Summaries of Wildlife Research Findings

2010

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

SUMMARIES OF WILDLIFE RESEARCH FINDINGS
2010

Edited by:
Glenn D. DelGiudice
Marrett Grund
Jeffrey S. Lawrence
Mark S. Lenarz

Minnesota Department of Natural Resources
Division of Fish and Wildlife
Wildlife Populations and Research Unit
500 Lafayette Road, Box 20
St. Paul, MN 55155-4020
(651) 259-5203

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For more information contact:

DNR Information Center
500 Lafayette Road
St. Paul, MN 55155-4040
(651) 296-6157 (Metro Area)
1 888 MINNDNR (1-888-646-6367)

TTY (651) 296-5484 (Metro Area)
1 800 657-3929
http://www.mndnr.gov

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Wetlands Wildlife Populations and Research Group
102 – 23rd Street NE
Bemidji, Minnesota 56601
(218) 308-2282

NESTING ECOLOGY OF RING-NECKED DUCKS IN THE BOREAL FOREST OF NORTHERN MINNESOTA

Charlotte Roy and Christine Herwig

SUMMARY OF FINDINGS

We have completed 3 years of fieldwork on this research project. Thus far, we have searched 110 wetlands, located 66 ring-necked duck (*Aythya collaris*) nests, marked 41 hens, and followed 20 broods. We have searched lakes with (7%) and without (93%) boat accesses, near both dirt (56%) and paved roads (44%), and with (51%) and without (49%) houses. Nest success (30%, 27%, and 46%) was within the range of previous reports from the late 1970's and early 1980's for north-central Minnesota. Hen survival during the breeding season and brood survival have not been previously estimated in Minnesota. Additional data collection will enable more robust estimates of these parameters.

INTRODUCTION

The ring-necked duck is a characteristic and important species for the Laurentian Mixed Forest province of Minnesota (Minnesota Department of Natural Resources [MNDNR] 2006), also known as the Boreal or Coniferous Forest biome. Recent surveys near Bemidji have indicated declines in ring-necked duck numbers, despite increases elsewhere in their breeding range (Zicus et al. 2005). Unfortunately, basic information on nest success, hen survival, and brood survival in north-central Minnesota are unavailable, limiting informed interpretation of these local survey data and our understanding of how vital rates affect population growth of ring-necked ducks in the forest. These data are particularly pertinent given the increasing development and recreational use in the forest (MNDNR 2006) and predictions that the spruce-fir forest will shift north of Minnesota as a result of global climate change (Iverson and Prasad 2001).

Nest success, hen survival, and brood survival in the forest are largely unknown. Some data are available for nest success and brood survival in Maine (McAuley and Longcore 1988, 1989), but data for the boreal forest of the upper Midwest are over 35 years old (Sarvis 1972). Limited data are available for nest success outside the forest; Maxson and Riggs (1996) studied nest success of ring-necked ducks in the forest-prairie transition during 1985–1987, and Koons and Rotella (2003) compared nest success of ring-necked ducks to that of lesser scaup (*Aythya affinis*) in the parkland of Manitoba. However, neither study examined hen or brood survival during the breeding season. In general, nesting and brood-rearing information for diving ducks are limited in comparison to the data available for dabbling ducks (Yerkes 2000).

Gathering information on vital rates during the breeding season is an important first step to understanding recent population patterns of ring-necked ducks in Minnesota. Although sensitivity analyses of vital rates on population growth rates are not available for ring-necked ducks, sensitivity analyses for mid-continent mallards indicated that nest success explained the most variation (43%) in population growth rates (Hoekman et al. 2002). A similar analysis for the Great Lakes Region indicated that duckling survival (32%) and nest success (16%) accounted for the greatest variation in mallard population growth rates during the breeding season (Coluccy et al. 2008).

OBJECTIVES

1. To obtain baseline information on ring-necked duck nest success, hen survival, and brood survival before fledging in the forest.

2. To examine how these vital rates vary along a gradient of development and recreational use (e.g., number of dwellings, boat access, proximity to roads).

METHODS

We used multiple methods and data sources to identify lakes to search, including locations of pairs and lone males from a ring-necked duck helicopter survey conducted during 2004–2010 and ground surveys conducted on 10–14 lakes in the Bemidji area beginning in 1969. The survey data were used to identify land cover attributes of wetlands that ring-necked ducks used (U. S. Geological Survey Gap Analysis Program [GAP] types 12 and 13 surrounded by GAP types 10, 14, and 15). We identified 103 lakes within a 40-km (25-mile) radius of Bemidji with land cover attributes similar to those used in the 2 surveys. In 2009, we scouted wetlands in early spring and focused nest- searching efforts on the wetlands where ring-necked ducks had been seen. In 2010, we used scouting data from 2007 to 2009 to identify lakes where ring-necked ducks had been observed. We excluded lakes considered unsafe to search or where we had been denied access. This process resulted in 95 basins as targets for nest-searching in 2010.

We searched for ring-necked duck nests in the springs and summers of 2008–2010. To locate nests, we searched emergent vegetation on floating bog mats and along wetland margins using bamboo poles and nest drags. When a nest was located, we determined the stage of incubation by candling eggs (Weller 1956) and from the appearance of new eggs in the nest. We determined water depth, concealment using a Daubenmire frame and Robel pole (Daubenmire 1959, Robel et al. 1970), predominant vegetation (e.g., cattail, sedge), and distance to open water at each nest after it hatched or failed, and at one random point 25 m from the nest.

Late in incubation, we trapped hens on nests with Weller traps (Weller 1957) to attach radio-transmitters. Because initially we were concerned that a surgical transmitter attachment method might be too disruptive to incubating hens, we tried a bib-type transmitter attachment method, which had been used with previous success in wood ducks (Montgomery 1985). This attachment method was faster and less invasive than surgical methods. Hens received a transmitter fastened to a Herculite® fabric bib with dental floss and superglue (total weight of approximately 11 g). We modified the method used unsuccessfully with redheads (*Aythya americana*) by Sorenson (1989) by securing the bib more tightly and by preening the bib into the breast feathers as in Montgomery (1985). After the transmitter was in place, we trimmed any excess fabric so that feathers concealed the transmitter. Due to concerns about low hen and brood survival during 2008 and 2009, we changed the transmitter attachment method in 2010. We tried the surgical transmitter attachment method that we had been using for the MNDNR-funded study on post-fledging ring-necked ducks (Korschgen et al. 1996). However, we used a local anesthetic (i.e., lidocaine) instead of isoflurane so that we could do surgeries in the field (Corcoran et al. 2007). We also used propofol, injected intravenously, on 6 hens to reduce nest abandonment (Rotella and Ratti 1990, Machin and Caulkett 2000). When propofol was used, hens were placed on nests rather than being released from the edge of the wetland.

Nests were monitored every 4–7 days to determine fate (abandoned, depredated, or successful) and Mayfield nest success (Mendall 1958, Mayfield 1975). After nests hatched, we attempted to monitor broods every 3–7 days. At each observation, we counted the ducklings present, and when possible, aged them from a distance based on plumage characteristics (Gollop and Marshall 1954). Broods were monitored until ducklings reached age Class III (i.e., 39-49 days old) or until total brood loss occurred. We considered hens to have lost their entire brood when hens were observed without any ducklings for 3 observations or if the hen was found >16 km (10 miles) from the nesting lake. We continued to monitor hens after the brood-rearing period for as long as they could be tracked before migration to examine their survival using the Kaplan-Meier method (Kaplan and Meier 1958).

RESULTS

Thus far, we have searched 110 wetlands (Figure 1), located 66 active nests, marked 41 hens, and followed 20 broods. We searched for nests on 37 wetlands for a total of 73 searches (17 wetlands searched once and 20 wetlands searched >1 time) between 22 May and 22 July 2008, 37 wetlands searched 54 times (21 wetlands once and 16 wetlands searched >1 time) between 29 May and 22 July 2009, and 73 wetlands searched 128 times (35 wetlands once and 38 wetlands searched >1 time) between 19 May and 12 July 2010.

Nest Survival

We located 18 (14 active, 4 depredated when found) ring-necked duck nests on 10 wetlands in 2008, 20 active nests on 11 wetlands in 2009, and 32 active nests on 17 wetlands in 2010. In 2008, 8 nests hatched, 4 were depredated when found, 3 were depredated after they were found, and 3 nests were flooded by rising water levels following rain events. Average clutch size for nests that were incubated was 9.1 ± 0.6 (mean ± SE, range = 7–15, *n* = 12 nests with 109 eggs) and 86.6 ± 0.1% of eggs hatched in nests that hatched. In 2009, 7 nests hatched, 9 were depredated, and 4 were abandoned, with at least 2 cases of abandonment likely due to trapping. The average clutch size for incubated nests was 8.3 ± 0.3 (range = 7–11, *n* = 19 nests with 158 eggs) and 89.5 ± 0.6% of the eggs hatched in nests that were successful. In 2010, 13 nests hatched, 9 were depredated, 6 were abandoned after trapping and transmitter attachment, 2 were abandoned for other reasons, 1 had an unknown fate, because we could not determine the outcome based on evidence at the nest site, and 1 failed, because the hen died during transmitter-implantation surgery. We began using propofol on all hens mid-way through the field season, because 5 of 13 hens marked without propofol later abandoned their nests. Average clutch size for incubated nests was 8.3 ± 0.3 (range = 5–10, *n* = 30 nests with 250 eggs) and 84.5 ± 0.1% of eggs hatched. Mayfield nest success for a 35-day period was 30% in 2008, 27% in 2009, and 46% in 2010.

Hen Survival

We put transmitters on 8 hens in 2008, 14 hens in 2009, and 19 hens in 2010. In 2008, 2 hens died due to predation during the tracking season; 1 lost her nest late in incubation and the other had a brood. Both of these birds had been observed preening more than other birds with transmitters, although this behavior occurred during the first 2 weeks after marking and then subsided. Both deaths occurred after this period, one 3 weeks post-marking and the other 4 weeks post-marking. All birds in 2008 continued to nest and rear broods after transmitter attachment, with the exception of birds that lost their nests to flooding. In 2009, 6 hens died during the monitoring period (17, 20, 32, 33, 55, and 84 days post-marking). Evidence obtained at the recovery sites indicated that radioed birds were either depredated or scavenged by avian predators (3) or by mammalian predators (1). Additionally, there were 2 cases in which a probable cause of death could not be determined, because the transmitter was underwater and no carcass was found. All of the hens that died did not have broods at the time of death; 3 lost their nest late in incubation, 1 abandoned her nest due to trapping, and 2 lost broods early after hatching. In 2010, only 1 hen died during the monitoring period. She died 17 days after marking and appeared to have been killed by a mammalian predator. She did not have a brood. Twelve of 19 transmitters dehisced 55.1 ± 6.0 days (range = 30–121 days) after attachment. Hen survival through mid-September was 0.80 ± 0.18 for 2008, 0.54 ± 0.08 for 2009, and 0.88 ± 0.11 for 2010.

Brood and Duckling Data

In 2008, 7 radiomarked hens had broods (*n* = 57 ducklings). One brood survived to fledge 5 ducklings. Other broods dwindled slowly, with total brood loss at the IA (1), IB (1), IC (1), and IIA (2) age classes (Gollop and Marshall 1954). The fate of 1 brood could not be determined, because the hen died when the brood was at the IIA stage, and we could no longer relocate the ducklings without the marked hen. We also monitored the brood of 1 unmarked hen that was not trapped in time to give her a transmitter. Her brood made it to the IC stage, but they were not observed again and their fate was uncertain.

Seven broods were monitored in 2009 (*n* = 56 ducklings). Total brood losses occurred at IA (3), IB (1), and IC (1) age classes. One brood fledged 2 young. Another brood matured to IIA before the hen left the wetland, after which time 1 duckling was seen on the wetland and no hens were present.

We observed 6 broods in 2010 (*n* = 40 ducklings); 3 broods survived to age Class III and likely fledged 14 ducklings, 1 brood was located as Class IA ducklings, but the hen was not located again, 1 brood survived until age Class 1A, and another brood survived to age Class IB. Seven marked hens were believed to have hatched ducklings, but were not located with broods before total brood loss.

Brood movements also were observed. In 2009, for example, a hen moved her 3 (IC) young from the nesting wetland to another wetland (~1,205 m) from which they fledged. In another instance, a hen and her brood of 6 (IB) were seen walking to another wetland ~365 m from their nesting wetland. In 2010, 4 hens moved their broods to nearby wetlands at various ages (i.e., IA, IC, IIA, IIB). One of these hens later returned with her brood to the wetland where they hatched and her brood later fledged.

DISCUSSION

Our success finding nests has been comparable to that in other studies that found ring-necked duck nests (45 nests in 3 years, Maxson and Riggs 1996; 35 nests in 2 years, Koons and Rotella 2003, 188 nests in 6 years by R. T. Eberhardt). Thus far, our results have been similar to findings by R. T. Eberhardt in northern Minnesota during 1978–1984 (Hohman and Eberhardt 1998). Our nest survival rates are comparable to his estimates of 44% based on 188 nests. The causes of nest failure in our study (17% flooding, 70% depredation, and 13% abandonment) were similar to those of other studies (16–24% flooding, 67–80% depredation, and 5% abandonment; Mendall 1958, McAuley and Longcore 1989), if we exclude nests where abandonment was attributed to investigator disturbance. Early estimates of hatching success appeared to be lower than those of Eberhardt's previous study in north-central Minnesota (94%, Hohman and Eberhardt 1998), but the springs and summers of 2008 and 2009 were very cool and rainy, and early summer 2010 was very rainy as well, which may have chilled eggs and flooded nests.

Our hen survival rates for the period June–mid-September were low compared to reports for hen mallards during April–September (0.80, Cowardin et al. 1985; 0.60, Blohm et al. 1987; 0.67, Brasher et al. 2006), likely because we marked hens late in incubation, which is a period when hen mortality was expected to be greatest. Brood survival rates also seemed low. Brood survival in ring-necked ducks has only been examined previously in Maine (77% to 45 days, *n* = 64, McAuley and Longcore 1988). Duckling survival in the same study was 37% (*n* = 381). Further investigation is necessary to get better estimates of hen survival and brood survival in ring-necked ducks in Minnesota.

This study is ongoing. Results should be viewed as preliminary and are subject to change with further data collection.

LITERATURE CITED

Blohm, R. J., R. E. Reynolds, J. P. Bladen, J. D. Nichols, J. E. Hines, K. H. Pollock, and R. T. Eberhardt. 1987. Mallard mortality rates on key breeding and wintering areas. Transactions of the North American Wildlife and Natural Resources Conference. 52:246-257.

Brasher, M. G., T. W. Arnold, J. H. Devries, and R. M. Kaminski. 2006. Breeding-season survival of male and female mallards in Canada's prairie-parklands. Journal of Wildlife Management 70:805-811.

Coluccy, J. M., T. Yerkes, R. Simpson, J. W. Simpson, L. Armstrong, and J. Davis. 2008. Population dynamics of breeding mallards in the Great Lakes states. Journal of Wildlife Management 72:1181-1187.

Corcoran, R. M., J. R. Lovvorn, M. R. Bertram, and M. T. Vivion. 2007. Lesser scaup nest success and duckling survival on the Yukon flats, Alaska. Journal of Wildlife Management 71:127-134.

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

Daubenmire, R. F. 1959. A canopy-coverage method of vegetational analysis. Northwest Science 33:43-64.

Gollop, J. B., and W. H. Marshall. 1954. A guide to aging duck broods in the field. Unpublished report. Mississippi Flyway Council Technical Section, Online. http://www.npwrc.usgs.gov/resource/birds/ageduck/index.htm Last accessed 14 October 2011.

Hoekman, S. T., L. S. Mills, D. W. Howerter, J. H. Devries, I. J. Ball. 2002. Sensitivity analyses of the life cycle of mid-continent mallards. Journal of Wildlife Management 66:883-900.

Hohman, W. L., and R. T. Eberhardt. 1998. Ring-necked Duck (*Aythya collaris*). *In* A. Poole and F. Gill, eds. The Birds of North America, No. 329. The Birds of North America, Inc., Philadelphia, Pennsylvania. 32 pp.

Iverson, L. R., and A. M. Prasad. 2001. Potential changes in tree species richness and forest community types following climate change. Ecosystems 4:186-199.

Kaplan, E. L., and P. Meier. 1958. Nonparametric estimation from incomplete observations. Journal of the American Statistical Association 53:457-481.

Koons, D. N., and J. J. Rotella. 2003. Comparative nesting success of sympatric lesser scaup and ring-necked ducks. Journal of Field Ornithology 74:222-229.

Korschgen, C. E., K. P. Kenow, W. L. Green, M. D. Samuel, and L. Sileo. 1996. Technique for implanting radiotransmitters subcutaneously in day-old ducklings. Journal of Field Ornithology 67:392-397.

Machin, K. L., and N. A. Caulkett. 2000. Evaluation of isoflurane and propofol anesthesia for intra-abdominal transmitter placement in nesting female canvasback ducks. Journal of Wildlife Diseases 36:324-334.

Maxson, S. J., and M. R. Riggs. 1996. Habitat use and nest success of overwater nesting ducks in westcentral Minnesota. Journal of Wildlife Management 60:108-119.

Mayfield, H. F. 1975. Suggestions for calculating nest success. The Wilson Bulletin 87:456-466.

Mendall, H. L. 1958. Ring-necked duck in the Northeast. University of Maine Orono, Study no. 73, Bulletin 60(16). 317pp.

McAuley, D. G., and J. R. Longcore. 1988. Survival of juvenile ring-necked ducks on wetlands of different pH. Journal of Wildlife Management 52:169-176.

McAuley, D. G., and J. R. Longcore. 1989. Nesting phenology and success of ring-necked ducks in east-central Maine. Journal of Field Ornithology 60:112-119.

Minnesota Department of Natural Resources. 2006. A vision for wildlife and its use–goals and outcomes 2006–2012. Minnesota Department of Natural Resources, St. Paul. http://files.dnr.state.mn.us/aboutdnr/reports/strategic-documents/wildlife-stategic-plan05-12.pdf Last accessed 17 February 2010.

Montgomery, J. 1985. A collar radio-transmitter attachment for wood ducks and other avian species. Proceedings of the International Conference on Wildlife Biotelemetry 5:19-27.

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.

Rotella, J. J., and J. T. Ratti. 1990. Use of methoxyflurane to reduce nest abandonment in mallards. Journal of Wildlife Management 54:627-628.

Sarvis, J. E. 1972. The breeding biology and ecology of ring-necked ducks in northern Michigan. M. S. thesis, Utah State University, Logan.

Sorenson, M. D. 1989. Effects of neck collars on female redheads. Journal of Field Ornithology 60:523-528.

Weller, M. W. 1956. A simple field candler for waterfowl eggs. Journal of Wildlife Management 20:111-113.

Weller, M. W. 1957. An automatic nest-trap for waterfowl. Journal of Wildlife Management 21:456-458.

Yerkes, T. 2000. Nest-site characteristics and brood-habitat selection of redheads: an association between wetland characteristics and success. Wetlands 20:575-580.

Zicus, M. C., D. P. Rave, J. Fieberg, J. Giudice, and R. Wright. 2005. Minnesota’s ring-necked ducks: a pilot breeding pair survey. Pages 137–158 *in* P. J. Wingate, R. O. Kimmel, J. S. Lawrence, and M. S. Lenarz, eds. Summaries of wildlife research findings, 2004. Minnesota Department of Natural Resources, Wildlife Populations and Research Group, St. Paul.

