treatments as estimated on 10 April 2001.

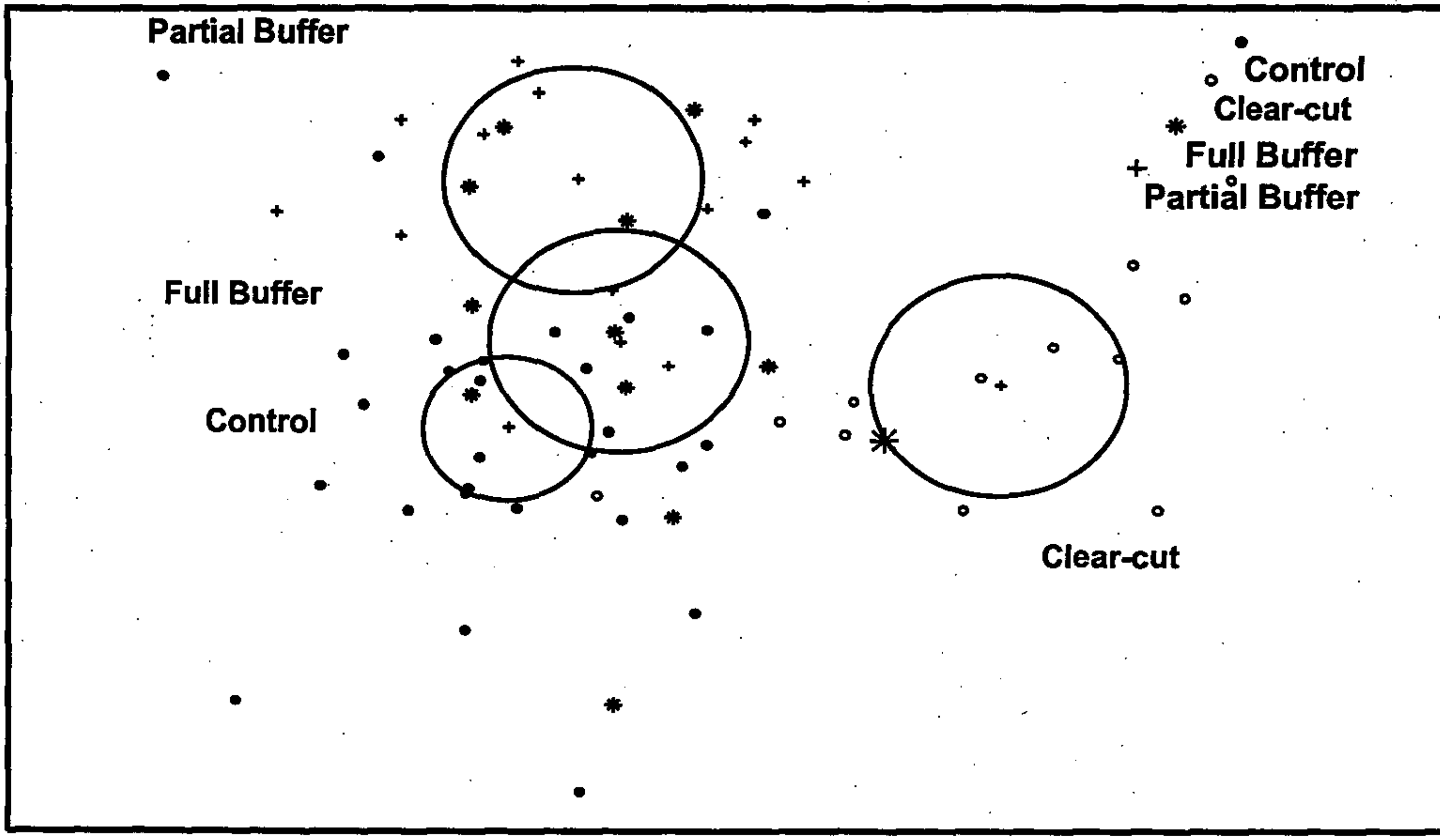


Figure 6. Canonical plot from discriminant analysis (details explained in text). Data points represent invertebrate community scores in canonical space, one score derived for each wetland during each study year. Circles represent buffer/harvest treatments, bounded by 95 percent confidence intervals. Crosshairs in the middle of each circle represent treatment means.

Table 1. Results from the full model MRPP and subsequent multiple comparisons. Values in bold indicate comparisons which were statistically significant.

Comparison	Chance corrected within group agreement	p-value
Full model MRPP	0.021	**0.005**
Control vs. Full Buffer	0.007	0.140
Control vs. Partial Buffer	0.004	0.229
Control vs. Clear-cut	0.018	**0.011**
Full buffer vs. Partial Buffer	0.021	0.057
Full buffer vs. Clear-cut	0.024	**0.030**
Partial buffer vs. Clear-cut	0.035	**0.006**

Table 2. Results of the indicator species analysis. Indicator values and p-values are given for each taxonomic group. Indicator values indicate % perfect indication (always present, exclusive to that group), based on combining the values of relative abundance and relative frequency.

Taxon	Control	Partial Buffer	Clear-cut	Full Buffer	p-value
Diptera	25	24	24	27	0.435
Odonata	14	8	18	17	0.837
Trichoptera	13	10	17	18	0.85
Hydracarina	26	23	25	26	0.661
Collembola	21	29	25	23	0.318
***Eubranchipus* sp.**	**10**	**7**	**11**	**35**	**0.019**
Conchostraca	9	25	2	26	0.259
Hirudinea	14	18	10	28	0.202
Oligochaeta	13	31	15	17	0.119
Coleoptera	24	25	25	26	0.627
Hemiptera	**13**	**14**	**36**	**22**	**0.01**
Ostracoda	23	27	26	25	0.525
Cladocera	25	20	28	28	0.325
Copepoda	24	24	26	26	0.393
Gastropoda	22	19	25	27	0.657
Sphaeridae	25	25	13	25	0.746

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 within individual wetlands sometimes exceeded spatial differences among similar sites on a given date. Our results indicated that few environmental variables measured in our study affected pre- and post-treatment invertebrate communities. With the exception of hydroperiod, influences of environmental variables appeared to be rather weak and fluctuated across study years. Our results also identified moderate influences of nitrogen ($NO_3^{-2,}$ 2000; TN, 2002) and this may reflect roles of organic matter inputs during these years. Overall weak correspondence between invertebrate communities and environmental variables may be due to the fact that invertebrate communities in seasonal wetlands are comprised of organisms with environmental tolerances well within the ranges of those measured in our study. This seems especially likely given that many invertebrates inhabiting freshwater wetlands are known to be well adapted to survival in ephemeral habitats subject to severe environmental fluctuations (Batzer et al. 2004, Euliss et al. 1999, Wiggins et al. 1980). Our RDA's also indicated that a large proportion of variance in these invertebrate communities remains unaccounted for by environmental characteristics of wetlands measured in our study. The latter may reflect the fact that key environmental variables were not included in our analyses, or that environmental extremes sufficient to limit invertebrate populations were not realized during the first several years of this research.

Clear-cut timber harvest resulted in distinguishable, community-level responses of aquatic invertebrates in adjacent study wetlands. Only two invertebrate taxa we reported (Hemiptera and *Eubranchipus*) showed significant associations with specific harvest/buffer treatments. Thus, treatment effects we observed may reflect subtle associations among buffers among a suite of invertebrates rather than sharp increases or decreases in abundance of a few taxa. Results of our DA indicated that full and partial buffers mitigated against effects of the adjacent timber harvest. Although preliminary, these data may indicate that harvest buffers have potential to preserve integrity of invertebrate communities in adjacent wetlands, such as those reported on here. We are unaware of other research specifically addressing efficacy of harvest buffers in Minnesota. However, these results support the notion that focusing residual trees (five % leave trees) adjacent to wetlands following clear-cut timber harvest (Minnesota Forest Resources Council 1999) may help sustain ecological continuity of the wetland component of forested landscapes.

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 may have influenced associated 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*. However, data useful for clarifying these and other influences were not available for our analysis.

Hydroperiod is a key determinant of invertebrate community structure in freshwater wetlands (Wiggins et al. 1980, Brooks 2000, Schneider and Frost 1996, Batzer et al. 2004, Bilton et al. 2001). Wetlands of longer duration often support higher taxonomic diversity due to prolonged habitat availability, although diversity is limited by influences of invertebrate and vertebrate predation (Welborn et al. 1996). Previous investigators have reported that timber harvest influences natural hydroperiods, at least for some wetland types (Dube and Plamondon. 1995, Roy et al. 1997, Verry 1986). We are aware of no research consensus linking timber harvest to hydrologic relationships of depressional wetlands in MN. However, we would certainly expect that modification of wetland hydroperiods would be associated with subsequent changes in resident invertebrate communities.