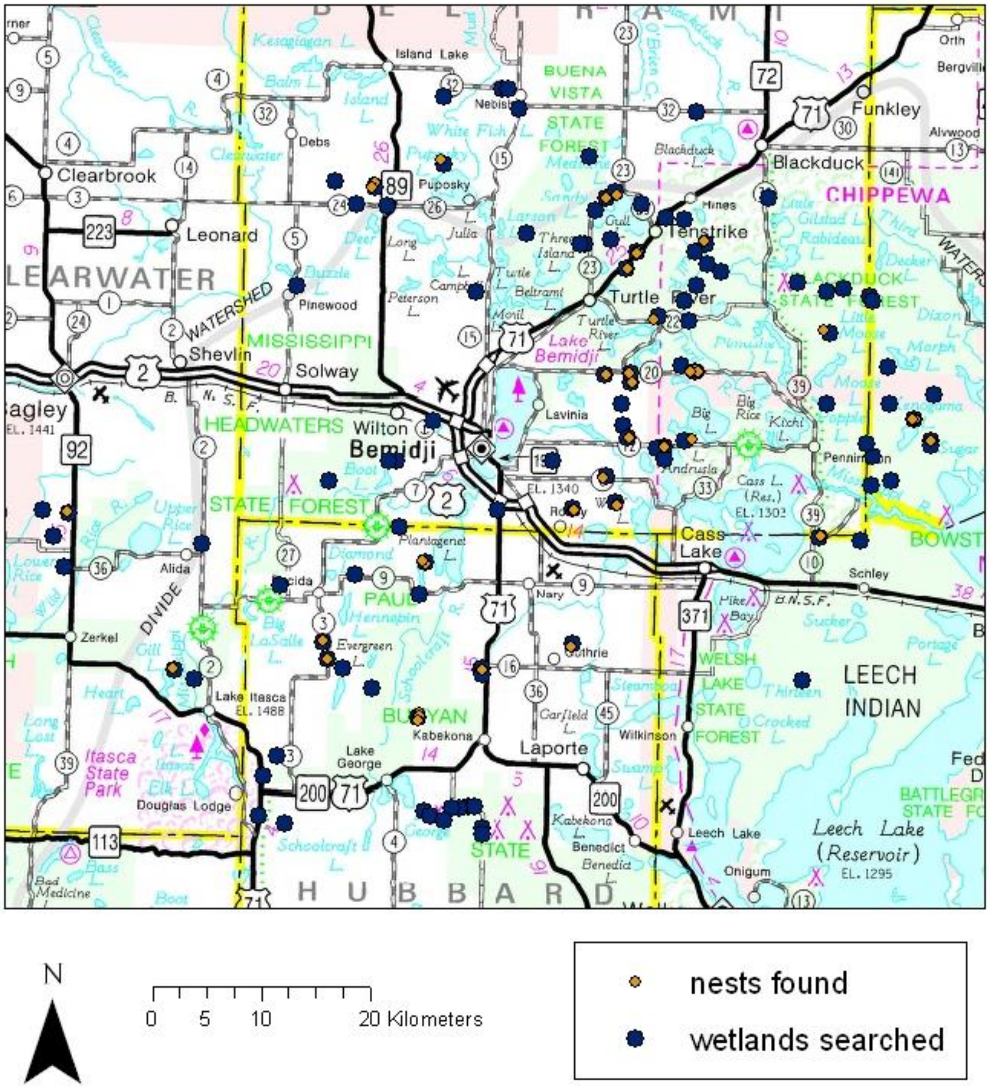

Figure 1. Wetlands searched for ring-necked duck nests in north-central Minnesota during 2008–2010.

MOVEMENTS, SURVIVAL, AND REFUGE USE BY RING-NECKED DUCKS AFTER FLEDGING IN MINNESOTA

Charlotte Roy, Christine Herwig, David Rave, Wayne Brininger[1], and Michelle McDowell[2]

SUMMARY OF FINDINGS

The Minnesota Department of Natural Resources (MNDNR) conducted a study that examined use and survival benefits of waterfowl refuges to locally produced ring-necked ducks (*Aythya collaris*). During 2007–2010, we captured and implanted radiotransmitters into 240 ring-necked ducks prior to fledging. Ducklings were tracked weekly by aircraft and from telemetry receiving stations located on 14 waterfowl refuges. Distances between weekly locations were 8.1 ± 1.8 km (mean ± SE) in 2007, 7.8 ± 2.3 km in 2008, 7.3 ± 1.8 km in 2009, and 10.5 ± 2.6 km in 2010. Young ring-necked ducks used state and federal waterfowl refuges, but this use was not evenly distributed among refuges. Three refuges received the majority of use with >16 birds detected at each refuge during the study. Only 4 of 14 refuges were used by marked ducklings in all 4 years of the study. Refuge use was higher during hunting season than prior to the season opening.

INTRODUCTION

Sizable populations of resident breeding ducks were recognized as a cornerstone to improving fall duck use in the MNDNR Fall Use Plan, yet factors influencing resident populations of ring-necked ducks were poorly understood. Although breeding ring-necked duck populations have been increasing continentally, they may have declined in Minnesota (Zicus et al. 2005). Furthermore, hunter harvest of ring-necked ducks has declined markedly in Minnesota in the last 40 years (U. S. Fish and Wildlife Service, Harvest Surveys, unpublished data), even as numbers of these birds staging on most traditional ring-necked duck refuges in the fall have increased in the state (MNDNR, unpublished data). Efforts to better understand population status began in 2003 with development of a ring-necked duck breeding-pair survey.

The Fall Use Plan also identified a need to better understand the role of refuges in duck management. The influence of north-central Minnesota refuges on the distribution and survival of resident ring-necked ducks was unknown. The intent of this research project was to determine whether refuges benefit locally produced ring-necked ducks and increase survival. Additionally, post-fledging ecology of many waterfowl species has not been investigated. Understanding movements and refuge use in the fall may provide valuable insights into the distribution of refuges required to meet management objectives for ring-necked ducks in Minnesota.

OBJECTIVES

1. Characterize post-fledging movements of local ring-necked ducks prior to their fall departure;
2. Estimate survival of locally produced birds before migration; and
3. Relate survival of locally produced birds to the proximity between natal lakes and established refuges (federal and state) and refuge use in north-central Minnesota.

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[1] U.S. Fish and Wildlife Service, Tamarac National Wildlife Refuge, Rochert, Minnesota 56578
[2] U.S. Fish and Wildlife Service, Rice Lake National Wildlife Refuge, McGregor, Minnesota 55760

STUDY AREA

The study area was primarily in the Laurentian mixed forest province of Minnesota (Figure 1). This area was characterized by mixed coniferous and hardwood forest interspersed with lakes, many of which were dominated by wild rice (*Zizania palustris*). The study area was ~200 x 135 km in size and encompassed a significant portion of the core of ring-necked duck breeding range in Minnesota and numerous important refuges for ring-necked ducks. Two federal and 12 state refuges were included in the study (Table 1) and were not open to public hunting, thus providing “refuge” for ducks during the fall migration.

METHODS

Night-lighting techniques similar to Lindmeier and Jessen (1961) were employed to capture ring-necked ducks prior to fledging during July and August in 2007–2010. Duckling age (Gollop and Marshall 1954) and sex were determined at capture. We implanted radiotransmitters dorsally and subcutaneously primarily on classes IIb (~25–30 days old) and IIc (~31–38 days old) ring-necked ducklings following techniques developed by Korschgen et al. (1996), with 1 modification; we attached mesh to the back of transmitters (D. Mulcahy, U. S. Geological Survey (USGS), Alaska Science Center, personal communication). This change was implemented to improve transmitter retention and minimize dehiscing that occurred during a pilot study in 2006. Ducks were then allowed several hours to recover from surgery before release at their capture location. We also marked ducklings with nasal saddles in 2007 to allow examination of natal philopatry in the spring, but because few birds were resighted in 2008, this marking technique was discontinued.

By early September each year, radiotelemetry stations were established at each refuge as a means of quantifying refuge use. Receivers were programmed to scan each of the established frequencies each hour, 24 hours per day. Data were downloaded weekly from data-loggers from mid-September through early November. Reference transmitters were stationed permanently at each refuge to ensure receivers and data-loggers functioned properly.

Aerial flights with telemetry equipment were also conducted once weekly throughout the fall to document the locations and survival of radiomarked birds within the study area. Additional location and survival information came from USGS Bird Banding Lab banding and harvest reports. These reports included the hunters' names and the dates and locations of harvest.

RESULTS

Capture and Tracking

We captured 52 ducklings between 4 August and 3 September 2007. In 2008, we captured 56 ducklings between 29 July and 26 August, and in 2009, we captured 68 ducklings between 27 July and 25 August. In 2010, 64 ducklings were captured and marked with radiotransmitters from 29 July to 20 August. Capture locations were distributed throughout the study area, but a greater proportion of ducklings were captured on the western half of the study area in all years (31 in 2007, 32 in 2008, 46 in 2009, and 36 in 2010 in western counties compared to 21, 24, 22, and 28 in each respective year in eastern counties; Table 2 and Figure 1).

The number of locations per bird varied from 1 to 17 (10.5 ± 0.3) for the 240 marked birds. On average, 67% of birds in 2007, 82% in 2008, 82% in 2009, and 76% in 2010 were located weekly during surveys beginning when the first bird was marked and continuing through early November. However, success locating birds from aerial flights was higher before hunting season (87% in 2007, 95% in 2008, 95% in 2009, 90% in 2010) than during the week hunting opened in all years (66% in 2007, 83% in 2008, 83% in 2009, 62% in 2010). Success locating birds also appeared to decline as birds began moving more in preparation for migration.

Average weekly movements tended to increase as fall progressed until mid to late October when birds started leaving the study area. For the tracking period, average weekly movements were 8.1 ± 1.8 km in 2007, 7.8 ± 2.3 km in 2008, 7.3 ± 1.8 km in 2009, and 10.5 ± 2.6 km in 2010. Average weekly movements prior to the start of hunting, after birds started moving (6.9 ± 1.0 km in 2007, 7.0 ± 1.6 km in 2008, 7.5 ± 1.7 km in 2009, and 9.3 ± 1.7 km in 2010) appeared to be shorter than after hunting season opened (14.4 ± 3.0 km in 2007, 16.8 ± 4.9 km in 2008, 14.4 ± 2.4 km in 2009, and 22.8 ± 2.7 km in 2010) in all years. All but 3 birds left their natal lake before hunting opened over the 4-year period. These 3 birds should have been able to fly by the start of hunting, based on their age at capture. All radiomarked ducklings should have been able to fly by opening day of the migratory waterfowl hunting season based on their age at capture; however, a few ducks (7 in 2007, 6 in 2008, 5 in 2009, and 4 in 2010) may not have been capable of flight for the youth hunt that occurred 2 weeks prior to the regular season.

Mortalities and Transmitter Losses

In 2007, 15 radiomarked birds (*n* = 52) were known to have died by the end of the monitoring period (8 November); 5 were shot and retrieved by hunters (all in Minnesota), and 10 were depredated. Four of the 5 hunter-harvested birds were harvested during the first 2 days of the waterfowl hunting season (29 and 30 September). Evidence obtained at the recovery site indicated that radioed birds were either depredated or scavenged by mink (*Mustela vison*) and other mammals (7), or great-horned owls (*Bubo virginianus*) and other raptors (3). Six additional birds were harvested after the monitoring period ended; 3 were harvested during the 2007 hunting season (2 in Louisiana and 1 in Illinois), 2 were harvested in 2008 (1 in South Carolina and 1 in Arkansas), and 1 was harvested in 2009 (Arkansas). Six transmitters retrieved from open water in 2007 were assumed to have dehisced; thus the fate of these birds was unknown.

In 2008, 25 radiomarked birds (*n* = 56) were known to have died by the end of the monitoring period (18 November); 8 were harvested by hunters (all in Minnesota), 11 were depredated, and 6 died of unknown causes. Four of the 8 hunter-harvested birds were shot during the first 2 days of the waterfowl hunting season (4 and 5 October). Radioed birds were either depredated or scavenged by mink, raccoon (*Procyon lotor*) and other mammals (5), raptors (1), or unknown sources (5) based on evidence at the recovery site. A cause of mortality could not be determined for 6 birds whose transmitters were found with no additional evidence at the site, and they were not believed to have dehisced, because they were not located in water. Six additional birds were harvested after the monitoring period ended; 4 were harvested during the 2008 hunting season (2 in Louisiana, 1 in Arkansas, and 1 in South Carolina), and 2 were harvested during 2009 (1 in Minnesota and 1 in Cuba). Six radios were found in open water and assumed to have dehisced in 2008. Two of the birds that dehisced their transmitters were subsequently harvested (1 in 2008 in Oklahoma and 1 in 2009 in Cuba, mentioned above). The fate of the 4 other birds was unknown.

In 2009, 31 radiomarked birds (*n* = 68) were known to have died by the end of the monitoring period (9 November); 7 birds were shot by hunters (all in Minnesota), 13 were depredated, 10 died of unknown causes. One bird may have died as a result of surgery. Examination of the carcass revealed that the transmitter had migrated forward toward the crop and may have affected the bird's ability to feed. Two of the 7 harvested birds were shot during the youth-opener (19 September) and only 1 was shot during opening weekend (3 and 4 October). Radioed birds were either depredated or scavenged by mink, river otter (*Lontra canadensis*) and other mammals (10), raptors (1), and unknown sources (2). Seven additional birds were harvested after the monitoring period ended during the 2009 hunting season (1 each in Alabama, Florida, Illinois, Missouri, and Texas) and 2 were harvested during 2010 (1 in Minnesota and 1 in Georgia). Four transmitters appeared to have dehisced in 2009, and the fates of 3 of these birds were unknown. One of the birds that lost its transmitter was subsequently harvested in Minnesota and was included in the harvest total above.

In 2010, 24 radio-marked birds were known to have died by the end of the monitoring period (8 November); 10 birds were shot by hunters (all in Minnesota), 13 were depredated, and 1 was classified as unknown as no cause of death was determined during an examination of the carcass. Three were shot during the first 2 days of the waterfowl hunting season (2 and 3 October). Radioed birds were either depredated or scavenged by mink and other mammals (5), raptors (2), or unknown sources (6). Five additional birds were harvested (3 in Louisiana, 1 in Missouri, and 1 in Texas) after the monitoring period ended in 2010. Eight transmitters appeared to have dehisced in 2010. One of the birds that lost its transmitter was subsequently harvested in Minnesota and included in the harvest total above. The fates of the 7 other birds were unknown.

Refuge Use

In the 4 years of the study, 75 birds were documented at refuges based on aerial surveys and tower detections, which was 31% of birds marked (*n* = 240) and 37% of birds that fledged (*n* = 204). Refuge use by radiomarked birds increased with the onset of hunting (Figure 2). Although some birds used refuges both prior to hunting and during the hunting season, fewer ducks appeared to use refuges prior to hunting (8 birds in 2007, 6 in 2008, 7 in 2009 and 11 in 2010) than during hunting season (16 birds in 2007, 10 in 2008, 15 in 2009 and 25 in 2010).

All refuges were used at least once during the study (Table 1); however, not all refuges were used equally. The most heavily used refuges (based on number of marked birds) were Drumbeater, Mud Goose, and Tamarac NWR (Table 1). Additionally, although use of individual refuges varied each year, only 4 refuges were used every year: Drumbeater, Mud Goose, Tamarac NWR, and Rice Pond. Most birds visited only 1 refuge (46 of 75 birds), but 29 birds used >1 refuge during the fall (Table 3). Refuge use was diurnal (0700 to 1800 hr), nocturnal (1900 to 0600 hr), or throughout a 24-hour period based on detections by refuge tower receivers (Table 4, Figure 3).

In 2007, 17 radiomarked birds used 6 different refuges. Mud Goose and Tamarac NWR were used by the most individual birds (Table 1). A similar pattern was observed in 2008 with 11 radiomarked birds using 8 refuges. The most heavily used refuge was Mud Goose. In 2009, refuge use was documented for 16 radiomarked birds at 11 refuges. The most heavily used refuge in 2009 was Drumbeater. In 2010, aerial and tower locations indicated refuge use by 28 radiomarked birds. All 14 refuges were used by marked birds, but the refuge used by the most birds was Drumbeater. In 2010, Rice Lake NWR did not participate in monitoring ducklings; however, 4 ducklings were detected there in 2010 based on aerial monitoring.

DISCUSSION

This study documents use of state and federal waterfowl refuges by locally-produced, young ring-necked ducks. We located approximately 30% of marked ducklings at a waterfowl refuge. Most refuge use occurred during the hunting season, which is consistent with these areas being used to escape hunting pressure. We would expect most of the use by ring-necked ducks to occur during the day if birds used these areas primarily to avoid hunters. However, we also observed night use, with some refuges used primarily at night. This night use suggests refuge use may have more benefits than just refuge from hunting.

Refuges were often designated as refuges, because they received heavy bird use and were important as foraging or staging areas. Thus, young birds may have used some of these refuges for foraging or staging for migration. These other uses might also explain refuge use before hunting opened. We would expect less use for foraging and staging before hunting season opened, because young birds were slowly gaining the ability to fly in the weeks preceding the opening of hunting.

Diversity in benefits of different refuges can also explain the variability in use of refuges by birds. For example, Drumbeater Lake State Waterfowl Refuge received a lot of day use during hunting hours, but the birds left in the evenings, presumably to forage elsewhere. Mud Goose Waterfowl Refuge was used at all times of day, perhaps, because of the abundant food resources (e.g., rice) available on the refuge. In contrast, Rice Pond Refuge was used primarily at night and received very little day use.

Although we knew at the outset that some of our focal refuges received intense use by ring-necked ducks in the fall, we did not know whether this use was by Minnesota birds or by migrants. For example, Drumbeater Lake State Waterfowl Refuge was used consistently by ring-necked ducks during the fall based on fall waterfowl surveys flown by the Wetland Wildlife Population and Research Group. However, use by local birds had not been previously documented, so it's utility to resident populations was uncertain. In contrast, Rice Lake NWR, which often has large numbers of ring-necked ducks counted each fall, was not used by marked ducklings until the 2010 season. In 2010, Rice Lake NWR had the largest number of ring-necked ducks observed in over a decade (~250,000 ducks on 19 October). Although, this refuge is outside the capture area, we did not know if it would be important to birds from within the capture area as they began migrating south. Thus, we have learned that local birds do use state and federal refuges, that the time and amount of use varies among refuges and among years, and that refuges may provide benefits that are not necessarily related to hunting.

The 2010 and final field season was just completed. Results are preliminary and subject to revision. Future analyses will attempt to explore the benefits of these state and federal refuges more quantitatively. Additional results and discussion of these analyses will be included in future progress reports.

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LITERATURE CITED

Gollop, J. B., and W. H. Marshall. 1954. A guide to aging duck broods in the field. Unpublished report. Mississippi Flyway Council Technical Section, Online. http://www.npwrc.usgs.gov/resource/birds/ageduck/index.htm Last accessed 14 October 2011.

Korschgen, C. E., K. P. Kenow, W. L. Green, M. D. Samuel, and L. Sileo. 1996. Technique for implanting radiotransmitters subcutaneously in day old ducklings. Journal of Field Ornithology 67:392–397.

Lindmeier, J. P., and R. L. Jessen. 1961. Results of capturing waterfowl in Minnesota by spotlighting. Journal of Wildlife Management 25:430–431.

Zicus, M. C., D. P. Rave, J. Fieberg, J. Giudice, and R. Wright. 2005. Minnesota's ring-necked ducks: a pilot breeding pair survey. Pages 137–158 *in* P. J. Wingate, R. O. Kimmel, J. S. Lawrence, and M. S. Lenarz, eds. Summaries of wildlife research findings, 2004. Minnesota Department of Natural Resources, Wildlife Populations and Research Group, St. Paul.

Table 1. National Wildlife Refuges and Minnesota State Refuges included in the study area, peak numbers of ring-necked ducks during fall migration (migrant and local birds), number of recording telemetry stations established on each refuge, and use of each refuge by radiomarked, post-fledging ring-necked ducks, Minnesota, 2007–2010. Individuals may have been detected at more than 1 refuge, Minnesota, 2007-2010. Note that the data collected by receivers at each refuge are still under examination and the number of birds detected by towers is subject to revision.

Refuge	Peak numbers	Stations	No. radiomarked birds using refuge				Total
			2007	2008	2009	2010	
National Wildlife Refuge							
Rice Lake	120,000	4	0	0	0	NA[1]	0
Tamarac	50,000	3	6	1	3	6	16
State Waterfowl Refuge/State Game Refuge							
Donkey Lake	350	1	1	0	1	3	5
Drumbeater Lake	280,000	1	3	2	7	15	27
Fiske and Blue Rock Lakes	40,000	1	4	0	0	5	9
Gimmer Lake	3,500	1	0	3	0	7	10
Hatties and Jim Lakes	0	1	0	0	1	4	5
Hole-in-Bog Lake	4,000	1	0	0	4	1	5
Mud Goose	4,000	1	6	6	3	7	22
Lower Pigeon Lake	700	1	0	1	3	3	7
Pigeon River Flowage	700	1	0	1	3	3	7
Preston Lakes	1,800	1	0	2	2	3	7
Round Lake	11,000	1	0	0	2	3	5
Rice Pond	32	1	2	2	2	1	7

[1] NA–Not applicable; Rice Lake National Waterfowl Refuge did not participate in monitoring for ducklings in 2010.

Table 2. Ring-necked duckling captures per county (%) in Minnesota, 2007–2010.

County	Captures			
	2007 (*n* = 52)	2008 (*n* = 56)	2009 (*n* = 68)	2010 (*n* = 64)
Aitkin	2	0	3	3
Becker	12	2	6	6
Beltrami	29	13	25	17
Cass	17	18	9	13
Clearwater	10	27	19	14
Hubbard	10	13	10	14
Itasca	17	18	18	25
Koochiching	4	7	3	3
Polk	0	4	4	5
Wadena	0	0	3	0

Table 3. Number of ring-necked ducklings that used 1 or more refuges, Minnesota, 2007–2010.

No. refuges visited	No. birds
1	46
2	12
3	8
4	5
5	2
6	2

Table 4. Minnesota refuges classified as day use, night use, and 24-hour use based on data collected by monitoring equipment established to detect refuge use by radiomarked post-fledging, ring-necked ducklings, Minnesota, 2007-2010. Tamarac National Wildlife Refuge (NWR) had 3 towers established on 3 different lakes and were treated separately.