Conclusion

Preliminary results of this study support the suggestion of Palik et al. 1999 that seasonal wetland ecosystems 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 and showed weak correspondence with environmental features of the wetlands and the adjacent landscape. Forested buffers appeared to mitigate 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 assess causal mechanisms associated with results reported here and to evaluate interactions among wetland communities, harvest buffers, and timber harvest over longer time frames.

Acknowledgements

We thank the staff from North Central Forest Experiment Station (USFS), Minnesota Department of Natural Resources (MDNR) Wetland Wildlife Population and Research Group, MDNR Nongame Wildlife Program, Cass County Land Department, Potlatch Corporation, North Dakota Water Resources Research Institute and North Dakota State University Department of Biological Sciences for financial and logistical support of this work.

Literature Cited

Adamus, P. R. 1996. Bioindicators for assessing ecological integrity of prairie wetlands. U.S. Environmental Protection Agency, National Health and Environmental Effects Research laboratory, Western Ecology Division, Corvallis, OR, USA. EPA/600/R-96/082.

Almendinger, J. C., and D. S. Hanson. 1998. Ecological land classification handbook for the northern Minnesota Drift & Lake Plains and the Chippewa National Forest. Minnesota Department of Natural Resources, Division of Forestry, St. Paul, MN.

Batzer, D. P., C. R. Jackson, and M. Mosner. 2000. Influences of riparian logging on plants and invertebrates in small, depressional wetlands of Georgia, U.S.A. Hydrobiologia 441:123-132.

Batzer, D. P., B. J. Palik, and R. Buech. 2004. Relationships between environmental characteristics and macroinvertebrate communities in seasonal woodland ponds of Minnesota. Journal of the North American Benthological Society 23:50-68.

Borcard, D., P. Legendre, and P. Drapeau. 1992. Partialling out the spatial component of ecological variation. Ecology 73:1045-1055.

Braccia, A., and D. P. Batzer. 2001. Invertebrates associated with woody debris in a southeastern U.S. forested floodplain wetland. Wetlands 21:18-31.

Bilton, D. T., A. Foggo, and S. D. Rundle. 2001. Size, permanence and the proportion of predators in ponds. Arch. Hydrobiol. 151:451-458.

Brooks, R. T. 2000. Annual and seasonal variation and the effects of hydroperiod on benthic macroinvertebrates of seasonal forest ("vernal") ponds in central Massachusetts, USA. Wetlands 20:707-715.

Brooks, R. T. 2004. Weather-related effects on woodland vernal pool hydrology and hydroperiod. Wetlands 24:104-114.

Colburn, B. A. 2001. Small pools close up: examining vernal of the Northeast. Jan-Feb 2001:7-18.

Dube, S., A. P. Plamondon, and R. L. Rothwell. 1995. Watering up after clear-cutting on forested wetlands of the St. Lawrence lowland. Water Resources Research 31:1741-1850.

Dufrene, M. and P. Legendre 1997. Species assemblages and indicator species definition: the need of an asymmetrical and flexible approach. Ecological Monographs 67:345-366.

Euliss N. H., D. A. Wrubleski, and D. M. Mushet. 1999. Wetlands of the Prairie Pothole Region: invertebrate species composition, ecology, and management. Pages 471-514 in D. P. Batzer, R. D. Rader, and S. A. Wissinger (eds.). Invertebrates in freshwater wetlands of North America: Ecology and Management. John Wiley and Sons, New York.

Hanson, M. A., C. C. Roy, N. H. Euliss, Jr., K. D. Zimmer, M. R. Riggs, and M. G. Butler. 2000. A surface-associated activity trap for capturing water-surface and aquatic invertebrates in wetlands. Wetlands 20:205-212.

McCune, B., and J. B. Grace. Analysis of Ecological Communities. MjM Software Design, Gleneden Beach, OR.

McCune, B., and M. J. Mefford. 1997. Multivariate analysis of ecological data. Version 3.0 MjM Software, Gleneden Beach, OR.

Merritt, R. W., and K. W. Cummins. 1996. Aquatic insects of North America. 3rd ed. R.W. Merritt and K.W. Cummins (eds.) Kendal/Hunt Pub., Dubuque, IA.

Minnesota Forest Resources Council. 1999. Sustaining Minnesota Forest Resources: Voluntary Site-Level Forest Management Guidelines for Landowners, Loggers and Resource Managers. Minnesota Forest Resources Council, St. Paul, Minnesota.

Murkin, H. R., and B. D. Batt. 1987. The interactions of vertebrates and invertebrates in peatlands and marshes. Mem. Ent. Soc. Can. 140:15-30.

Oertli, B. 1993. Leaf litter processing and energy flow through macroinvertebrates in a woodland pond (Switzerland). Oecologia 96:46-47.

Palik, B., D. P. Batzer, R. Buech, D. Nichols, K. Cease, L. Egeland, and D. E. Streblow. 2001. Seasonal pond characteristics across a chronosequence of adjacent forest ages in Northern Minnesota, USA. Wetlands 21:532-542.

Pennak, R. W. 1989. Fresh-water invertebrates of the United States: Protozoa to Mollusca. 3rd ed. John Wiley and Sons, New York.

Resh, J. H., and J. H. Jackson. 1993. Rapid assessment approaches to biomonitoring using benthic macroinvertebrates. *In*: Rosenberg and V. H. Resh (eds.). Freshwater biomonitoring and benthic macroinvertebrates. Chapman and Hall, New York, NY.

Roy, V., J. K. Jeglum, and A/ P. Plamondon. 1997. Water table fluctuations following clearcutting and thinning on Wally Creek wetlands. P. 239-251 *In* Trettin, C. C. et al. (eds.) Northern Forested Wetlands: Ecology and Management. CRC Press Inc., Lewis Publishers, New York, USA.

Schneider, D. W., and T. M. Frost. 1996. Habitat duration and community structure in temporary ponds. Journal of the North American Benthological Society 15:64-86.

Stewart, R., and H. A. Kantrud. 1971. Classification of natural ponds and lakes in the glaciated prairie region. U. S. Fish and Wildlife Service Resource Publication 92. U. S. Government Printing Office, Washington, DC, USA.

Tangen, B. A., M. G. Butler, and M. J. Ell. 2003. Weak correspondence between macroinvertebrate assemblages and land use in the Prairie Pothole Region wetlands, USA. Wetlands 23:104-115.

Ter Braak, C. J., and P. Smilauer. 1998. Canoco Reference and User's Guide to Canoco for Windows: Software for Canonical Community Ordination (version 4). Microcomputer Power, Ithaca, NY, USA.

Thorp, J. H., and A.P. Covich. 1991. Ecology and classification of North American freshwater invertebrates. Academic Press, New York.

Verry, W. S. 1986. Forest harvesting and water: the Lakes States experience. Water Resources Research Bulletin 22:1039-1047.

Wellborn, G. A., D. K. Skelly, and E. E. Werner. 1996. Annual Review of Ecology and Systematics 27:337-363.

Wiggins, G. B., R. J. Mackay, and I. M. Smith. 1980. Evolutionary and ecological strategies of animals in annual temporary pools. Arch. Hydrobiol. Supplement 58:97-207.

WALLEYE STOCKING AS A TOOL TO SUPPRESS FATHEAD MINNOWS AND IMPROVE HABITAT QUALITY IN SEMIPERMANENT AND PERMANENT WETLANDS IN THE PRAIRIE POTHOLE REGION OF MINNESOTA

Brian R. Herwig[1], Mark A. Hanson[2], Jeffrey R. Reed[3], Bradford G. Parsons[3], Anthony J. Potthoff[4], Matthew C. Ward[5], Kyle D. Zimmer[6], Malcolm G. Butler[4], David W. Willis[5], and Vaughn A. Snook[3]

In the absence of extreme winterkill or predators, fathead minnows *Pimephales promelas* often reach very high densities, and influence both aquatic invertebrate abundance and community structure in Minnesota's prairie pothole wetlands. Furthermore, fathead minnows reduce herbivorous zooplankton, likely reducing water transparency and contributing to loss of aquatic vegetation in many wetlands. We stocked walleye (*Sander vitreus*) in wetlands in western Minnesota's Prairie Pothole Region to test the efficacy of "biomanipulation" as a tool to 1) suppress fathead minnow populations, 2) enhance invertebrate communities, and 3) improve wetland quality by inducing potential shifts from turbid, phytoplankton-dominance to clear water, macrophyte-dominance.