Day use	Night use	24-hour use	Not used
Donkey	Pigeon River	Mud Goose	Tamarac NWR - Chippewa
Drumbeater	Rice Pond	Round	
Fiske Blue Rocks		Tamarac NWR – Little Flat	
Gimmer			
Hatties and Jim			
Hole-in-Bog			
Lower Pigeon			
Preston Lakes			
Tamarac NWR – Flat			
Rice Lake NWR			

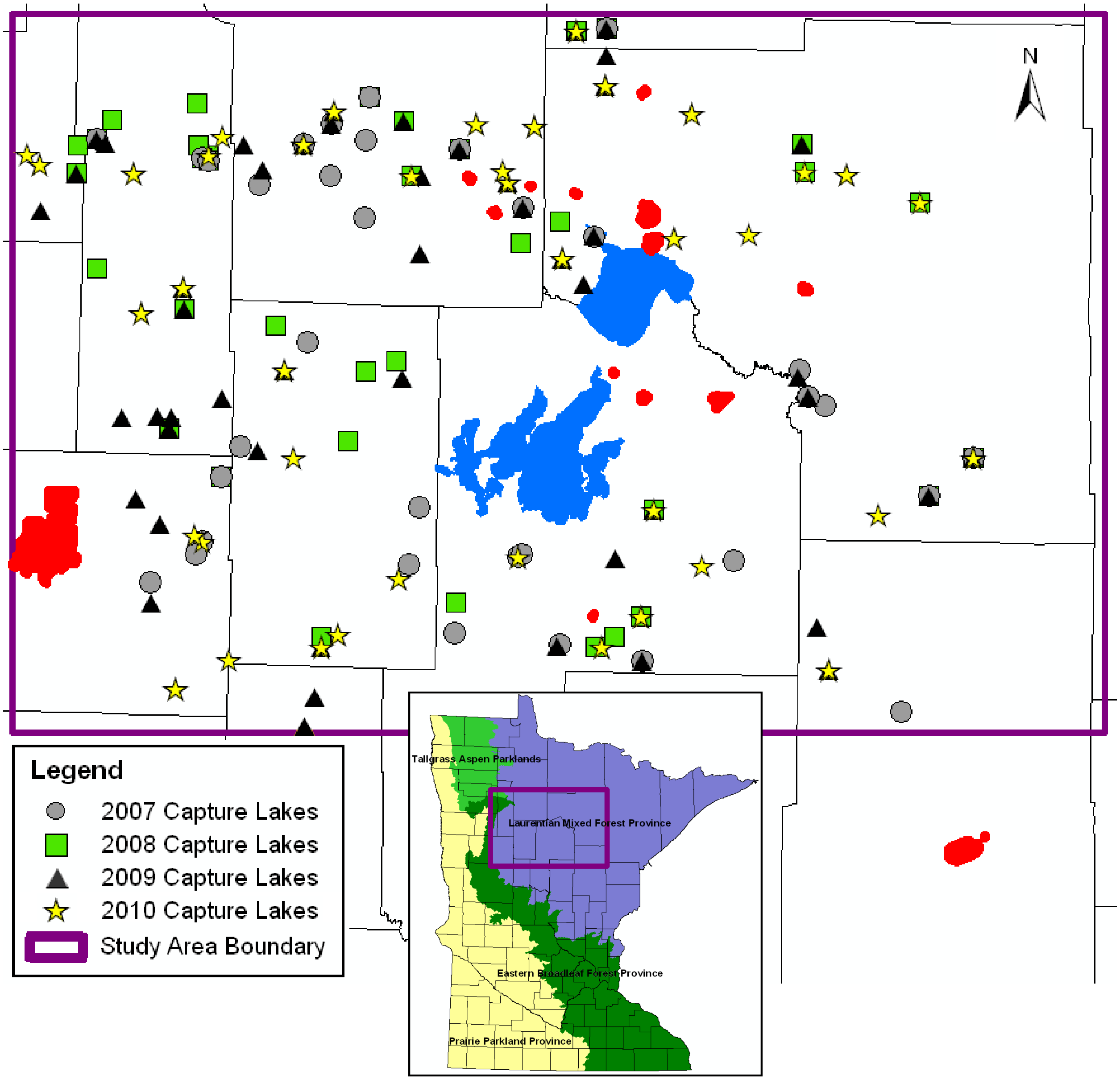

Figure 1. Study area map showing capture locations for ring-necked duck ducklings in north-central Minnesota, 2007–2010. Twelve state waterfowl/game refuges and 2 National Wildlife Refuges are depicted in red. Winnibigoshish and Leech lakes are shown in blue.

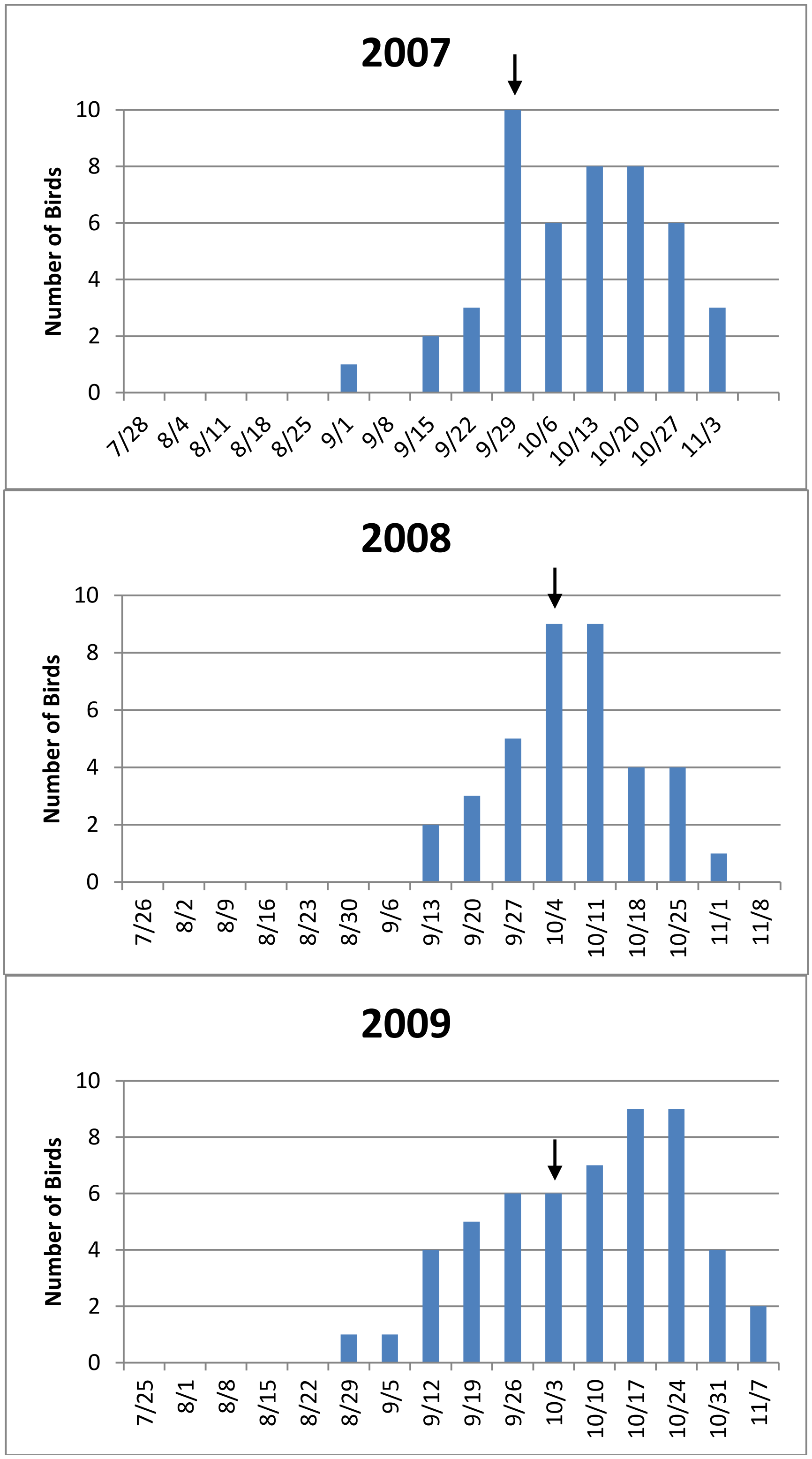

Figure 2. Weekly use of refuges by post-fledging ring-necked ducks before and during hunting season in Minnesota, 2007–2010. Weeks are from Saturday through Friday with the Saturday date shown. Arrows indicate the week waterfowl hunting opened.

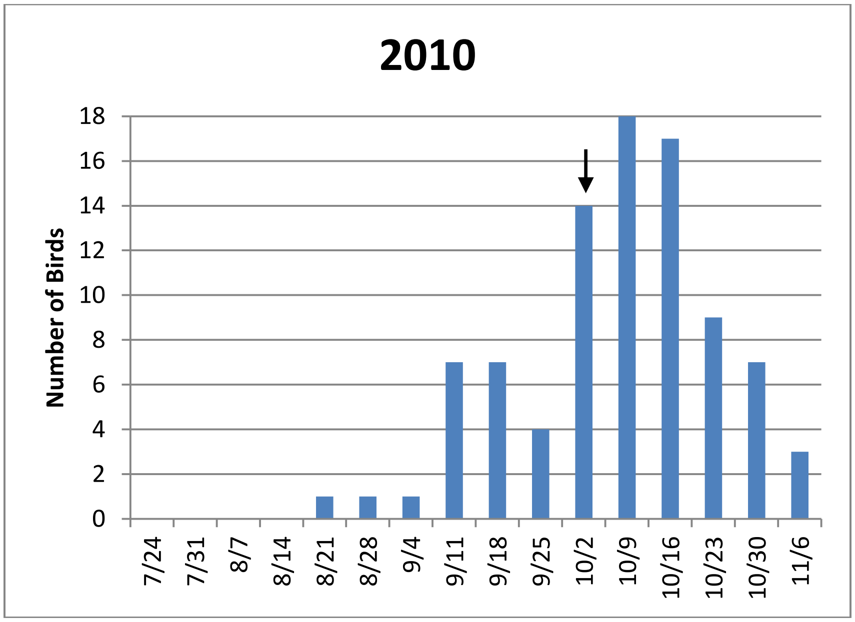

Figure 2. Continued

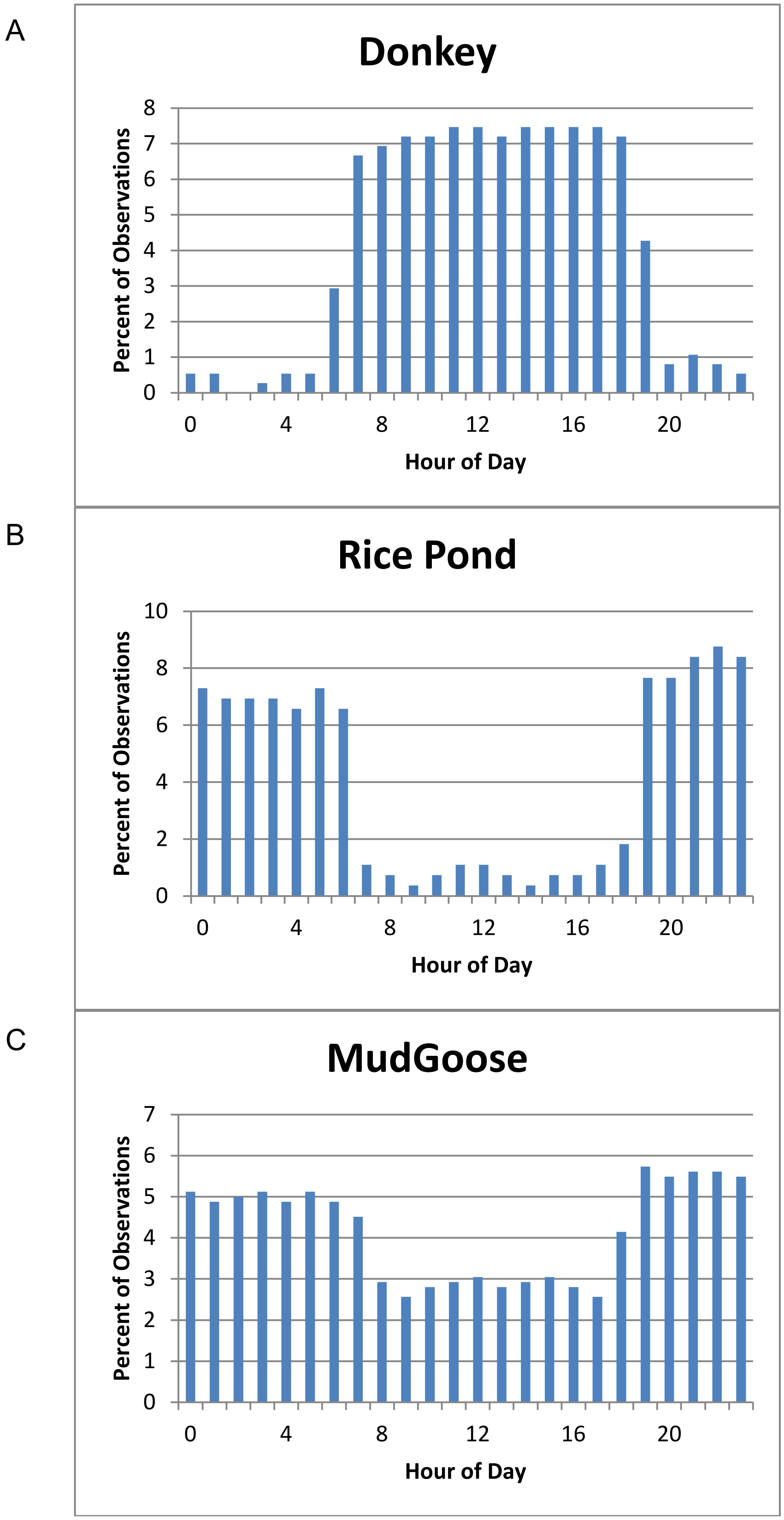

Figure 3. Examples of data depicting refuge use to show the difference among day use (A), night use (B), and 24-hour use (C) of refuges in Minnesota, 2007-2010.

INVESTIGATION OF TREMATODES AND FAUCET SNAILS RESPONSIBLE FOR LESSER SCAUP AND AMERICAN COOT DIE-OFFS

Charlotte Roy and Christine Herwig

SUMMARY OF FINDINGS

Trematodiasis was first reported in Lake Winnibigoshish in the fall of 2007 when 7,000 lesser scaup *(Aythya affinis)* and a few hundred American coots (*Fulica americana*) died. Since then, thousands more birds have died from trematodiasis, which they acquired from ingesting faucet snails (*Bithynia tentaculata*) infected with trematodes. Faucet snails are an invasive snail from Europe, which serve as a host to the trematodes. We have been studying the trematodes, faucet snails, and lesser scaup in lakes and rivers in Minnesota that have been recently designated as infested with faucet snails; including the Shell River and Winnibigoshish, Sugar, Upper Twin, and Lower Twin lakes. We also sampled locations where lesser scaup were observed foraging at Thief, Round, and Bowstring lakes. We observed birds with trematodiasis on Bowstring and Round lakes, where the faucet snail has not yet been detected. Bowstring is 11.3 km northeast and Round is 9.7 km north of Lake Winnibigoshish. We are also monitoring these lakes for snail presence each year. This study aims to examine the factors that influence faucet snail abundance and distribution, parasite prevalence within snails, and food and site attributes (e.g. water depth, distance from shore, substrate composition) of lesser scaup foraging locations.

INTRODUCTION

During the autumns of 2007 and 2008, thousands of lesser scaup and hundreds of American coots died on Lake Winnibigoshish in north-central Minnesota. These deaths were attributed to trematodiasis caused by non-native intestinal trematodes (*Cyathocotyle bushiensis*, *Sphaeridiotrema globules,* and *Leyogonimus polyoon)* and concerned both waterfowl hunters and non-consumptive users.

The trematode species responsible for the die-offs have a complex life cycle that involves 2 intermediate hosts. The faucet snail, a non-native species from Europe (Sauer et al. 2007), is the only known first intermediate host of these trematodes in the Midwest and also serves as the second host for *C. bushiensis* and *S. globules.* The second host of *L. polyoon* is 1 of a variety of larval aquatic insects, including damselflies (Zygoptera) and dragonflies (Odonata) (National Wildlife Health Center, unpublished data). Adult trematodes develop in waterfowl after they consume infected snails and in American coots (*Fulica americana*) and common moorhens (*Gallinula chloropus*) after consumption of infected insects. Parasite eggs are then defecated by sick birds and later ingested by snails, continuing the cycle. Because of this complex life cycle, the dynamics of faucet snail distribution and transmission of these parasites to lesser scaup and other birds are poorly understood.

The first U. S. detection of the faucet snail was in Lake Michigan in 1871 (Mills et al. 1993). It has since been documented in the mid-Atlantic states, the Great Lakes Region, and Montana, and undoubtedly will continue to spread (Sauer et al. 2007). In 2002, the faucet snail was detected in the Upper Mississippi River. Since then, trematodiasis has killed an estimated 52,000-65,000 waterbirds, primarily lesser scaup and American coots, but also dabbling ducks such as blue-winged teal (*Anas discors*), northern shoveler (*Anas clypeata*), mallard (*Anas platyrhynchos*), American black duck (*Anas rubripes*), and northern pintail (*Anas acuta*); diving ducks such as ring-necked ducks (*Aythya collaris*) and redheads (*Aythya americana*); and other waterfowl such as ruddy ducks (*Oxyura jamaicensis*), buffleheads (*Bucephala albeola*), and tundra swans (*Cygnus columbianus,* R. Cole, USGS National Wildlife Health Center, personal communication).

The faucet snail was detected in Lake Winnibigoshish in the spring of 2008, following the loss of 7,000 lesser scaup and a few hundred coots to trematodiasis the previous fall (Lawrence et al. 2008). In 2008, 2,000 more birds died (Lawrence et al. 2009). The severity of the outbreaks seems to have lessened in Lake Winnibigoshish over time, but it is not known whether this is because fewer birds are stopping over on the lake during migration or if there is another explanation for the reduction in the number of birds observed dead and dying. In the fall of 2009, very few lesser scaup were observed on the lake, which is consistent with declining use of this lake, despite its historical importance for migrating scaup. In 2009, the faucet snail was detected in the Twin lakes and the Shell River, which have been designated by the Minnesota Department of Natural Resources (MNDNR) as Infested Waters (MNDNR 2010). Most recently, Leech Lake River (downstream of Mud Lake), Cut Foot Sioux Lake, Egg Lake, First River Lake, Little Cut Foot Lake, Little Winnibigoshish, portions of the Pigeon River, Rabbits Lake, Raven Lake, Ravens Flowage, Sugar Lake, and Third River (downstream of Highway 33) were designated Infested Waters (MNDNR 2010). All of these water bodies share some connectivity with Lake Winnibigoshish, which also has been designated.

OBJECTIVES

1. Improve understanding of lesser scaup foraging as it relates to faucet snail and other food source distribution and density, water depth, distance from shore, and substrate composition;
2. Examine factors (e.g., temperature, substrate, vegetation, other snail species) that are associated with the distribution and movement of faucet snails;
3. Examine factors that influence the prevalence of parasites in faucet snails (e.g., snail density, temperature, microhabitat, time of year); and
4. Examine how faucet snail distribution varies during spring, summer, and fall.

METHODS

Between 8 October and 18 November 2010, we sampled for snails at the Shell River and Winnibigoshish, Sugar, Bowstring, Round, Upper Twin, Lower Twin, and Thief lakes (Figure 1, Table 1). In sum, 15 transects were sampled. We also sampled locations where scaup were observed foraging on 1 occasion each at Thief Lake (1 point), Round Lake (4 points), and Bowstring Lake (5 points). Snail transects were 1,000 m in length, with sampling locations at 100-m intervals. The large size of Winnibigoshish, Round, Bowstring, and Thief lakes easily accommodated 1,000 m transects without overlap of sampling locations. Because of the small size of the Upper and Lower Twin lakes, we laid transects parallel to each other to prevent transect overlap and to allow for the greatest coverage. We also wanted to obtain maximal coverage of the length of the Shell River between Lower Twin and the Shell River Campground, which was known to be infested with faucet snails, so we placed sampling locations 500 m apart and sampled the entire distance at consistent intervals.

Starting locations for snail transects were determined by taking a random direction from the lake center and finding the point of intersection between the bearing and the shoreline. Transects were placed approximately perpendicular to shore at the point of intersection using ArcMap Version 9.3.1 (Environmental Systems Research Institute, Inc., Redlands, California). At the Twin lakes, subsequent transects were placed parallel to the initial transect as noted above, rather than determining additional random starting points, to ensure that sampling transects did not overlap each other or intersect.

When scaup were sighted on a lake, we first confirmed foraging with a scope or binoculars and visually estimated the number of birds. Foraging was identified crudely as 20%

or more of a flock diving without provocation (e.g., raptors, planes, or other sources of disturbance). After confirmation of foraging, we used a compass to determine a bearing from our location to the center, left, and right extremes of the raft of birds. This technique was repeated 2 more times from 2 new locations. These 3 Global Positioning System (GPS) points and 9 bearings were then used to determine an area that represented the location of the raft of birds. Thus, when we approached and the birds flushed from their original locations, we could still be certain that we were sampling the lake bottom in the areas where they had been foraging. Once an area was identified for the flock location, a transect was placed across the longest axis of that area and points were spaced 100 m apart such that we maximized the number of points within the delineated area. The flock at Thief Lake was small (~50 birds) and only 1 sampling point was established. At Round Lake, the flock was located on the snail transect and so we extended the snail transect to include the scaup points and just beyond (i.e., 17 points sampled).

Sampling for snails at 30 and 60-cm water depths was done with a bottomless sampling cylinder (0.2 m^2). All organisms within the bottomless cylinder were collected with a dip net for identification. For snail and scaup sampling, a kick net (457 x 229 mm) was generally used for water depths of < 150 cm and a benthic sled (305 x 508 mm) was generally used in waters >150 cm. Both the kick net and benthic sled were dragged 1.2 m through the water along the substrate, and the samples were collected for later identification. At all sampling locations an artificial substrate (i.e., a clay brick) with a floating marker was left in place for 8-28 days. At each snail and scaup sampling location, we recorded substrate (e.g., silt, rock, sand, vegetated, muck), surface and bottom temperatures (˚C), water depth (cm), and a secchi depth (cm) reading was taken 8 times (4 times on the way down and 4 times on the way up) from the shaded side of the boat and averaged. Salinity ($^{o}/_{oo}$), conductivity (µS/cm), pH, and dissolved oxygen (mg/L) were measured with a Hach HQd (Hach Company, Loveland, Colorado) portable meter that was calibrated daily for pH and weekly for conductivity. Flow (mps) was measured at 60% of the total depth (from the surface) with a Global Water Flow Probe (Global Water Instrumentation, Inc, Gold River, California) when flow was detectable ($\geq$0.1 m/s) and averaged over a 40-s interval (the USGS “6 tens method,” www.globalw.com/downloads/flowprobe/flowprobe_manual_past.pdf).

We stored samples in the refrigerator at 3-5 ˚C until processed. We used a magnifying lens and microscope as needed to identify all invertebrates to Order and noted their presence in each sample. We identified all snails to genus and counted their numbers in each sample. We determined the size of *B. tentaculata* and *Amnicola* spp., a similar species, with digital calipers, as measured along the central axis from the apex.

Parasite prevalence was determined for all samples containing at least 50 *B. tentaculata* (R. Cole, USGS National Wildlife Health Center, unpublished data). We recorded Trematode stages (cercariae or metacercariae), species (*C. bushiensis, S. globules, L. polyoon*), and numbers.

At each sample lake, we collected a water sample that was sent to the Minnesota Department of Agriculture for analysis. Total phosphorus (ppm), nitrite plus nitrate nitrogen (ppm), chlorophyll a (ppb), total alkalinity (ppm), ammonia nitrogen (ppm), and calcium (ppm) were quantified, but have not yet been interpreted.

RESULTS

Faucet Snails

We detected faucet snails at Lake Winnibigoshish, Upper and Lower Twin lakes, and the Shell River (Table 1). We found 4 suspect snails on Round Lake, but they were too small to make a positive identification from the pattern on the operculum and to distinguish them from *Amnicola* spp.

Faucet snails were found associated with most substrates including detritus/muck, rocks, sand, silt, mud, and vegetation, but we did not find them associated with the few samples with clay. Early indications were that faucet snails avoided detritus/muck, clay/silt, and mud and seemed to prefer sand, rocks, and vegetation. Snails were located at an average depth of 226 ± 40 cm (range = 30-854 cm).

Trematodes

We detected both *C. bushiensis* and *S. globules* at Winnibigoshish and Lower Twin lakes and the Shell River. Although faucet snails were detected in Upper Twin Lake, none of the samples had 50 *B. tentaculata,* so prevalence was not determined this season. Both primary (cercariae) and secondary (metacercariae) infections were observed at all 3 locations. Determination of parasite prevalence is still underway, but appears to be much higher at Lake Winnibigoshish than at Lower Twin Lake and the Shell River.

We also detected *S. globules* metacercariae in 5 of 40 *Physa* snails and 2 of 22 *Amnicola* snails examined from Lake Winnibigoshish. Initial indications are that parasite prevalence was much higher in *B. tentaculata* samples than in these other species of snails from the same lake. To date, we have not discovered primary infections in other snail species.

Scaup

We observed scaup foraging at Thief, Bowstring, and Round lakes, and samples were collected under these flocks. On October 13 at Thief Lake, 40-64 scaup were observed ~1.35 km from shore (as measured to the center of the flock) in 116 cm of water with submergent vegetation. Samples collected below the scaup included snails of the genera *Physa* and *Lymnaea*, as well as the additional invertebrate Orders of Amphipoda, Diptera, Hemiptera, Trichoptera, Trombidiformes, and Veneroida. On 21 October at Bowstring Lake, we observed approximately 8,800 scaup ~1.38 km from shore in 952 cm of water over a substrate of silt and mud. Snails of the genera *Amnicola* and *Valvata* were identified in the samples. The subclasses Hirudinea and Oligochaeta, as well as the Orders Diptera, Trombidiformes, and Veneroida also were found in the samples. On 1 November at Round Lake, 2,000-3,000 scaup were observed ~1.34 km from shore in 562 cm of water over a silt substrate. Only *Physa* spp. snails were found, but other invertebrate Orders were identified, including Amphipoda, Cladocera, Diptera, Ephemeroptera, Hemiptera, Trombidiformes, and Veneroida. Faucet snails were not detected at these sampling locations.

A large raft also was observed at Lake Winnibigoshish (S. Cordts, personal communication), but we did not observe the raft nor any sick birds during our visits. We observed sick scaup at both Bowstring and Round lakes during our sampling (Table 2). Colleagues from the Grand Rapids MNDNR office observed healthy and sick birds during their visits to Winnibigoshish, Bowstring, and Round lakes, and these observations are included in this summary (Table 2). Diagnostics from carcasses sent to the National Wildlife Health Center in Madison, Wisconsin indicated that the birds were positive for *S. globules* and *C. bushiensis*.

DISCUSSION

We did not observe notable die-offs on Winnibigoshish in the fall of 2010. We could not determine whether this was because the birds were there very briefly and thus had limited exposure to the snails and trematodes, infected birds departed to nearby areas (like Bowstring and Round lakes, Figure 1) before becoming symptomatic, or other factors were at play. We did observe sick birds at Bowstring and Round lakes despite not detecting *B. tentaculata* there. Previous sampling efforts at Bowstring Lake by J. Lawrence and P. Loegering were more exhaustive, but also did not discover *B. tentaculata*. Sick birds have been observed at Bowstring in the last few years (Roy and other MNDNR staff, personal observation). To our knowledge, this was the first record of sick birds from Round Lake. We will continue to monitor these lakes for faucet snails and scaup use. Several additional water bodies that contain faucet snails have now been designated as Infested Waters and will be considered for inclusion in this study.

This report summarizes activities for the first field season of 7 planned seasons (3 fall, 2 spring, 2 summer). Data entry and analysis is preliminary and still underway. Data from the first field season will be used to improve data collection in upcoming field seasons.

LITERATURE CITED

Lawrence, J. S., S. D. Cordts, R. A. Cole, and P. Loegering. 2008. Determining gastropod hosts of trematodes and monitoring lesser scaup and coot mortality in north central Minnesota. Unpublished Phase 1 Proposal, Minnesota Department of Natural Resources, Wildlife Populations and Research Unit, St. Paul.

Lawrence, J. S., P. Loegering, R. A. Cole, and S. D. Cordts. 2009. Scaup and coot die-off at Lake Winnibigoshish-2008: an update. Unpublished report. Minnesota Department of Natural Resources, Wildlife Populations and Research Unit.

Mills, E. L., J. H. Leach, J. T. Carlton, and C. L. Seacor. 1993. Exotic species in the Great Lakes: a history of biotic crises and anthropogenic introductions. Journal of Great Lakes Research 19:1–54.

Minnesota Department of Natural Resources. 2010. Designation of Infested Waters, 12-28-10. Retrieved March 31, 2010, from http://files.dnr.state.mn.us/eco/invasives/infested_waters.pdf

Sauer, J. S., R. A. Cole, and J. M. Nissen. 2007. Finding the exotic faucet snail (*Bithynia tentaculata*): investigation of water bird die-offs on the Upper Mississippi River National Wildlife and Fish Refuge. U. S. Geological Survey Open-File Report 2007–1065. 3pp.

Table 1. Sampling sites for faucet snails and scaup presence in northern Minnesota during fall 2010.

Location	No. snail transects	No. sample points[a]	Faucet snails detected	Scaup observed foraging during sampling efforts
Winnibigoshish	3	34	Yes	Present[b]
Sugar	1	10	No	No
Bowstring	2	31	No	Yes
Round	2	34	Possibly[c]	Yes
Upper Twin	2	24	Yes	No
Lower Twin	3	39	Yes	No
Thief	1	14	No	Yes
Shell River	1	22	Yes	No
Total	15	208		

[a] Including snail, random, scaup, and 30-cm and 60-cm sample points.

[b] Scaup were present during the season, but a raft was not observed during sampling.

[c] Four snails were collected that could have been *B. tentaculata*, but they were too small to see the operculum pattern to distinguish them from *Amnicola* spp.

Table 2. Time-line of reports of sick scaup and coots observed by Minnesota Department of Natural Resources staff or the authors on lakes in northern Minnesota during fall 2010.

Date	Location	Total no. of birds observed	No. of dead or sick birds observed
10/7/10	Winnibigoshish	No flocks of scaup	1 dead scaup, 1 sick scaup
10/7/10	Rabbit Flowage (on Winnibigoshish)	500-1000 coots	No sick birds
10/7/10	Third River Flowage (on Winnibigoshish)	200 coots	No sick birds
10/18/10	Bowstring	Raft of scaup in middle	8 dead scaup, 25 sick scaup
10/20/10	Third River Flowage (on Winnibigoshish)	Few scaup	2 dead coots at Mallard Point, no sick or dead scaup
10/21/10	Bowstring	8,800 scaup	100s of sick scaup
10/25/10	Bowstring	Large raft of scaup	20 dead, 6 sick scaup
10/25/10	Round	Not reported	12 dead scaup
11/1/10	Round	2,000-3,000 scaup	<50 sick scaup
11/1/10	Bowstring	Not reported	53 dead scaup on 142 m of shore (Area Wildlife Manager estimated fall mortality of 1,200 scaup on 8,360 m of shore)
11/3/10	Winnibigoshish	Few small flocks of birds	No sick birds
11/5/10	Bowstring	Not reported	Duck hunter reported 20-30 dead scaup
11/8/10	Bowstring	Large raft (5,000) of scaup	7 dead scaup
11/8/10	Round	2,000-3,000 scaup	No sick birds
11/15/10	Bowstring	200 scaup	6 dead scaup
11/16/10	Round	2,000 scaup	Not foraging, did not approach raft

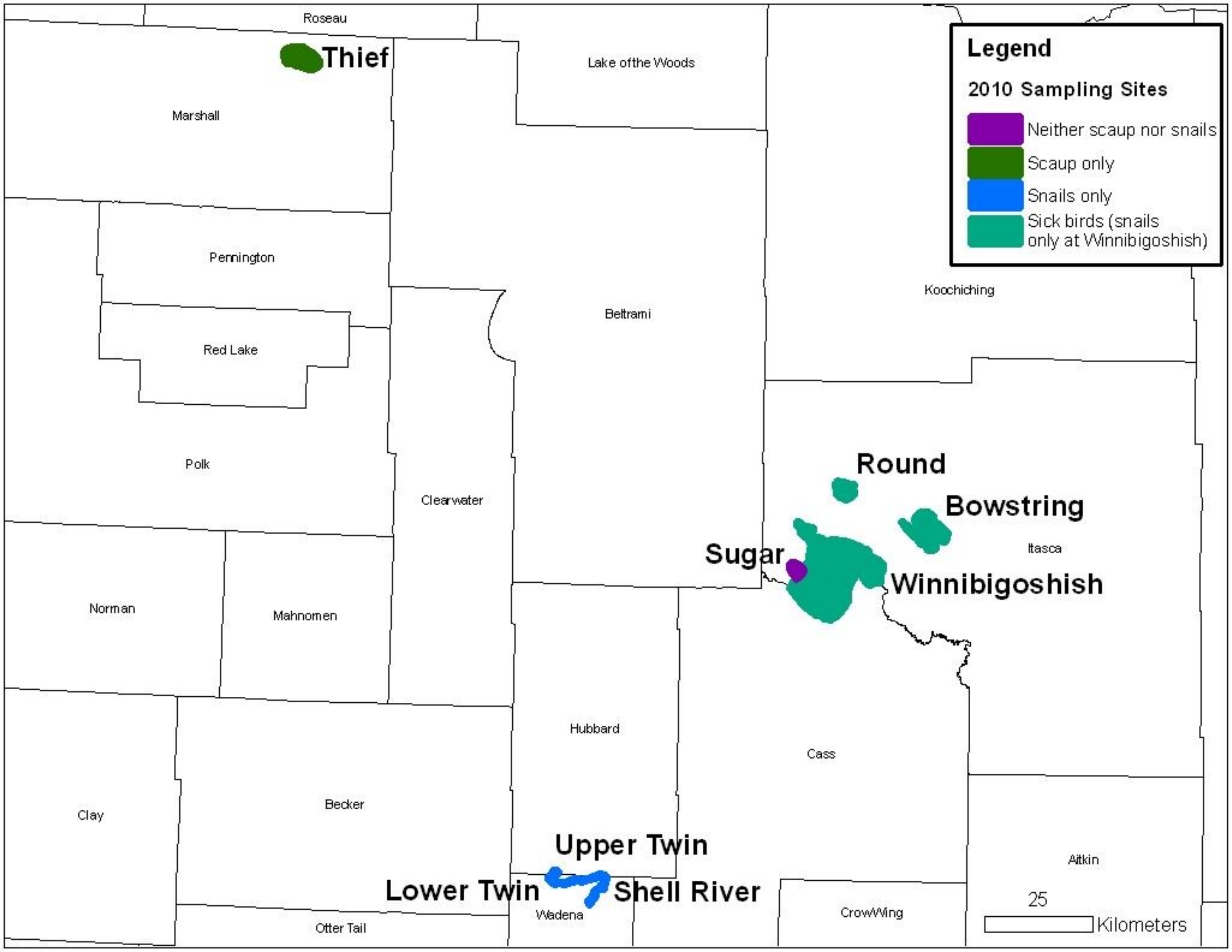

Figure 1. Lakes sampled for faucet snails and scaup presence in northern Minnesota in fall 2010.

COMPARISON OF NATIVE GRASSLAND MANAGEMENT TREATMENTS TO SPRING PRESCRIBED BURNS

David Rave, Kevin Kotts, and John Fieberg

SUMMARY OF FINDINGS

We conducted a pilot study in 2008 to measure the response of restored native grasslands to: (1) grazing, (2) fall biomass harvest, and (3) spring prescribed burning. Among field variability was substantial in the pilot study, which indicated the need to control for this variability when making treatment comparisons. Therefore, in 2009, we dropped the grazing element of the study, and added 6 additional sites using a split-plot design, in which matched subplots were biomass harvested in fall 2008, or burned in spring 2009. This gave us a total of 10 sites with fall biomass harvest and spring burned subplots. Fields were located on Wildlife Management Areas (WMAs) or Waterfowl Production Areas (WPAs) in Working Lands Initiative Focus Areas of Chippewa, Grant, Kandiyohi, Lac Qui Parle, Renville, Stevens, and Swift counties in 2008. For 2009, to increase sample sizes, we included some fields within Minnesota's Prairie area, but outside of Focus Areas. We conducted visual obstruction measurements, Daubenmire frame analysis, and we measured litter depth and vegetation height in all study fields. We also examined temporary and seasonal wetlands in bioharvested fields and recorded wetland type, and waterfowl presence. Biomass harvested and burned subplots appeared similar in most vegetative characteristics in both 2008 and 2009. In 2010, we again surveyed vegetation in all plots in which biomass harvest/burn treatments were applied. Vegetation response continued to be similar between biomass harvested and burned subplots, thus, we determined there was no need to continue the project in 2011.

INTRODUCTION

Minnesota's Department of Natural Resources' (MNDNR) Draft Grassland Biomass/Bioenergy Harvest on WMAs and Aquatic Management Areas (AMAs) management document states, "Grassland biomass harvest from WMAs and AMAs shall be in concert with fish and wildlife habitat management activities, consistent with the habitat or wildlife species management goals and habitat management objectives for each individual WMA/AMA." Further, Sample and Mossman (1997) found that differences in habitat structure are likely more important to bird communities than differences in vegetative species composition. They recommend that the following features of grassland habitat are important to grassland nesting birds: vegetation height and density, height and cover of woody vegetation, litter depth and cover, standing residual (dead) and live herbaceous cover, and ratio of grass vs. forb cover. However, the response of native grassland stands on WMAs and AMAs to grassland biomass harvest is unknown. We conducted this study with the following objectives:

- to determine vegetative response to biomass harvest,
- to determine whether vegetative response to fall biomass harvest is similar to vegetative response to spring controlled burning, and
- to determine whether fall biomass harvest can be used by wildlife managers to maintain restored prairie grasslands.

STUDY AREA

The study was conducted in Chippewa, Grant, Kandiyohi, Lac Qui Parle, Renville, Stevens, and Swift counties, within the prairie portion of Minnesota (Figure 1), and was targeted at Working Lands Initiative (MNDNR unpublished brochure, http://files.dnr.state.mn.us/

assistance/backyard/privatelandsprogram/working-lands-ini.pdf) Focus Areas. We could not find enough study fields close enough to biomass harvesters within Working Lands Initiative Focus Areas in 2009; therefore, to increase sample size, some fields in 2009 were located outside Working Lands Initiative Focus Areas, but all were within the prairie portion of Minnesota. Fields sampled were all located on state-managed WMAs or federally managed WPAs. Sites in 2009 consisted of 9 fields with bioharvest and burn subplots, and 6 sites with only a bioharvest subplot. Spring burns on these latter 6 fields were not accomplished.

METHODS

We compared the response of restored native grasslands to fall biomass harvest (hayed) and spring prescribed burning (control) using paired subplots and a split-plot design (Steel et al. 1997). Visual obstruction measurements (VOMs, Robel et al. 1970) were taken every 2 weeks from mid-May through mid-August in hayed and control subplots of each field following methods described by Zicus et al. (2006). Three VOM sample stations were established at the 3 quarter points along the longest straight-line transect across each subplot within a field (hereafter the VOM transect). GIS locations were permanently marked with stakes to define starting and sampling points along the VOM transect. Each station had 4 sampling points located 20 m north, east, south, and west of a starting point. At each field sampling point, vegetation height and density was measured in each cardinal direction. This provided 48 VOMs for each treatment from each field on a given date.

A Daubenmire square (Daubenmire 1959) was used to determine coverage by various species across hayed and burned subplots. We sampled at 10 locations along the VOM transect in all subplots of each field every 2 weeks. The 1m^2 Daubenmire frame was placed on the ground approximately 10 meters from the VOM transect every tenth of the entire transect distance determined using a GPS. Each plant species (and % coverage within the frame) within the frame was recorded.

Litter depth (nearest 1mm) and vegetation height (nearest 0.5 dm) were also measured at 10 locations, each 1 tenth of the entire transect distance as determined using a GPS, on the VOM transect in all subplots of each field every 2 weeks. While walking the VOM transect, all exotic and woody species present were recorded.

We also examined seasonal and temporary wetlands in mid-April that had vegetation removed, primarily cattails, during biomass harvest in fall 2007. For each wetland, we recorded wetland type (Stewart and Kantrud 1971), waterfowl numbers, and waterfowl pair status.

RESULTS

Vegetative characteristics were largely similar in hayed and burned subplots (Figures 2a – 6b). The most notable exception was Klason in 2008. At this site (in 2008), vegetation was taller (with larger VOM readings), litter depth was greater, and a higher number of species were located in the hayed treatment subplot than the burned subplot; however, these differences were largely absent the next year. In 2009, litter depths again varied in subplots hayed in fall 2008 and burned in spring 2009 (Beaver Falls WMA, Danvers WMA, Lac Qui Parle WMA, and Towner WMA), whereas other vegetative characteristics were similar between treatment subplots. By 2010, litter depths were similar among treatments in all subplots.

We examined 12 seasonal and temporary wetlands in mid-April that had been at least partially harvested during the biomass treatment in fall 2007. Cattail growth in summer of 2008 filled in these wetlands, and there were no waterfowl pairs using the wetlands in spring 2009 or 2010.