We evaluated two distinct treatments using either larval (fry) walleye, or age-1 and older (advanced) walleye. We added larval walleye to 6 wetlands (walleye fry treatment) and age-1 and older walleye to 6 additional wetlands (advanced walleye treatment) in May 2001 and 2002. Six other wetlands served as control or reference sites during both study years (reference treatment).

We observed significant reductions in abundance of fathead minnows in our walleye fry treatment during 2001 and 2002. This included larval fathead minnows, and was likely due to a combination of direct predation on minnows and decreased recruitment by adult fish. Concurrently, higher abundances of *Daphnia* spp. and several macroinvertebrate taxa, including amphipods, were also observed. Phytoplankton abundance and turbidity decreased, but not until the second year (2002). In 2001, we did not observe a decrease in phytoplankton, despite a pronounced increase in large cladocerans later during the growing season.

---

[1]Minnesota Department of Natural Resources, Division of Fish and Wildlife, 2114 Bemidji Avenue, Bemidji, MN 56601. Phone: (218)-755-4628. Email: Brian.Herwig@dnr.state.mn.us.
[2]Minnesota Department of Natural Resources, Wetland Wildlife Populations and Research Group, 102 23rd Street, Bemidji, MN 56601. Phone: (218)-755-3920. Email: Mark.Hanson@dnr.state.mn.us.
[3]Minnesota Department of Natural Resources, Division of Fish and Wildlife, 23070 North Lakeshore Drive, Glenwood, MN 56334.
[4]Department of Zoology, North Dakota State University, Stevens Hall, Fargo, ND 58105.
[5]Wildlife and Fisheries Sciences, South Dakota State University, Box 2140B, NPB Room 138, Brookings, SD 57007.
[6]Department of Biology, University of St. Thomas, Mai l# OWS 390, 2115 Summit Avenue, St. Paul, MN 55105.

In 2002, both phytoplankton abundance and turbidity decreased, perhaps indicating that presence of a robust zooplankton population during early summer was critical for the switch to a clear water state. Submerged aquatic vegetation showed modest, continued improvement in the walleye fry treatment throughout the study, although this response developed rather gradually following the initial manipulation. Zooplankton populations, and water transparency remained high in 2003 despite the reestablishment of fathead minnow populations in most wetlands. We hypothesize that this is due to the stabilizing influence of submerged aquatic vegetation in maintaining the clear water conditions.

Positive responses of major invertebrates persisted in the walleye fry treatment, despite dietary dependence of walleye on invertebrate prey following elimination of prey fish (fathead minnows). One major group of benthic invertebrates that may have been affected by walleye fry was Chironomidae. Although the trend was not significant, chironomids appeared to be less abundant in the walleye fry treatment in 2002 relative to the other treatments.

Overall, we observed very limited wetland responses to our advanced walleye treatment, except for some modest suppression of fathead minnows in 2002. Increases in amphipods and benthic chironomids were also evident in 2002, but we saw no increases in other invertebrate taxa.

Our results indicate that biomanipulation via walleye fry stocking may be useful to suppress fathead minnow populations and improve habitat quality in Minnesota wetlands, at least over short time periods. To facilitate persistent clear water and macrophyte-dominated conditions, we recommend that fry stocking be conducted at least every other year to control fathead minnows where they persist. Fathead minnows have prolific recruitment potential and exhibit strong predation pressure on invertebrates, thus sites with persistent minnow populations may require continued treatments to sustain clear water. We believe this is warranted given persistence of fathead minnow populations in prairie wetlands and the limnological basis for recovery to a "clear water state" in these habitats. Before and after all walleye stocking, wetlands should be monitored to allow managers to assess effectiveness of the biomanipulation efforts and to ensure that walleyes are not added to fishless wetlands.

We emphasize that we are not advocating establishment of permanent walleye populations in prairie marshes. To minimize predation on invertebrates, we recommend that walleye fingerlings be removed during fall. We also recommend that walleye fry not be stocked in wetlands with intermittent surface-water connections. Efforts should be made to select sites that are isolated from other surface waters because the success of biomanipulation efforts will be lower in systems where flooding and fish immigration frequently occur. Stocking of advanced walleye (age-1 and older) was not successful in producing desired food web responses in our study wetlands. Here, consumption rates were simply not high enough to reduce fathead minnow abundance. We acknowledge that our stocking rate was low compared to other biomanipulation studies using adult piscivorous species. Logistically it was very difficult to obtain the numbers/biomass of advanced walleye needed in this study. Obtaining numbers of advanced walleye necessary for a successful biomanipulation is probably not practical in a wetland management context.