DISCUSSION

Recently, the cost of fossil fuels has increased as their supply tightened. Alternative sources of energy are being sought. Wind, solar, and other renewable energy sources are being developed. One potential source is biomass energy derived from agricultural or other cellulose residues. Based on estimates from 2005, there is approximately 194 million tons of biomass available each year from the agricultural sector (Perlack et al. 2005). However, the United States Department of Agriculture projects that to replace 30% of petroleum use by 2030 will require over 1 billion tons of biomass. To acquire this amount of biomass, new sources of biomass will need to be developed. One possible source of biomass is native grass. However, the effects of biomass harvest on vegetation in native grass fields and the birds that nest in those fields are unknown.

The Minnesota Department of Natural Resources acquires and manages Wildlife Management Areas primarily to establish and maintain optimal population levels of wildlife while maintaining ecological diversity; maintaining or restoring natural communities and ecological processes; and maintaining or enhancing populations of native species (including uncommon species and state- and federally-listed species; The Draft Grassland Biomass/Bioenergy Harvest on WMAs & AMAs directive, unpublished MNDNR publication). Prior to settlement and implementation of agriculture, natural disturbance in the form of fire and grazing maintained native grassland diversity and productivity (Anderson 1990). Wildlife managers have traditionally used spring prescribed burns to simulate these natural disturbances (K. Kotts, personal communication). However, there are a variety of management options available to wildlife managers to create disturbances in native grass stands. These options are not typically the first choice of managers; likely because there is little known about the response of native grass stands to these treatments. Our study was designed to compare the vegetative response of a biomass harvest for disturbing native grass stands, and compare the response to that from a spring controlled burn.

After 3 field seasons, there appears to be little difference in vegetation characteristics between bioharvested and burned subplots. In the first spring after treatment, litter depths were higher in fall biomass harvested than burned subplots. This difference was only evident in the first spring, and litter depths were similar in subsequent years. We conclude that managers can use a fall biomass harvest in restored native fields and expect a vegetative response that is similar to that from a spring prescribed burn.

We found that the removal of wetland vegetation in the fall is a promising way to open choked wetlands, making them available to waterbirds such as dabbling ducks, geese, swans, and shorebirds. Fall wetland conditions play an important role in determining how successful this technique will be. Wetlands must be fairly dry when the haying occurs to allow equipment to harvest vegetation within the wetland basin. Fall 2007 was a dry year, and machinery was able to remove vegetation from dry basins. On the contrary, fall of 2008 was very wet, and machinery was unable to remove emergents from wetland basins in fall 2008. Basins that were harvested in 2007 contained open water areas in spring 2008, and were utilized by migrating and nesting waterfowl. However, cattail growth in summer of 2008 was sufficient enough to eliminate most of the open water in these basins, and they were not utilized by waterfowl in spring 2009 or 2010. Management of these seasonal wetlands would be enhanced greatly if a way could be found to remove the emergent growth from these wetlands for several consecutive seasons with a single treatment. Therefore, we recommend further research of management of emergent vegetation in temporary and seasonal wetlands.

ACKNOWLEDGEMENTS

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C. Vacek, S. Vacek, K. Varland, and J. Zajac helped with study logistics. A. Bochow, H. Curtis, J. Gregory, J. Lennox, L. Plath, and B. Stenberg were interns on this project, and collected most of the field data. The University of Minnesota at Morris and Minnesota Alfalfa Producers harvested grass from our treatment fields.

LITERATURE CITED

Anderson, R. C. 1990. The historic role of fire in the North American grassland. Pages 8-18 *in* S. L. Collins and L. L. Wallace, eds. Fire in North American Tallgrass Prairies. University of Oklahoma Press, Norman.

Daubenmire, R. F. 1959. A canopy-cover method of vegetational analysis. Northwest Science 33:43-46.

Perlack, R. D., L. L. Wright, A. F. Turhollow, R. L. Graham, B. J. Stokes, D. C. Erbach. 2005. Biomass as feedstock for bioenergy and bioproducts industry. The technical feasibility of a billion ton annual supply. U. S. Department of Energy, Document Number: DOE/GO-102995-21 35.

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.

Sample, D. W., and M. J. Mossman. 1997. Managing habitat for grassland birds. A Guide for Wisconsin. Wisconsin Department of Natural Resources, Madison. 154pp.

Steel, R. G. D., J. H. Torrie, and D. Dickey. 1997. Principles and procedures of statistics: a Biometrical approach. Third edition. McGraw-Hill, Boston, Massachusetts.

Stewart, R. E., 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.

Zicus, M. C., D. P. Rave, A. Das, M. R. Riggs, and M. L. Buitenwerf. 2006. Influence of land use on mallard nest-structure occupancy. Journal of Wildlife Management 70:1325-1333.

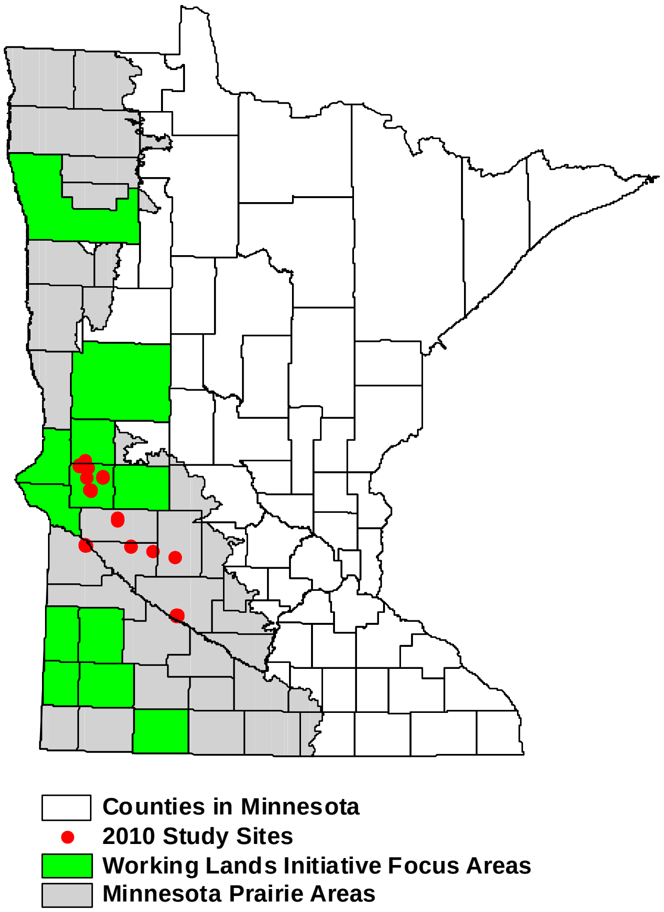

Figure 1. Minnesota counties showing study sites, Minnesota's prairie areas, and Working Lands Initiative focus areas, 2010.

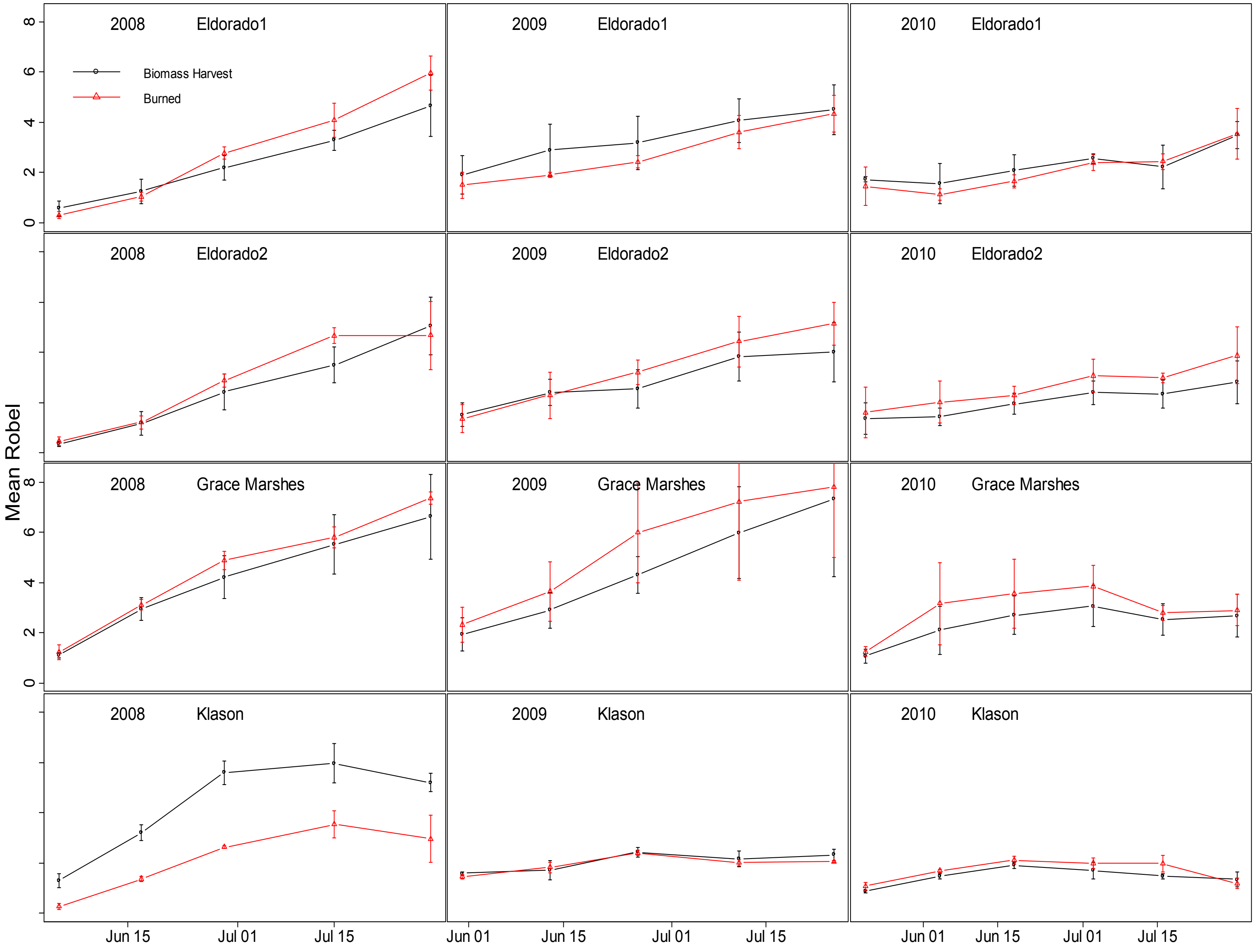

Figure 2a. Comparison of mean Robel measurements (dm) and 95% confidence intervals between 2 treatment subplots (a fall 2007 biomass harvest and a prescribed burn in spring 2008) within the same restored native grass field on 3 State Wildlife Management Areas (2 fields on one area) in west-central Minnesota, in summers 2008 – 2010.

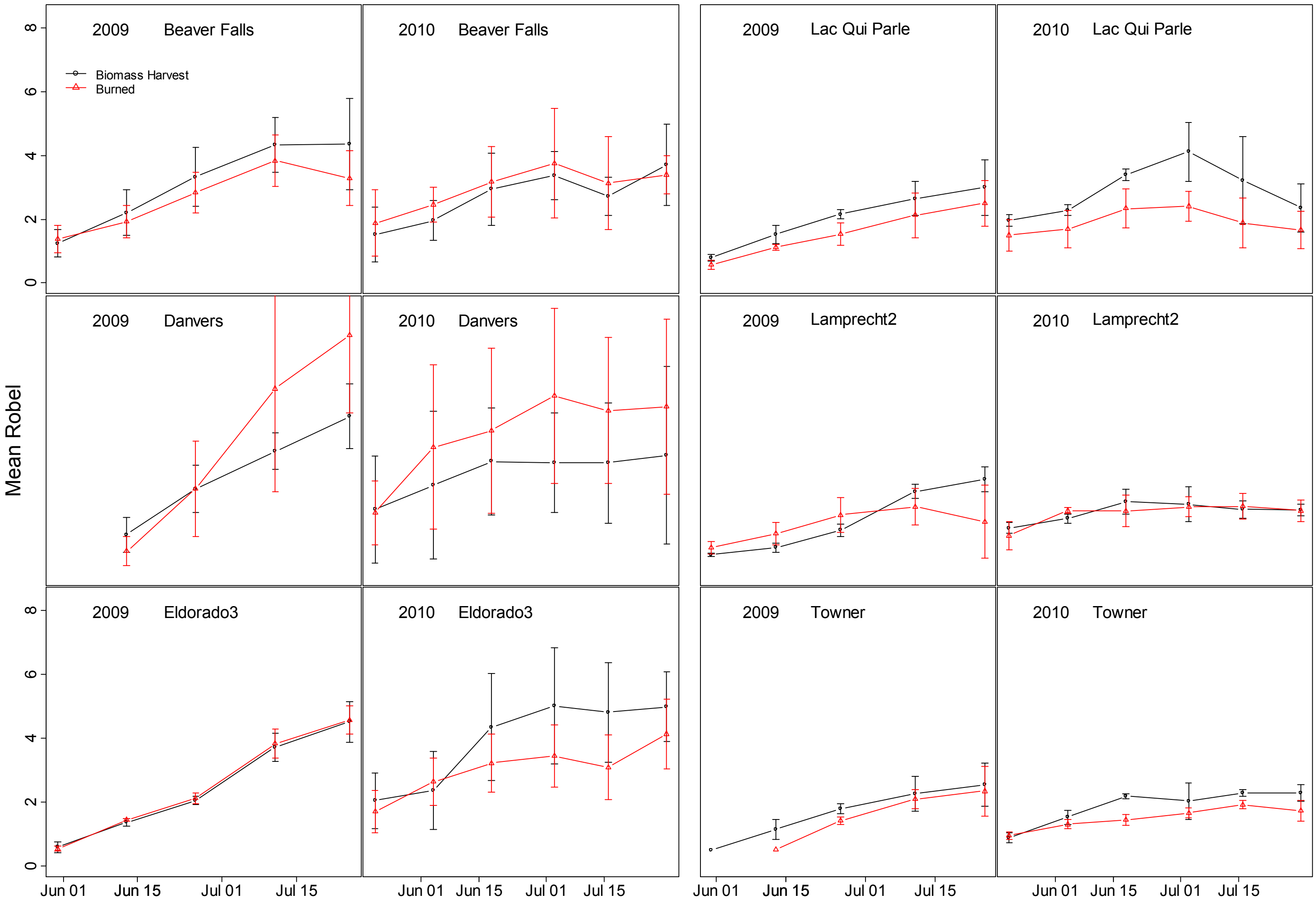

Figure 2b. Comparison of mean Robel measurements (dm) and 95% confidence intervals between 2 treatment subplots (a fall 2008 biomass harvest and a prescribed burn in spring 2009) within the same restored native grass field on 5 State Wildlife Management Areas and 1 Federal Waterfowl Production area in west-central Minnesota, in summers 2008 – 2010.

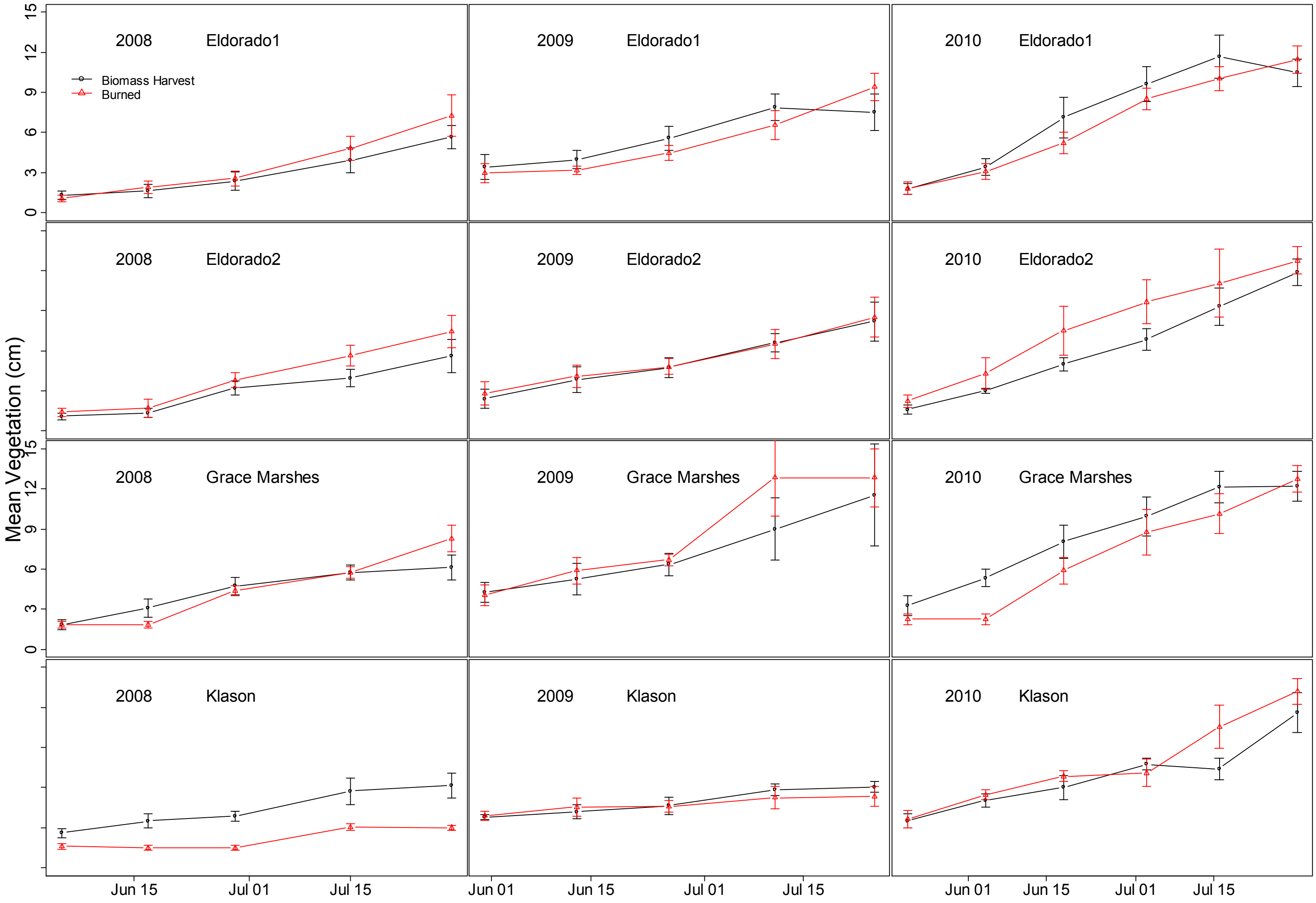

Figure 3a. Comparison of mean vegetation height (dm) and 95% confidence intervals (a fall 2007 biomass harvest and a prescribed burn in spring 2008) within the same restored native grass field on 3 State Wildlife Management Areas (2 fields on one area) in west-central Minnesota, in summers 2008 – 2010.

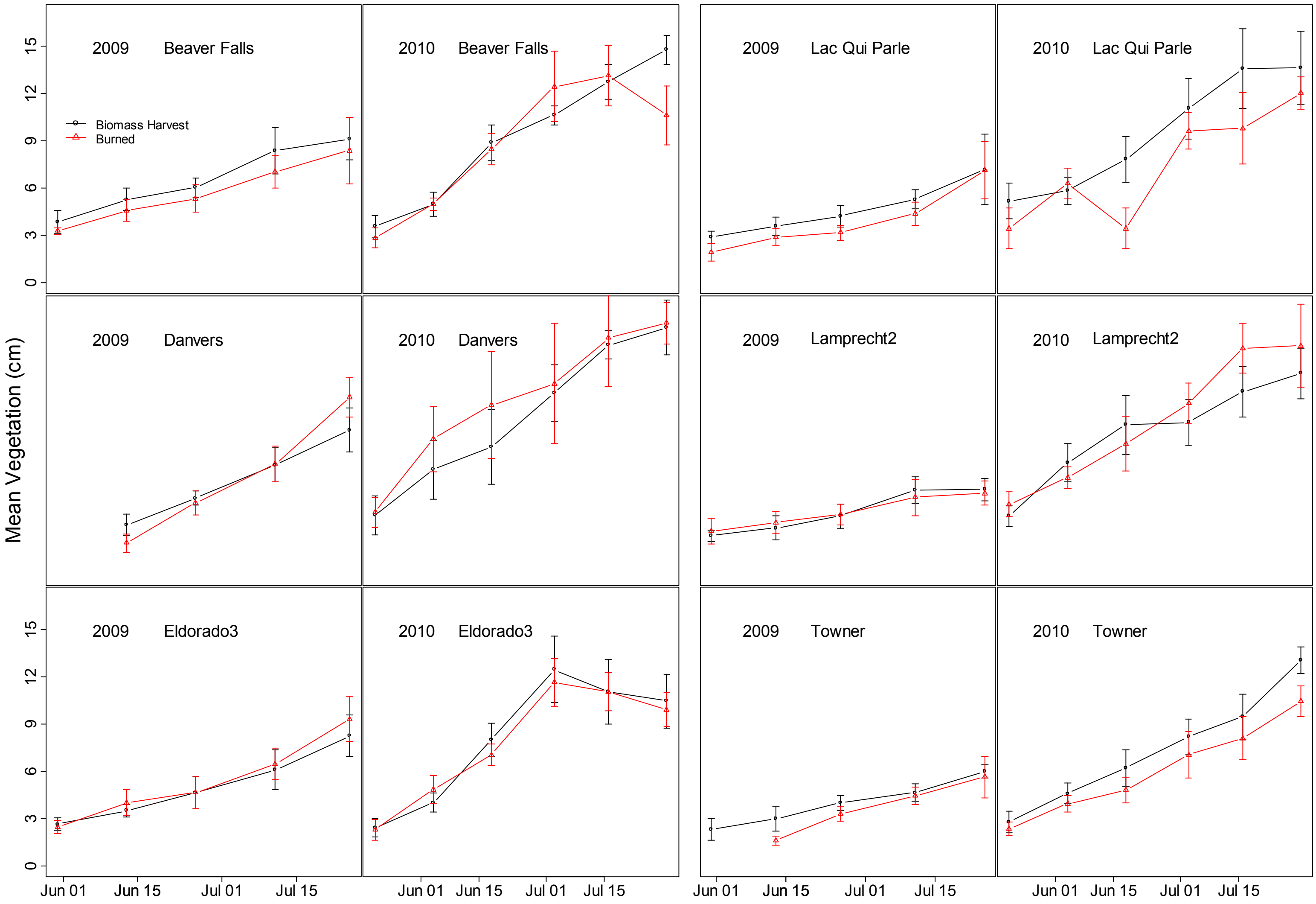

Figure 3b. Comparison of mean vegetation height (dm) and 95% confidence intervals between 2 treatment subplots (a fall 2008 biomass harvest and a prescribed burn in spring 2009) within the same restored native grass field on 5 State Wildlife Management Areas and 1 Federal Waterfowl Production area in west-central Minnesota, in summers 2008 – 2010.