PUBLICATIONS

Below are lists of scientific reports and other publications by personnel in the Wildlife Populations and Research Unit for the approximate period March 2003 through February 2004. 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).

PUBLICATIONS

Angeler, D.G., P. Chow-Fraser, **M.A. Hanson,** S. Sanchez-Carillow, and K. D. Zimmer. 2003. Biomanipulation: a useful tool for freshwater wetland mitigation? Freshwater Biology 48:2203-2213.

Beyea, J., M. Benner, C. Dunn, M. A. Fajvan, R. Freed, **M. D. Grund,** S. B. Horsley, R. E. Latham, A. F. Rhoads, and B. Shissler. Managing deer from an ecosystem perspective. Stackpole Books. In Review.

Bigalke, B.J., J.A. Jenks, and **C.S. DePerno**. 2003. An efficient lower jaw removal technique for large mammals. *South Dakota Academy of Science* 82:67-72.

Brinkman, T.J., **C.S. DePerno,** J.A. Jenks, and **B.S. Haroldson**. 2004. Movement of female white-tailed deer in an intensively cultivated region of Minnesota. *Journal of Wildlife Management*. In Review.

Brinkman, T.J., J.A. Jenks, **C.S. DePerno, B.S. Haroldson**, and **R.G. Osborn**. 2004. Survival of white-tailed deer in an intensively farmed region of Minnesota. *Wildlife Society Bulletin*. In Press.

Brinkman, T.J., J.A. Jenks, **C.S. DePerno, and B.S. Haroldson**. 2004. *Clostridium perfringens Type A* induced disease in a free-ranging white-tailed deer fawn in Minnesota? *The Prairie Naturalist*. In Press.

Brinkman, T.J., K.L. Monteith, J.A. Jenks, and **C.S. DePerno**. 2004. Predicting neonatal age of white-tailed deer in the northern Great Plains. *The Prairie Naturalist*. In Press.

Carstensen, M., and **G. D. DelGiudice**. 2003. Birth, morphological, and blood characteristics of free-ranging white-tailed deer neonates. Journal of Wildlife Diseases. In Press.

*Carstensen, M., **G. D. DelGiudice**, and **B. S. Sampson**. 2003. Using doe behavior and vaginal implant transmitters to capture white-tailed neonates in north central Minnesota. Wildlife Society Bulletin 31: 634-641.

DePerno, C.S., and J.A. Jenks. 2004. Antlered female deer: a scientific perspective. *Whitetails* Winter 2004:38-39.

DePerno, C.S., and J.A. Jenks. 2004. Antlered female deer harvested in Minnesota. *Minnesota Conservation Volunteer*. In Press.

DePerno, C.S., and J.R. Anderson. 2003. Documentation of a late born fawn in south central Minnesota. *Whitetails* Summer 2003:54.

DePerno, C.S., and J.A. Jenks. 2003. Antlered female deer: how and why? *Whitetail Journal* October 2003:70-74.

DePerno, C.S., S.L. Griffin, J.A. Jenks, and L.A. Rice. 2003. Multidimensional cover characteristics: Is variation in habitat selection related to white-tailed deer sexual segregation? *Journal of Mammalogy* 84:1316-1329.

Dingman, K.J., **R.O. Kimmel**, J.D. Krenz, and B.R. McMillan. 2003. Hunter interference and ease of access for spring wild turkey hunting in Minnesota. Northern Wild Turkey Workshop Proceedings. Page 20.

Edwards, A.J., M.W. Schrage, and **M.S. Lenarz.** 2004. Northeastern Minnesota moose management – a case study in cooperation. Alces. In Press.

***Erb, J.**, and H.R. Perry, Jr. 2003. Muskrats. Pages 311-348 in G.A. Feldhamer, B.C. Thompson, and J.A. Chapman, editors. Wild Mammals of North America: Biology, Management, and Conservation. 2nd edition.

DelGiudice, G.D., B.A. Sampson, D.W. Kuehn, M. Carstensen, and **J. Fieberg**. 2004 Improving our understanding of the margins of safe capture, chemical immobilization, and handling of free-ranging white-tailed deer. Wildl. Soc. Bull. In Press.