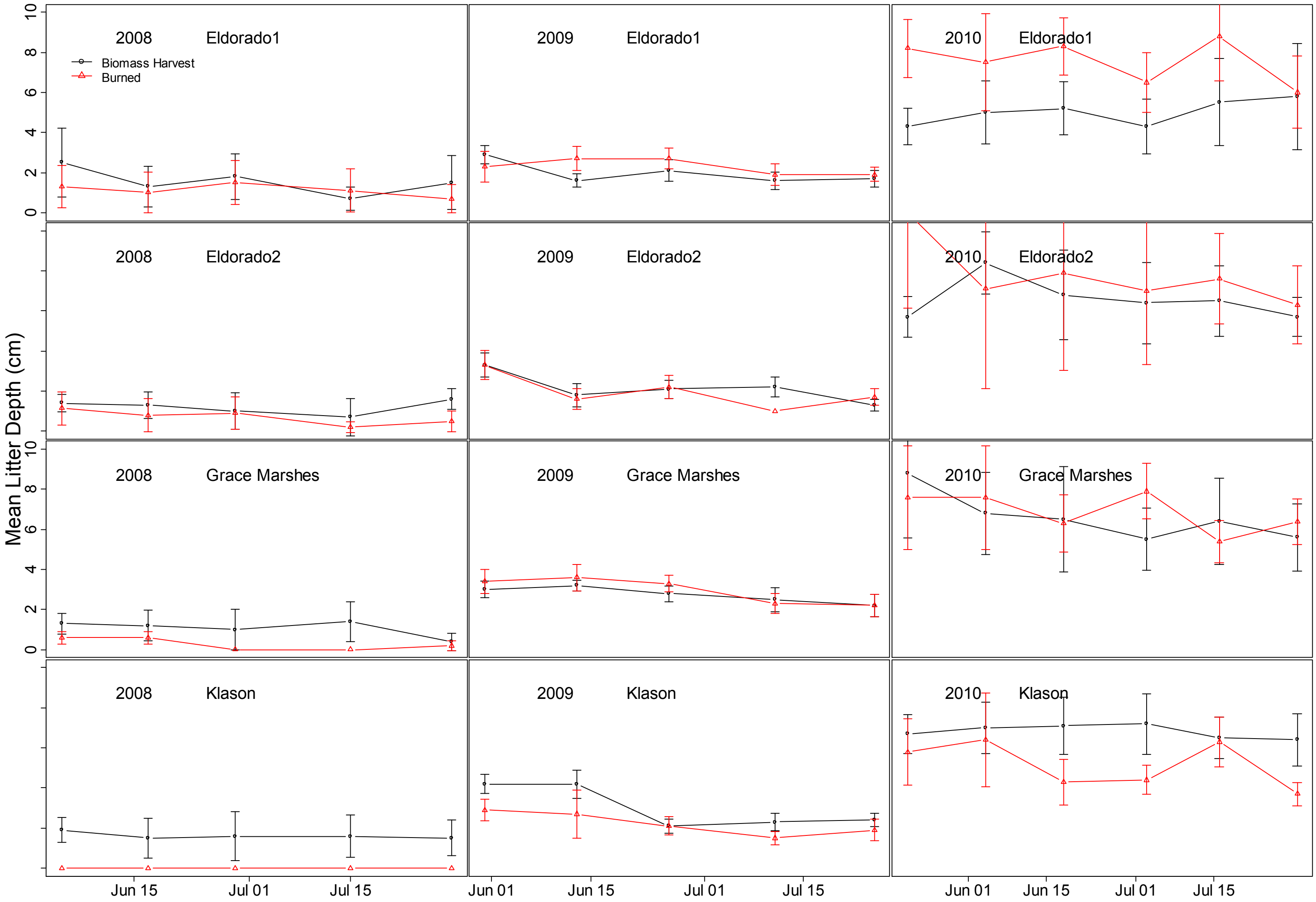

Figure 4a. Comparison of mean litter depth (dm) and 95% confidence intervals between 2 treatment subplots (a fall 2007 biomass harvest and a prescribed burn in spring 2008) within the same restored native grass field on 3 State Wildlife Management Areas (2 fields on one area) in west-central Minnesota, in summers 2008 – 2010.

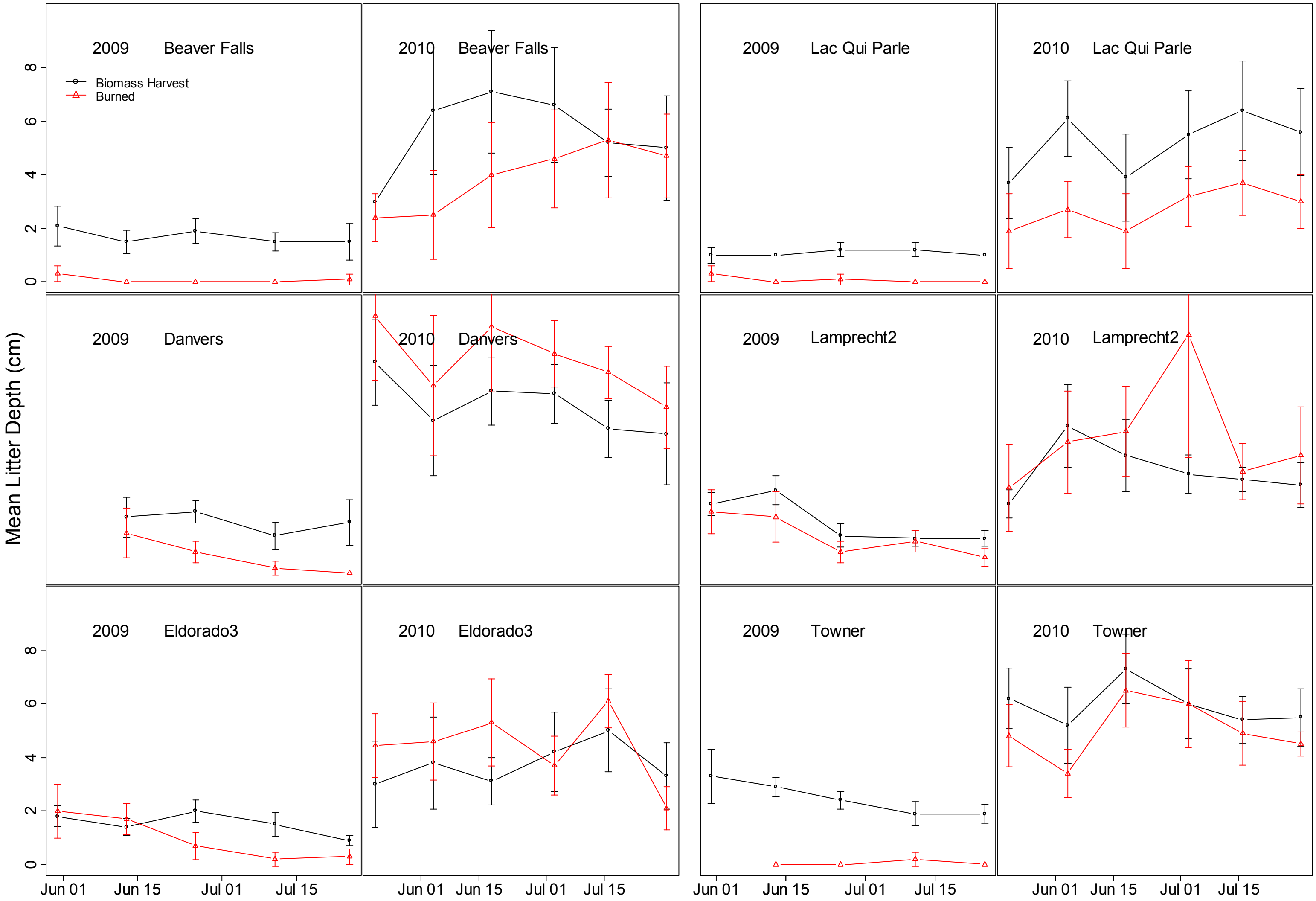

Figure 4b. Comparison of mean litter depth (dm) and 95% confidence intervals between 2 treatment subplots (a fall 2008 biomass harvest and a prescribed burn in spring 2009) within the same restored native grass field on 5 State Wildlife Management Areas and 1 Federal Waterfowl Production area in west-central Minnesota, in summers 2008 – 2010.

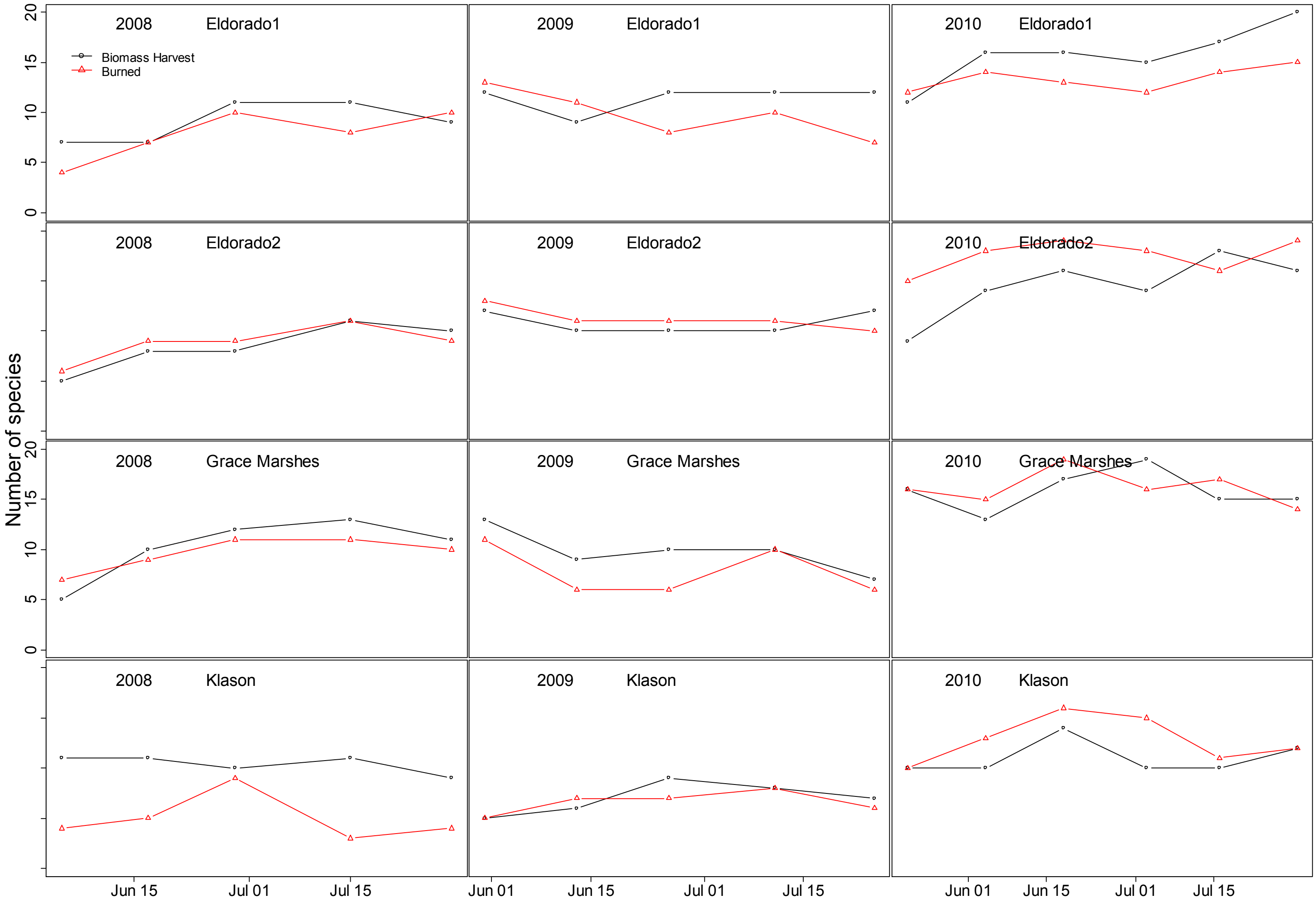

Figure 5a. Comparison of mean number of plant species per transect between 2 treatment subplots (a fall 2007 biomass harvest and a prescribed burn in spring 2008) within the same restored native grass field on 3 State Wildlife Management Areas (2 fields on one area) in west-central Minnesota, in summers 2008 – 2010.

Figure 5b. Comparison of mean number of plant species per transect between 2 treatment subplots (a fall 2008 biomass harvest and a prescribed burn in spring 2009) within the same restored native grass field on 5 State Wildlife Management Areas and 1 Federal Waterfowl Production area in west-central Minnesota, in summers 2008 – 2010.

Figure 6a. Comparison of the percent of native plant species per transect between 2 treatment subplots (a fall 2007 biomass harvest and a prescribed burn in spring 2008) within the same restored native grass field on 3 State Wildlife Management Areas (2 fields on one area) in west-central Minnesota, in summers 2008 – 2010.

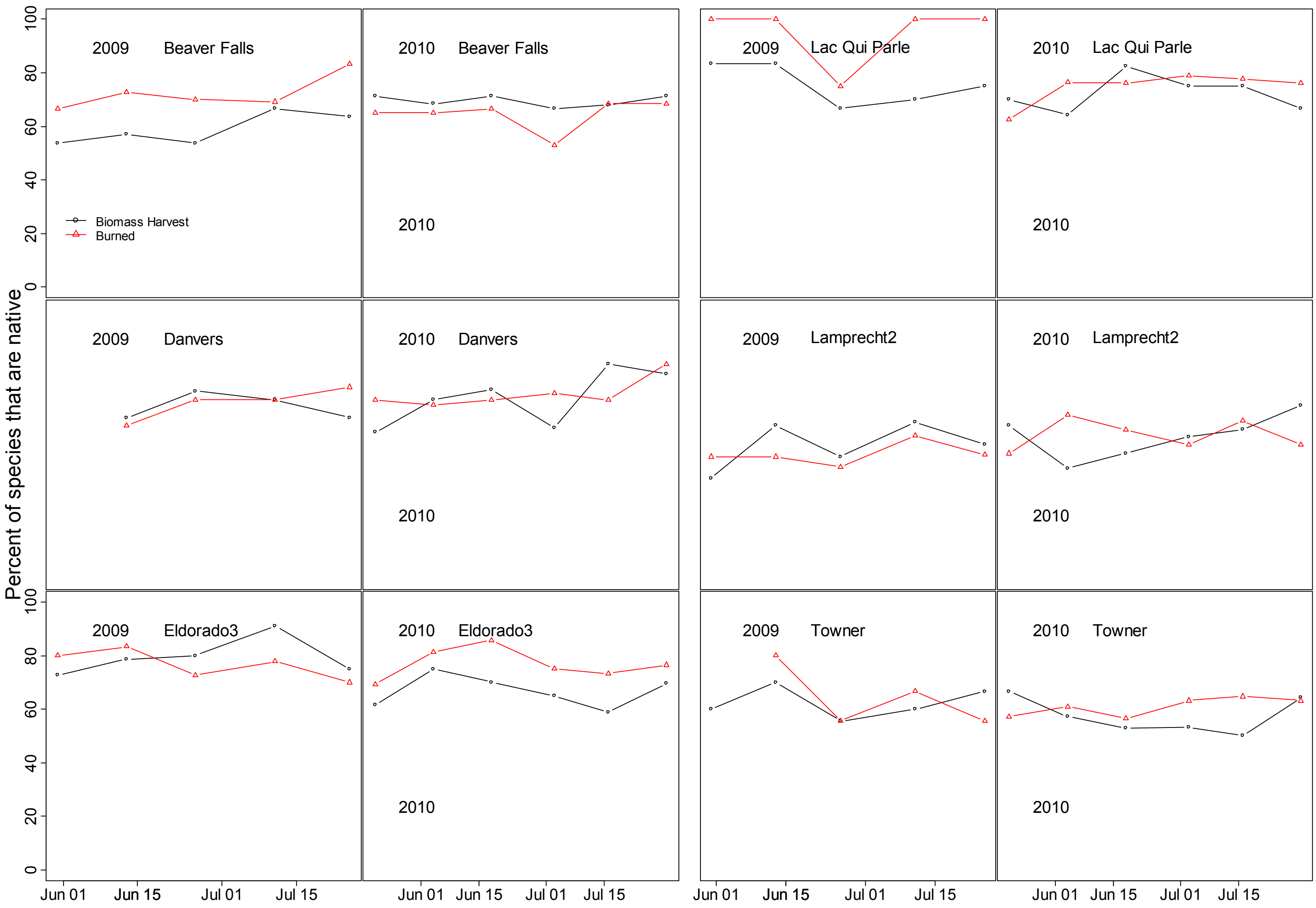

Figure 6b. Comparison of the percent of native plant species per transect between 2 treatment subplots (a fall 2008 biomass harvest and a prescribed burn in spring 2009) within the same restored native grass field on 5 State Wildlife Management Areas and 1 Federal Waterfowl Production area in west-central Minnesota, in summers 2008 – 2010.

FINAL REPORT: HARVEST CHARACTERISTICS OF LARGE CANADA GEESE IN MINNESOTA, 2002–2007

James B. Berdeen, Stephen J. Maxson[1], and David P. Rave

SUMMARY OF FINDINGS

Improved knowledge of harvest parameters, demographic vital rates, influence of harvest regulations, and fall movements can contribute to the effectiveness of population management of large Canada geese (*Branta canadensis maxima*; hereafter, geese). Consequently, we conducted a band recovery study in Minnesota during 2002 to 2007. The objectives of this study were to (1) generate estimates of direct-recovery, recovery, harvest, and annual survival rates; (2) examine the influence of age class, time (year), and geographic strata on these parameters; (3) determine the relative influence of 10 harvest regulations on direct-recovery, recovery, and annual survival rates; (4) examine the structural relationship between each regulation and these parameters; and (5) characterize the spatiotemporal distribution of Minnesota-banded geese that were directly-recovered.

The harvest parameters and demographic vital rates of geese are influenced by life history characteristics (Miller et al. 2007), age class, harvest regulations, and geographic area (Sheaffer et al. 2005). Variation of some demographic and harvest parameters has been detected at multiple spatial scales (i.e., local, states, groups of states), but other geographic strata (e.g., hunting zones, broad-scale habitat classifications) may better explain such variation. Identification of the spatial scale that best explains this variation would improve our knowledge of the variables that influence goose populations.

We developed and evaluated a set of models in which harvest regulations, habitat, age class, and time were used to explain the variation of harvest parameters and annual survival rates. In subsets of models, we aggregated banding locations into 4 geographic strata: ecological province, hunting zone, a combination of ecological provinces and hunting zones (region), and statewide. Models in which banding locations are aggregated into these strata are consistent with hypotheses that spatial variation of parameters is attributable to habitat quality, harvest regulation package, both habitat quality and harvest regulation package, and no variation of these parameters at a spatial scale less than that of a state, respectively.

The recovery rates of geese varied by age, region, and time (year), and annual survival rates varied by age and region. The model-averaged region-specific annual survival rate estimates of adult (AHY) geese ranged from 0.580 (0.547–0.612) to 0.741 (0.650–0.815), and those of flightless juveniles (L) ranged from 0.769 (0.719–0.813) to 0.875 (0.806–0.922).

The direct-recovery rates of geese also varied by age, region, and time. Region- and time-specific estimates associated with the AHY cohort ranged from 0.050 (0.021–0.115) to 0.220 (0.153–0.306), and those of the L cohort ranged from 0.058 (0.024–0.132) to 0.238 (0.174–0.317).

We converted estimates of direct-recovery rates to harvest rate estimates using the 2003–2004 to 2005–2006 band reporting rate estimate of geese in the Upper Mississippi Flyway (Zimmerman et al. 2009*b*). Region- and time-specific harvest rate estimates of the AHY cohort ranged from 0.066 (0.009–0.123) to 0.292 (0.196–0.388), and estimates associated with the L cohort ranged from 0.077 (0.011–0.143) to 0.316 (0.216–0.415). Because some of our harvest rate estimates exceeded the corresponding mortality rate estimates, it appears that there may be bias of some parameter estimates. Potential sources of this bias are (1) a difference between the estimated band reporting rate of AHY Mississippi Flyway Giant Population (MFGP) of geese during 2003–2005 and that of some cohorts banded in Minnesota during 2002–2007, (2) an age-specific difference in the loss rate of leg bands, and (3) underestimation of the annual mortality rate of the L cohort (Heller 2010).

We examined the relative influence of 10 harvest regulations on direct-recovery, recovery, and annual survival rates, and examined the relationship between each regulation and

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[1] retired.

these parameters. The number of days that hunting is permitted during the early goose season is the harvest regulation that most influenced these parameters. However, the quadratic structure of this predictor variable in the best approximating models suggests complex responses of harvest parameters and annual survival rates to this regulation.

We examined the dates and locations of direct recoveries to make inferences about the fall movements and harvest distributions of geese. In all regions, >45% of direct recoveries occurred within the geographic area of capture. Further, 75% of all directly-recovered geese were harvested in-state. Of the Minnesota-banded geese directly-recovered during the early goose hunting season, >90% were harvested in-state. However, this proportion decreased and became more variable among regions during the regular and late goose-hunting seasons.

Our results can be used to further develop harvest management strategies for MFGP geese, which could be integrated with the harvest management of migratory geese (e.g., Eastern Prairie Population [EPP]).

INTRODUCTION

The effectiveness of population management of large Canada geese can be improved with reliable estimates of harvest parameters and demographic vital rates. Some of these parameters may be influenced by variables such as life history characteristics (Miller et al. 2007), time (Heller 2010), age class, harvest regulations, and geographic area (Sheaffer et al. 2005). Geographic variation of harvest parameters and annual survival rates has been documented at the spatial scale of local areas (Balkcom 2010), states (Sheaffer et al. 2005) and groups of states (Hestbeck 1994), but it is not known if other geographic strata (e.g., hunting zones, broad-scale habitat classifications) also explain variation of parameters of interest. Knowledge of the spatial scale at which variation of parameters is most influenced would allow inferences to be made regarding the causes of variation.

Similarly, the harvest management of geese could be improved with knowledge of the relative importance of each individual harvest regulation. Harvest regulations are thought to be important influences on the annual survival rate of geese (Hestbeck and Malecki 1989), but there has been relatively little examination of the influence of these management tools on harvest parameters and demographic vital rates (e.g., Rexstad 1992, Sheaffer et al. 2005). Further, the timing of the goose harvest could be better managed with knowledge of fall movement characteristics. Such information could be inferred from the spatiotemporal harvest distribution of leg-banded geese.

Consequently, the Minnesota Department of Natural Resources (MNDNR) conducted a 6-year summer goose banding program to improve current knowledge of the population biology, harvest effects, and spatiotemporal characteristics of geese. The objectives of this study were to (1) generate estimates of direct-recovery, recovery, harvest, and annual survival rates; (2) examine the influence of age class, time (year), and geographic strata on these parameters; (3) determine the relative influence of 10 harvest regulations on direct-recovery, recovery, and annual survival rates; (4) examine the structural relationship between each regulation and these parameters; and (5) characterize the spatiotemporal distribution of Minnesota-banded geese that were directly-recovered.

STUDY AREA

A statewide annual banding quota of 5,500 geese was established, with 500 birds marked per year in each of 11 Goose Management Blocks (GMB, Figure 1). These GMBs encompassed the entire state. Capture sites were selected non-randomly by field personnel. We generally captured geese at different sites during each year of the study and limited the number of geese banded per site to $\leq$ 100. We augmented the MNDNR data set with geese banded by the White Earth Tribal banding program.

METHODS

Most flightless geese were captured by driving birds into panel and drive traps during mid-June to mid-July 2002–2007 (Cooch 1953, Costanzo et al. 1995). A few geese were captured with rocket-nets during August 2002 (Dill and Thornsberry 1950). Field crews usually banded geese in flocks with flightless young to avoid capturing molt migrants, which may have temporarily immigrated to Minnesota (Zicus 1981, Fashingbauer 1993), but were not part of the population that was the focus of this investigation. The age of captured birds was classified using external feather characteristics (Hanson 1962) and relative body size. Sex was classified by cloacal examination (Dimmick and Pelton 1994). Birds were marked with standard size 8 U. S. Geological Survey (USGS) aluminum butt-end leg bands and released at their respective capture locations. The toll-free telephone number of the USGS Bird Banding Lab (BBL) was printed on the standard bands placed on geese. Re-encounters of banded geese could be reported via this telephone number.

The BBL provided reports that included date and location of recovery, and probable cause of death. Locations of banding and recovery were recorded in a latitude-longitude coordinate system.

Modeling and Estimation of Recovery and Annual Survival Rates

We used maximum likelihood methods to model the recovery (f) and annual survival rates (S) of geese (Brownie et al. 1985). Recovery rate is defined as the probability that a banded bird is shot or found dead during the hunting season, retrieved, and reported; and annual survival is defined as the probability that a banded bird alive at the time of banding in year *t* survives until the banding period in year *t* + 1 (Williams et al. 2001).

In this analysis, we only used data from (1) normal, wild large Canada geese of known age class and sex marked with standard bands on which a toll-free telephone number was printed, and (2) unsolicited bands from geese that were shot or found dead during the 2002–2003 to 2007–2008 hunting seasons. We stratified band and recovery data by age class (after hatch-year [AHY] or local [L]), sex (male or female), time (years of capture and recovery), and geographic location of capture (8 levels: Northwest Zone–Aspen Parkland, West Zone–Prairie, West-central Zone–Prairie, Rest-of-State Zone–Prairie, Rest-of-State Zone–Forest, Rest-of-State Zone–Transition, Metro Zone–Transition, Southeast Zone–Transition; Figure 2). These geographic locations are a combination of hunting zones and ecological provinces, which we refer to as regions.

Initially, we developed a global model, $S_{\text{age * time * region}}$ $f_{\text{age * time * region}}$, in which S and f varied by age, time, and region of capture in interactive relationships. We further developed and examined support for 55 reduced models, which represent different hypotheses regarding the sources of variation of S and f. More specifically, S and f in the reduced models were parameterized to vary by age, time, and geographic stratification of capture locations in different combinations and structural relationships. Sex was not included as a predictor in the global or most reduced models, because previous studies did not detect sex-specific differences in the annual survival rate estimates of Canada geese (e.g., Rexstad 1992, Eichholz and Sedinger 2007), and our band and recovery data for some sex cohorts were sparse.

To make inferences about causes of geographic variation of S and f , we developed and evaluated support for a subset of models in which banding locations were aggregated into 4 strata: (1) ecological provinces (4 levels: Prairie Parkland [Prairie], Eastern Broadleaf Forest [Transition], Tallgrass Aspen Parkland [Aspen Parkland], Laurentian Mixed Forest [Forest]; Hanson and Hargrave 1996), (2) hunting zones (6 levels: Metro, Northwest, Southeast, West, West-central, Rest-of-State), (3) the 8 regions comprised of both ecological provinces and hunting zones, and (4) statewide. Corresponding hypotheses for spatial variation in these parameters are geographical differences in: (1) habitat quality, (2) harvest regulation package, and (3) an interaction between these variables. Stratum (4) supports the hypothesis that S and f do not vary at a spatial scale less than that of a state. Our approach requires the assumptions that demographic vital rates are positively associated with habitat quality (sensu Van Horne 1983) and negatively associated with harvest parameters.

The smallest strata we examined (regions) ranged in area from 284,677 to 9,483,281 ha. All regions were greater in spatial extent than second-order habitat selection (i.e., home range), but less than that of first-order habitat selection (i.e., geographical range, Johnson 1980). We refer to the geographic strata examined as habitat at a broad spatial scale. Thus, results would suggest that habitat quality at a broad spatial scale is an important influence on parameters of interest if there is substantial support for models in which capture locations are stratified by ecological provinces.

We conducted a second analysis to examine the influence of individual harvest regulations on recovery and annual survival rates. We developed a series of 30 models in which age class is a predictor and 10 regulatory tools (Julian dates of the first days of the early and regular goose-hunting seasons; daily bag limits during the early, regular, and late seasons; number of hunting days permitted during the early, regular, and late seasons; number of non-hunting days between the early and regular seasons; total number of days that goose hunting was permitted [Appendix 1]) associated with each hunting zone–year combination are treated as covariates (see Rexstad 1992). Although we examined the geographic variation of S and f at multiple spatial scales in the first analysis, we modeled the relationship between regulations and these parameters associated with hunting zones in the second analysis, because this is the spatial scale at which the regulatory treatment was applied to the goose population. We included age class as a predictor in these models, because this variable likely influences the recovery and annual survival rates of MFGP geese (e.g., Sheaffer et al. 2005), and it is possible that each age class is affected differently by harvest regulations.

For each regulation, we developed 3 models that were structured as follows: (1) additive relationship between age class and the regulation of interest, (2) interactive relationship between age class and a regulation, and (3) additive relationship between age class and a regulation, with each regulation structured in a quadratic manner. More specifically, the equations used to explain the relationship between individual regulations and S and f in (3) are:

$$\text{logit}\,(S) = \beta o + \beta_1(x_1) + \beta_2(x_2) + \beta_3(x_3^2) + \varepsilon \qquad (1)$$

$$\text{logit}\,(f) = \beta o + \beta_1(x_1) + \beta_2(x_2) + \beta_3(x_3^2) + \varepsilon \qquad (2)$$

in which βo is the vector of the intercept parameter, β_1 is the vector of the slope parameter for age class (L = 0, AHY = 1), β_2 is the vector of slope parameter for the regulation of interest, β_3 is the vector of slope parameter for this regulation^2; x_1, x_2, and x_3 are the vectors of age class, the regulation of interest, and this regulation^2, respectively; and ε is the random error term.

We used the Brownie et al. Recoveries option of Program MARK (Brownie et al. 1985, White and Burnham 1999) to fit the candidate models to the data, generate model-averaged estimates and associated 95% confidence intervals (CI) of S and f, calculate the Akaike Information Criterion (AICc) values adjusted for small sample size of each model, and perform a bootstrap simulation to estimate a median $\hat{C}$ - value associated with the global model (Cox and Snell 1989, Burnham and Anderson 2002). Because data are considered overdispersed if the median $\hat{C}$ - value associated with the global model is >1.0 (i.e., sampling variance that is greater than the theoretical model-based variance; Burnham and Anderson 2002), it is necessary to adjust the AICc-values of such models for overdispersion. Thus, we evaluated the relative support of each model based on AICc-values adjusted for overdispersion (i.e., QAICc values). We generated model-averaged parameter estimates of S and f using general models with weights of positive value. However, we did not generate model-averaged estimates of these parameters using harvest regulation models, because the objectives of this modeling effort were to determine the relative influence of individual regulations on parameters of interest and examine the relationship between each regulation and these parameters.

Modeling and Estimation of Direct-Recovery Rates

Direct-recovery rates ($f*$) can be modeled and estimated within the Brownie et al. (1985) framework, but we chose to generate estimates of this parameter using logistic regression (Otis and White 2002). Using the latter method permitted us to avoid the influence of nuisance parameters (i.e.; indirect recovery rates, S) on the associated area of maximum likelihood and ultimately on estimates of $f*$. This parameter is defined as the proportion of birds marked in year *t*, shot, and retrieved during the first hunting season after banding, and reported to the BBL (Williams et al. 2001). In this analysis, we used data from only (1) normal, wild large Canada geese of known age class and sex that were marked with standard bands on which a toll-free telephone number was printed, and (2) unsolicited, directly-recovered bands from these marked birds. We stratified band and recovery data from 2002–2007 by age, time (year of capture), and region of capture.

We developed a series of logistic-regression models in which the structures of $f*$ are similar to those of f in most of our Brownie et al. (1985) models. The parameter $f*$ varied by age, time (year), and region of capture in the global model, $f*_{\text{age * time * region}}$. The 27 reduced models used different combinations of age, time, and location of capture as predictor variables in the initial analyses. Sex was not used as a predictor in any models.

We developed a second series of 30 models to examine the influence of 10 harvest regulations (Appendix 1) on direct-recovery rates. In each model, age class is a predictor variable, and the individual regulation associated with each hunting zone–year combination is treated as a covariate (sensu Rexstad 1992). We developed 3 models for each regulation using the structures described in the *Modeling and Estimation of Recovery and Annual Survival Rates* subsection.

We used the Known Fate option of Program MARK (White and Burnham 1999) to fit models to the data, generate model-averaged direct-recovery rate estimates, and calculate information-theoretic model-selection statistics (Burnham and Anderson 2002). We used the AICc-values to rank the relative support for each model (Akaike 1973, Burnham and Anderson 2002). We generated model-averaged parameter estimates of $f*$ using general models with weights of positive value. We did not generate model-averaged parameter estimates of this parameter using the harvest regulation models, because the objectives of this portion of the analyses were to ascertain the relative importance of individual regulations and examine the relationship between each regulatory tool and $f*$.

Conversion of Direct-Recovery Rate Estimates to Harvest Rate Estimates

To convert $\hat{f}*$ to harvest rate estimates ($\hat{H}$), we used the equation

$$\hat{f}* = \hat{H}\hat{\lambda}, \qquad (3)$$

in which $\hat{\lambda}$ is the band reporting rate estimate (Henny and Burnham 1976). We used the values of our model-averaged $\hat{f}*$ and the 2003–04 to 2005–06 $\hat{\lambda}$ associated with the Upper Mississippi

Flyway (0.754, 1 $\hat{SE}$ = 0.040; Zimmerman et al. 2009*b*) to estimate $\hat{H}$. We used the delta method (Seber 1982) to estimate the variance ($\hat{\text{var}}$) of $\hat{H}$ as:

$$\hat{\text{var}}\,(\hat{H}) = \left(\frac{1}{\hat{\lambda}}\right)^2 \hat{\text{var}}\left(\hat{f}\,*\right) + \left(\frac{\hat{f}\,*}{(\hat{\lambda})^2}\right)^2 \hat{\text{var}}\left(\hat{\lambda}\right), \tag{4}$$

with $\hat{SE}\,(\hat{H}) = \sqrt{\hat{\text{var}}\,(\hat{H})}$ and the bounds of the 95% CIs associated with this parameter estimated as $\hat{H}$ ± 1.96 * $\hat{SE}$ ($\hat{H}$). Our application of this method requires the assumption of spatiotemporally constant band reporting rates in Minnesota during the study period. Further, values of $\hat{\text{var}}$ ($\hat{H}$) should be viewed with caution, because the delta method (Seber 1982) "may not approximate variance well" if the values of transformed variables are very nonlinear (Powell 2008).

Spatiotemporal Characteristics of the Harvest Distribution

We used the dates and locations of direct recoveries of banded geese from each region to make inferences about fall movements and harvest distributions. We used only the direct-recovery data for which reliable information about the date and location of re-encounters were available.

The BBL recorded most locations of bandings and recoveries as the southeastern corner of the 10' latitude-longitude blocks in which geese were encountered, and classified the state or province in which recoveries occurred. We assigned the locations of each banding and in-state direct recovery to the appropriate region; i.e., that which contained the majority of the 10' block in which each encounter occurred.

We stratified data by the region of capture; the region and state or province of recovery; and the time of recovery (3 periods: 1–22 September, 23 September–30 November, and 1 December–21 February). These periods approximately correspond with the September goose-hunting season of Minnesota, the regular season, and the December goose-hunting season plus late seasons in other states, respectively. We then calculated the proportion of recovered geese that were re-encountered (1) in each region and state or province, and (2) in-state during the 3 periods. These proportions were not adjusted by spatiotemporal differences in band reporting rates (see Otis 2004), because there is little evidence of such differences in the band reporting rates of geese captured in the U. S. during the contemporary period (Zimmerman et al. 2009*b*).

We created figures of recovery locations of geese banded in each region based on further stratification of recovery dates (2 periods: 1–22 September and 23 September–21 February). We stratified the hunting season into the early hunting season and the remainder of the hunting season to display the temporal differences between the spatial movement patterns of geese during these periods. It is less likely that fall migration occurred during the earlier period than the latter.

RESULTS

Modeling and Estimation of Recovery and Annual Survival Rates

We analyzed band and recovery data from 32,912 normal, wild geese of known age and sex, of which 5,191 were shot or found dead during the hunting season after banding, retrieved, and reported, and 5,080 were shot or found dead during a hunting season >1 year after banding. These data were slightly overdispersed, as indicated by a median $\hat{C}$ -value of 1.043.

Three approximating models had weights with values >0, and all indicated that recovery rates and annual survival rates were influenced by age class and geographic strata (Table 1). Time (year) was a predictor of recovery, but not annual survival in these 3 models. Although there were 2 geographic strata of capture locations among the 3 models, region (i.e., combined hunting zones and ecological provinces) was the stratum with the greatest support (97% of total model weight). Consequently, we chose to generate model-averaged parameter estimates using the 2 models with positive weight in which recovery and annual survival varied by age and region, and recovery varied by time.

The model-averaged age- and region-specific estimates of the annual survival rate ranged from 0.580 (0.547–0.612) to 0.741 (0.650–0.815) for the AHY cohort and from 0.769 (0.719–0.813) to 0.875 (0.806–0.922) for the L cohort (Table 2, Figure 3). The model-averaged region- and time-specific estimates of recovery rates for AHY birds ranged from 0.049 (0.020–0.115) to 0.177 (0.134–0.230), and those of the L cohort ranged from 0.058 (0.024–0.134) to 0.205 (0.156–0.264, Table 2).

In the examination of the influence of harvest regulations on recovery and annual survival rates, only the model $S_{\text{age + \# Days1 + \# Days1}^2}$ $f_{\text{age + \# Days1 + \# Days1}^2}$ had any support (Table 3). The relationships between the predictor number of days that goose hunting was permitted during the early goose season and the response variables S and f are described by the equations:

$$\text{logit}\,(\hat{S}) = 7.778 - 0.959\,(\text{age class}) - 0.771\,(\#\,\text{Days1}) + 0.023\,(\#\,\text{Days1}^2) \qquad (5)$$

$$\text{logit}\,(\hat{f}) = -5.876 - 0.196\,(\text{age class}) + 0.441\,(\#\,\text{Days1}) - 0.011\,(\#\,\text{Days1}^2). \qquad (6)$$

The vectors of the intercept and slope parameter estimates were significant, as indicated by 95% CIs that do not encompass 0 (estimated annual survival rate: $\hat{\beta}_0$ = 7.778 [4.199–11.358], $\hat{\beta}_1$ = -0.959 [-1.213– -0.706], $\hat{\beta}_2$ = -0.771 [-1.189– -0.353], and $\hat{\beta}_3$ = 0.023 [0.011–0.035]; estimated recovery rate: $\hat{\beta}_0$ = -5.876 [-7.106– -4.645], $\hat{\beta}_1$ = -0.196 [-0.259– -0.134], $\hat{\beta}_2$ = 0.441 [0.308–0.574], and $\hat{\beta}_3$ = -0.011 [-0.015– -0.008]). There were some significant differences among the estimates of S and f that were associated with different numbers of hunting days permitted during the early goose season (Table 4, Figure 4).

Modeling and Estimation of Direct-Recovery Rates

We analyzed the band and recovery data from 32,912 normal, wild geese of known age class and sex, of which 5,142 were directly recovered. The $\hat{C}$ - value associated with the global model was 0.844.

The 2 top-ranked logistic regression models (100% of model weight) indicated that direct-recovery rates were influenced by a complex relationship among age class, region, and time (Table 5). Specifically, there was complete support for models in which there were additive relationships between age and time, and an interactive relationship between time and region. The top-ranked model (74% of total weight) indicated an additive relationship between age and region, but there was an interactive relationship between these predictors in the second-ranked model.

We generated model-averaged estimates of direct-recovery rates from the models with 100% of total model weight ($f^*_{\text{age + time, time * region, age + region}}$ and $f^*_{\text{age + time, time * region, age * region}}$). These estimates ranged from 0.050 (0.021– 0.115) to 0.220 (0.153–0.306) for AHY birds, and from 0.058 (0.024–0.132) to 0.238 (0.174–0.317) for L birds (Table 6).