Garshelis, D.L. Variation in ursid life histories — Is there an outlier? *In* D. Lindburg and K. Baragona, editors. Panda conservation. Univ. California Press. In Press.

Garton, E. O., J. T. Ratti, and **J. H. Giudice**. 2004 (In Press). Research and experimental design. *In* C. E. Braun, ed. Research and management techniques for wildlife and habitats. Sixth edition. The Wildlife Society, Bethesda, Maryland. In Press.

Gibbs, M.C., J.A. Jenks, **C.S. DePerno**, F.F. Sowell, and K.J. Jenkins. 2004. Cervid range utilization in noncommercially thinned ponderosa pine forests. *Journal of Range Management*. In Press.

Grund, M. D., and A. Woolf. Evaluation of accounting models for managing white-tailed deer herds. Ecological Modeling. In Press.

Hansel-Welch, N., M.G. Butler, T.A. Carlson, and **M.A. Hanson**. 2003. Changes in macrophyte community structure in Lake Christina (Minnesota), a large shallow lake, following biomanipulation. Aquatic Botany 75:323-337.

Haroldson, B.S., E.P. Wiggers, J. Beringer, L.P. Hansen, and J.B. McAninch. 2003. Evaluation of aerial thermal imaging for detecting white-tailed deer in a deciduous forest environment. *Wildlife Society Bulletin* 31:1188-1197.

Haroldson, K.J. 2003. Weather, food, and roost cover: key factors limiting the distribution or northern wild turkeys. Northern Wild Turkey Workshop Proceedings. Page 11.

Higgins, K. F., **R. G. Osborn**, D. E. Naugle, and K. K. Bakker. Contrasting the potential effects of biomass fuel, soy-based fuel, ethanol and wind energy developments on Northern Great Plains wildlife. *Transactions of the North American Wildlife Conference* (In Press).

Kane, D.F., **R.O. Kimmel**, W.E. Faber, and G.C. Nelson. 2003. Winter survival of wild turkeys translocated north of their ancestral range in Minnesota. Northern Wild Turkey Workshop Proceedings. Page 14.

Kimmel, R.O. 2003. Northern wild turkeys-the final frontier. Northern Wild Turkey Workshop Proceedings. Pages 4-5.

Klaver, R.W., J.A. Jenks, **C.S. DePerno**, and S.L. Griffin. 2004. Fuzzy classification of seasonal range use by white-tailed deer. *Journal of Wildlife Management*. In Review.

Klaver, R.W., J.A. Jenks, **C.S. DePerno**, and S.L. Griffin. 2004. Effect of scale on habitat selection of white-tailed deer in the central Black Hills, South Dakota and Wyoming. *Wildlife Monograph*. In Review.

Krueger, W. 2003. Survey of northern turkey biologists regarding current wild turkey range and plans for continued expansion. Northern Wild Turkey Workshop Proceedings Page 6.

Laurich, L.M., K.D. Zimmer, M.G. Butler, and **M.A. Hanson**. 2003. Selectivity of zooplankton prey by fathead minnows and brook sticklebacks. Wetlands 23:416-422.

Leafloor, J. O., K. F. Abraham, F. D. Caswell, K. E. Gamble, R. N. Helm, D. D. Humburg, **J. S. Lawrence**, D. R. Luukkonen, R. D. Pritchert, E. L. Warr, G. G. Zenner. 2004. Canada goose management in the Mississippi Flyway. Proceeding of the International Canada Goose Symposium. In Press.

Long, E. S., D. R. Diefenbach, C. S. Rosenberry, B. D. Wallingford, and **M. D. Grund**. Landscape structure influences dispersal distance of white-tailed deer. Journal of Mammalogy. In Review.

Nielsen, C. K., R. G. Anderson, and **M. D. Grund**. 2003. Landscape influences on deer-vehicle accident areas in an urban environment. Journal of Wildlife Management: 67:46-51.

Schroeder, S., D. C. Fulton, and **J. S. Lawrence**. 2004. The 2002 Waterfowl Hunting Season in Minnesota: A study of hunters' opinions and activities. Minnesota Cooperative Fish and Wildlife Research Unit and Minnesota Department of Natural Resources. 124pp.

Sheaffer, S. E., D. H. Rusch, D. D. Humburg, **J. S Lawrence**, G. G. Zenner, M. M Gillespie, F. D. Caswell, S. Wilds, and S. C. Yaich. 2004. Survival, movements, and harvest of Eastern Prairie Population Canada geese. Wildlife Monographs 156. 54pp.

Zicus, M. C., J. S. Fieberg, and D. P. Rave. 2003. Does mallard clutch size vary with landscape composition: a different view. Wilson Bulletin 115:409-413.

Zicus, M. C., D. P. Rave, and M. R. Riggs. Factors influencing incubation egg-mass loss for three species of waterfowl. Condor 106: In Press.

Zicus, M. C. The common goldeneye. Pages xxx-xxx in J. Kear, ed. Bird families of the world: ducks, geese, and swans. Oxford University Press, Oxford, UK. In Press.

Zimmer, K.D., **M.A. Hanson**, and M.G. Butler. 2003. Relationships among nutrients, phytoplankton, macrophytes, and fish in prairie wetlands. Canadian Journal of Fisheries and Aquatic Sciences 60:721-730.

Zimmer, K.D., **M. A. Hanson**, and M. G. Butler. 2003. Interspecies relationships, community structure, and factors influencing abundance of submerged macrophytes in prairie wetlands. Wetlands 23:717-728.

OTHER PUBLICATIONS/REPORT/EDITORSHIPS

Carstensen, M., **G. D. DelGiudice**, and **D. W. Kuehn**. 2003. How effective are vaginal implant transmitters for locating white-tailed deer neonates? Proceedings of The Wildlife Society's Tenth Annual Conference, September 6-10, 2003, Burlington, Vermont.

DelGiudice, G. D., **B. A. Sampson**, and **D. W. Kuehn**. 2003. Assessing the effects of diminishing conifer cover on the winter distribution, movements, and survival of female white-tailed deer in north central Minnesota. Proceedings of the Second Annual Forest and Wildlife Research Review, January, 2003, University of Minnesota, Duluth, Minnesota.

Edwards, A.J., M.W. Schrage, and **M.S. Lenarz**. 2003. Northeastern Minnesota moose management – a case study in cooperation. Proceedings of the North American Moose Conference and Workshop. May 18-21 2003, Jackson, Wyoming.

Giudice, J. H. 2003. Minnesota August Roadside Survey. Technical report, Minnesota Department of Natural Resources, Farmland Wildlife Populations and Research Group, Madelia, Minnesota, USA. 17pp.

Prairie-Chicken Survey Committee (**J. H. Giudice,** chair). 2003. Minnesota prairie-chicken management survey: MNDNR study proposal. Minnesota Department of Natural Resources, Farmland Wildlife Populations and Research Group, Madelia, Minnesota, USA. 57pp.

Giudice, J. H., and **K. J. Haroldson**. 2003. Wildlife response to the Conservation Reserve Program and other land-use changes in Minnesota. Project Completion Report. USGS, Biological Resources Division, Midcontinent Ecological Science Center, Fort Collins, CO. 49pp.

Humpal, C. A., **R. G. Wright**, and **G. D. DelGiudice**. 2003. Comparison of vegetation type selection by female white-tailed deer during winter using two methods of analysis. Proceedings of The Wildlife Society's Tenth Annual Conference, September 6-10, 2003, Burlington, Vermont.

Lenarz, M.S. 2003. Effectiveness of changes to the aerial moose survey protocol in northeastern Minnesota. Proceedings of the North American Moose Conference and Workshop. May 18-21 2003, Jackson, Wyoming.

L.F.P. Gondim, N.E. Mateus-Pinilla, W. Pitt, D. Mech, M. Nelson, **M.S. Lenarz**, and M.M. McAllister. 2003. Transmission of Neospora caninum between wild and domestic animals, and isolation of the parasite from wildlife. Annual Midwest Conference of Parasitologists, held in East Lansing, Michigan, June 5 and 6.

Ogunremi, O., M. Lankester, E. Cox, D. Murray, W. Ballard, H. Whitlaw, **M. Lenarz**, M. Schrage, M. Nelson, A. Edwards. 2003. Serological diagnosis of Parelaphostronglylus tenuis infection in moose. Proceedings of the 52nd Annual Conference of the Wildlife Disease Association. August 11-14 2003, Saskatoon, Alberta.

Peek, J., J. Beecham, **D. Garshelis**, F. Messier, S. Miller, and D. Strickland. 2003. Management of grizzly bears in British Columbia: a review by an independent scientific panel. Submitted to Minister of Water, Land and Air Protection, Government of British Columbia, Victoria, B.C., Canada.