In the examination of the influence of harvest regulations on direct-recovery rates, only the model $f^*_{\text{age + \# Days1 + \# Days1}^2}$ had substantial support (Table 7). The relationships between the predictor number of days that goose hunting was permitted during the early goose season and the response variable f ***** is: described by the equation:

$$\text{logit}(\hat{f}^*) = -6.270 - 0.183\,(\text{age class}) + 0.480\,(\#\ \text{Days1}) - 0.012\,(\#\ \text{Days1}^2). \qquad (7)$$

The vectors of intercept and slope estimates of parameters in equation 7 were significant, because associated 95% CIs do not encompass 0: (i.e.; $\hat{\beta}_0$ **=** -6.270 [-7.797– -4.744], $\hat{\beta}_1$ = -0.183 [-0.258– -0.108], $\hat{\beta}_2$ = 0.480 [0.316–0.645], and $\hat{\beta}_3$ = -0.012 [-0.017– -0.008]). Similarly, there were significant differences among some estimates of f ***** that were associated with different numbers of hunting days permitted during the early goose season (Table 4, Figure 5).

Conversion of Direct-Recovery Rate Estimates to Harvest Rate Estimates

The time- and region-specific harvest rate estimates for the AHY cohort ranged from 0.066 (0.009–0.123) to 0.292 (0.196–0.388; Table 8). The time- and region-specific estimates for L birds ranged from 0.077 (0.011–0.143) to 0.316 (0.216–0.415; Table 8).

Spatiotemporal Characteristics of the Harvest Distribution

In all regions, >45% of the direct recoveries of banded geese occurred in the region of capture (Table 9). Overall, 75% of direct recoveries were harvested in-state, and minor proportions recovered in Missouri, Illinois, Iowa, North Dakota, and South Dakota (Table 10). Trace proportions were harvested in several other states and provinces (Table 10), but there were exceptions to these general findings. Specifically, >50% of the directly-recovered geese captured in the Southeast Zone – Transition region during 2007 and West-central Zone – Prairie region during 2005 and 2007 were harvested out-of-state (Table 10).

The proportion of geese both captured and directly-recovered in Minnesota varied temporally within the hunting season. During the September portion of the 2002–2003 to 2007–2008 hunting seasons, >90% of the direct recoveries of Minnesota-banded geese were harvested in-state. However, this proportion decreased and became more variable among regions during the latter portion of the hunting season (Table 11). For example, during 1 December–21 February, 7–40% of directly-recovered geese were harvested in-state (Table 11).

We present the period-specific (1–22 September and 23 September–21 February) spatial distribution of directly-recovered geese banded in each region in Figures 6–13.

DISCUSSION

Modeling and Estimation of Recovery and Annual Survival Rates

There was substantial support for models in which the annual survival rate of geese in Minnesota varied by age class and region during 2002–2007. The additive relationship between these predictors in the best approximating model suggests that the age-specific difference in annual survival rates did not vary among regions. The annual survival rate estimates of L birds were significantly greater than those of AHY birds in every region except the Northwest Zone–Aspen Parkland, based on 95% CIs (Table 2).

Although there was substantial support for geographic variation in annual survival, significant differences were detected only among the AHY birds from some regions (Table 2). Specifically, the estimate of AHY birds captured in the Northwest Zone–Aspen Parkland region was significantly greater than those of that age cohort captured in the Rest-of-State Zone–

Transition and West Zone–Prairie regions, based on 95% CIs (Table 2). We are not sure why significant region-specific differences were not detected in the L cohort, but it may be that (1) a greater sample size of the L cohort must be banded to detect geographic differences in annual survival rates, or (2) geographic differences in the effects of sources of mortality most influenced the AHY cohort.

The annual survival rate estimates of goose populations nesting in the temperate zone of the Mississippi Flyway generally varied by age class and geographic area, and the estimates of the L cohorts generally were greater than those of corresponding AHY cohorts (Sheaffer et al. 2005). However, the opposite was true for 3 Mississippi Flyway populations of geese nesting in the Subarctic (see Sheaffer et al. 2005). Although this relationship between age class and geographic area appears to be interactive at the spatial scale of this flyway (Sheaffer et al. 2005), our results suggest that an additive relationship between age classes occurred at the scale of regions within Minnesota. This dissimilarity in the structural relationship between age class and geographic area may be attributed to the difference in stress associated with the migration distances of subarctic- and temperate-nesting populations. The L cohort in the temperate zone may have had a greater annual survival rate than those of subarctic populations, because the latter migrated further to their wintering grounds than did MFGP geese (Sheaffer et al. 2004), and this age cohort may have relatively low survival during long migrations (Eichholz and Sedinger 2007). Alternatively, there may have been a slight negative bias of the annual survival rate estimates of AHY cohorts and a substantial positive bias of the estimates of L cohorts of temperate-nesting geese (Heller 2010), because of the molt migration of some banded MFGP geese away from the capture site and consequent violation of the assumption of equal probability of recovery and survival of all individuals of an identifiable cohort (Pollock and Raveling 1982).

The geographic variation of the annual survival rates of geese has been detected at several spatial scales. For example, significant differences in annual survival rates were detected among wintering areas composed of multiple states in the Atlantic Flyway (Hestbeck 1994). Within age classes and time periods, there were significant differences among the annual survival rate estimates of MFGP geese captured in some states (Sheaffer et al. 2005). A significant difference was detected between the annual survival rates of AHY geese captured at a 160-ha urban site and 2,428-ha rural site in Georgia (Balkcom 2010).

These investigations provided important information about the spatial variation of annual survival rates, but did not evaluate which of multiple geographic strata best explained variation of this parameter. By ascertaining which of 4 geographic stratifications had the greatest support, we were able to infer that the variation of annual survival rates occurred at the smallest spatial scale examined; i.e., region. This result suggests that annual survival rates of geese were influenced by both the harvest regulation package and broad-scale habitat classification, but there may be alternative explanations (e.g., spatial differences in hunter effort). Significant relationships between annual survival rates and harvest regulations have been detected in other studies (e.g., Sheaffer et al. 2005), but we are not aware of any documentation of the association between broad-scale habitat and annual survival rates of geese.

The predictors of recovery rates in the 2 Brownie et al. (1985) models used to generate model-averaged parameter estimates are similar to those in the 2 top-ranked logistic regression models used to estimate direct-recovery rates. Therefore, we will discuss only direct-recovery rates in the *Modeling and Estimation of Direct-Recovery Rates* subsection.

Modeling and Estimation of Direct-Recovery Rates

Our results suggest that the age-specific difference in direct-recovery rates was temporally and geographically constant, but that the difference among regions varied temporally. Further, the order of regions in terms of direct-recovery rate point-estimate values changed annually, and the temporal differences associated with 5 of 8 regions were significant (Table 6). Although we are not sure of the cause(s) of this spatiotemporal variation of direct-recovery rates, the substantial support for models that include region as a predictor suggests that both habitat at a broad spatial scale and harvest regulation package were important influences on this parameter.

The model-averaged direct-recovery rate point estimates of the L cohorts captured in all regions and years were greater than the corresponding estimates of AHY birds, but the associated 95% CIs indicate that age-specific differences were not significant. This finding generally is consistent with the results of Sheaffer et al. (2005); i.e., most direct-recovery rate estimates of the L cohorts of MFGP geese were slightly greater than those of the AHY cohorts within the same time periods and states (Sheaffer et al. 2005). The slightly greater direct-recovery rate estimates of the L cohorts observed in these 2 studies suggest that this age cohort is more vulnerable to hunting mortality than is the AHY cohort (see Samuel et al. 1990).

Harvest Rate Estimates

Because model-averaged direct-recovery rate estimates were converted to harvest rate estimates using a single value of a band reporting rate estimate (Henny and Burnham 1976), the relationships among harvest rate estimates of age class, region, and year cohorts are essentially the same as those observed among direct-recovery rate estimates. Specifically, the harvest rate estimates of the L cohort consistently were greater than those of the AHY cohort of corresponding years and regions, but these differences were not significant, as indicated by 95% CIs (Table 8). Within each age class, we detected some significant region-specific differences among harvest rate estimates during the same year. There also were some significant temporal differences in harvest rate estimates among geese of the same age class and region.

We are unaware of other studies in which age-specific differences in the harvest rates of MFGP geese were examined. However, the significant region- and time-specific variation of harvest rate estimates of Minnesota geese is consistent with the observation of spatiotemporal variation among multistate wintering areas (Hestbeck 1994), and similar to studies that detected spatial variation of this parameter at multiple spatial scales; i.e., local (Balkcom 2010) and major breeding areas (Zimmerman et al. 2009*b*).

The spatial variation of harvest rates of AHY geese in North America is best explained by major breeding areas (temperate, mid-latitude, and high-latitude; Zimmerman et al. 2009*b*), but it was most appropriate to compare our estimates with that of MFGP geese (0.167 [0.152–0.183]), because birds captured in Minnesota are part of this population. The harvest rate estimate of AHY MFGP geese is significantly greater than 5 of 47 of our estimates (Northwest Zone–Aspen Parkland [2002–2003, 2003–2004, 2004–2005, 2007–2008], West-central Zone–Prairie [2006–2007]), and significantly less than 6 estimates (Rest-of-State Zone–Transition [2003–2004, 2005–2006, 2006–2007], Southeast Zone–Transition [2007–2008], and West Zone–Prairie [2003–2004, 2007–2008]; Table 8). It is plausible that these differences between some of our estimates and those of MFGP geese during an overlapping time period (Zimmerman et al. 2009*b*) occurred, because of the (1) restrictive harvest regulations in effect in the Northwest Zone–Aspen Parkland and West-central Zone–Prairie regions, and (2) relatively liberal regulations associated with other regions (Appendix 1).

Influence of Individual Harvest Regulations

Although harvest regulations are used to manage some wildlife populations (Caughley 1985), there are few published investigations in which the effects of individual regulations on direct-recovery, recovery, and annual survival rates of temperate-nesting large Canada geese have been examined (see Rexstad 1992, Sheaffer et al. 2005). Our rankings of the influence of 10 regulations (Tables 3 and 7) on harvest parameters and annual survival rates provide information about the effectiveness of these management tools.

The number of hunting days permitted during the early goose-hunting season was the harvest regulation that most influenced each parameter of interest. The quadratic structure of this predictor (equations 5–7) in the best approximating models suggests that such relationships may be complex. It may be that after being hunted for some period during September, geese became more wary and altered their behavior in a manner that decreased direct-recovery and recovery rates and increased annual survival rates. Alternatively, the importance of the number of hunting days permitted during the early goose-hunting season as a predictor of direct recovery, recovery, and annual survival rates may have been a statistical artifact; i.e., this predictor may have been confounded with an unexamined variable that was an influential source of variation of harvest parameters and annual survival rates.

Intuitively, harvest regulations in effect during the early goose-hunting season should have been relatively important, because (1) few leg-banded geese appeared to have migrated away from the region of capture during this portion of the hunting season (Figures 6–13), (2) most directly-recovered geese were harvested in the region of capture (Table 9), and (3) a disproportionately great percentage of the total goose harvest occurred during the early goose-hunting season. Specifically, 38% (range: 29–47%) of the total 2002–2003 to 2007–2008 goose harvest occurred during the early goose-hunting season (Fronczak 2009, Rave et al. 2009), which was about 20% of the total number of days that goose hunting was permitted during the entire hunting season (Appendix 1).

Our finding of the importance of the early goose-hunting season is consistent with Sheaffer et al. (2005); i.e., generally, estimates of annual survival rates decreased and direct-recovery rates increased after the establishment of an early season in the Mississippi Flyway. However, the direction and significance of these changes were not consistent among some cohorts during that study. For example, after the establishment of an early hunting season in Minnesota, the annual survival rate estimate of the AHY cohort captured in this state decreased significantly and that of the L cohort decreased only slightly (Sheaffer et al. 2005). Further, the direct-recovery rate estimate of the AHY cohort increased slightly, but that of the L cohort decreased slightly after this regulatory change occurred (Sheaffer et al. 2005). Such counterintuitive results may be attributable to a counteracting effect between the relatively liberal regulations in effect during the early season and more restrictive regulations enacted during the regular hunting seasons (Sheaffer et al. 2005).

The relative importance of some individual harvest regulations we observed are somewhat similar to those detected in a study of temperate-nesting geese in Utah (Rexstad 1992). Specifically, opening date and daily bag limit did not significantly influence the annual survival rate of geese in Utah (Rexstad 1992), and there was no support in our analyses for models in which these 2 regulations were predictors of this demographic vital rate (Table 3). There was a weak negative relationship ($P = 0.04$) between the total number of days that goose hunting was permitted and annual survival rates of geese in Utah, but Rexstad (1992) concluded that the overall regression model did not adequately explain this relationship. Similarly, there was no support for our third-ranked model in which total number of days that goose hunting was permitted was a predictor of recovery and annual survival rates (Table 3).

The AIC values associated with our best approximating harvest regulation models (Tables 3 and 7) are substantially greater than those of the more general models (Tables 1 and 5). Thus, individual regulations appear not to be the most important influences on the parameters of interest. We attribute this limited influence of individual regulations to (1) the confounding effect of some geese using multiple hunting zones that each have different harvest regulations (Sheaffer et al. 2005), (2) variables not included in our analyses that have a greater effect on the survival of some age cohorts than do harvest regulations (e.g., density-dependence; Hill et al. 2003) or have an interactive effect with regulations, (3) the likelihood that the entire harvest regulation package has a greater effect on the parameters of interest than does any individual regulation, and (4) the relative lack of support for the predictors in our regulatory effects models. Specifically, all parameters of interest varied among regions (i.e., a combination of hunting zones and ecological provinces) and survival was temporally constant among years in the best approximating general models, but these parameters varied among hunting zones and years in all regulatory models.

Spatiotemporal Characteristics of the Harvest Distribution

Leg-banded geese that were captured in Minnesota and directly-recovered during the 2002–2003 to 2007–2008 hunting seasons were most likely to be harvested both in-state and in the region of capture (Tables 10 and 11). This phenomenon was most evident during 1–23 September, but continued until 30 November (Table 11).

It was not uncommon for geese banded in the prairie regions (i.e., Westcentral–Prairie, Rest-of-State–Prairie, West–Prairie) to have been directly recovered in the Dakotas (Table 10; Figures 8, 11, 12). This phenomenon was not observed in geese banded in other regions, and was more pronounced during the 2002–2003 to 2007–2008 hunting seasons than during 1976–1996 (see Lawrence 1997). This apparent change in harvest distribution may have occurred because of (1) temporal changes in fall movements of geese in western Minnesota, (2) temporal changes in harvest regulations of the Dakotas, or (3) other unexamined variables.

Study Limitations and Potential Biases

Although interesting, our results should be interpreted with caution for several reasons. First, the limited spatiotemporal scale (1 state, 6 years) of this study increased the likelihood that spurious results could have been generated. This situation can be addressed by analyzing goose band recovery data collected during a greater time period and at spatial scales based on sound ecological theory. Such analyses should examine variation of phenomena of interest at spatial scales that range from local to the geographic range of this species, including the geographic strata at which geese are managed (i.e., flyways, states and provinces, hunting zones within states).

Second, confounding effects often associated with observational studies (Shaffer and Johnson 2008) may have precluded a determination of whether harvest parameters and annual survival rates were attributable to (1) an interaction between habitat at a broad spatial scale and harvest regulation packages, or (2) unexamined variables (e.g., hunter effort, availability and effectiveness of waterfowl hunting refuges) that were spatially distributed in a manner similar to that of regions. Similarly, our non-experimental approach did not permit us to ascertain whether the relative ranking of each harvest regulation model was caused by changes in harvest regulations, or the results were a statistical artifact. For example, we could not determine whether the relatively high estimates of annual survival and low recovery rates associated with the only hunting zone (Northwest) with < 17 days of hunting permitted during the early goose season was attributable to conservative harvest regulations, other unexamined variables or characteristics of the data (i.e., differences in magnitude of variation among regulations [Appendix 1]).

These problems can be avoided by conducting future investigations in an experimental framework (Anderson et al. 1987, Shaffer and Johnson 2008) with controls (i.e., no-hunting zones), treatments (e.g., harvest regulation packages randomly assigned to ecological provinces), and replicates. Despite the advantages of such a design, it is unlikely that this experiment could be performed on temperate-nesting goose populations, because (1) the establishment of non-hunted areas would be unfeasible, because of anticipated increases in the number of nuisance goose problems, and (2) concern about the overharvest of EPP geese may preclude the assignment of a liberal harvest regulation treatment to geographic areas in which this population stages or winters.

Third, the violation of certain assumptions of banding studies may have contributed to biased estimates of some parameters. For example, the assumption of independent fates of marked individuals in the study population (Brownie et al. 1985) may have been violated, because of (1) behaviors such as flocking, the propensity to pair for life, and the relatively long-term association of young with parents (Burnham and Anderson 2002), and (2) the nonrandom selection (Hoeting 2009) of some capture sites in close spatial proximity (Christman 2008). Non-independence in count data contributes to overdispersion (Burnham and Anderson 2002).

However, the $\hat{C}$-value of slightly >1 in our global Brownie et al. (1985) model suggests that our data were not substantially overdispersed (Burnham and Anderson 2002) and that the non-independence of fates of marked individual geese was not a substantial problem in this data set.

Of greater concern are potential violations of the assumptions (1) no loss of leg bands, (2) appropriate structure of models used to estimate direct-recovery and band reporting rates, (3) the estimate of band reporting rate is applicable to all cohorts for which harvest rates are estimated, and (4) all leg-banded geese of an identifiable cohort had the same probability of recovery and annual survival (Pollock and Raveling 1982). Because 31 of 47 of the harvest rate point estimates associated with the L cohort are greater than the corresponding annual mortality rate point estimates ($\hat{M}$ = 1-$\hat{S}$), we suspect that $\geq$1 of these 4 assumptions were violated. Consequently, it is reasonable to question which parameters were biased, the direction of bias, and whether bias was age-specific.

A difference between the estimated band reporting rate of AHY MFGP geese during the 2003–04 to 2005–06 hunting seasons (Zimmerman et al. 2009*b*) and that of some of the L cohorts banded in Minnesota during 2002–2007 could have contributed to biased harvest rate estimates of the latter age cohort. Alternatively, the loss of both standard (Coluccy et al. 2002) and reward leg bands (Zimmerman et al. 2009*a*) could have contributed to biases of band reporting and harvest rate estimates, but determining the direction of such biases is confounded by the different loss rates of each type of band and the appropriateness of models used to estimate these parameters (see Zimmerman et al. 2009*a*). Estimates of band reporting and band loss rates of L geese are not available, because reward bands were not attached to this age cohort during a recent North American study (Zimmerman 2009*a*,*b*).

The observed rate of leg-band loss should not have greatly influenced estimates of annual survival rates generated in band recovery models (Coluccy et al. 2002), and estimates of annual survival and mortality rates generated from the analytical method we used were thought to be essentially unbiased (Brownie et al. 1985). However, a violation of the assumption that all leg-banded geese of an identifiable cohort had the same probability of recovery and annual survival (Pollock and Raveling 1982) may have contributed to a slight positive bias in our annual mortality rate estimates of the AHY cohorts and a substantial negative bias of the estimates associated with the L cohorts (Heller 2010).

The source of this violation may have been the inclusion of both molt migrants and non-molt migrants in the same cohort during analysis (Heller 2010). Molt migrants generally are failed nesters or nonbreeders in their second and third year that temporarily emigrate away from the breeding grounds to molt (Zicus 1981, Abraham et al. 1999), and thus may not have the same probabilities of recovery and annual survival as geese that do not emigrate (Heller 2010). Unfortunately, it is not possible to (1) ascertain at the time of capture which banded individuals will perform molt migration in future years, and (2) assign molt migrants and non-molt migrants to different cohorts in the Brownie et al. (1985) option of Program MARK (White and Burnham 1999). To address this problem, Heller (2010) developed an analytical method in which the age classes most commonly associated with molt migration (i.e., second and third year geese) can be assigned to an age cohort different than that of AHY geese captured on the breeding grounds.

Fourth, the locations of direct recoveries were not adjusted for spatiotemporal differences in band reporting rates (Nichols et al. 1995, Otis 2004, Zimmerman et al. 2009*b*) and thus should be interpreted with some caution. However, any adjustment of raw data likely would be minimal, because there is little evidence of spatial differences in the band reporting rates of geese captured in the U. S. during the contemporary period (Zimmerman et al. 2009*b*).

Management Implications

Our results contribute to the knowledge of the sources of variation of direct-recovery, recovery, and annual survival rates; contemporary estimates of these parameters and harvest rates; potential sources of bias in parameter estimates; relative importance of individual harvest regulations; the characteristics of fall movements; and the spatial scales at which several processes occur in Minnesota. Specifically, direct-recovery, recovery, and annual survival rates vary at the scale of regions (i.e., combined ecological provinces and hunting zones), but fall movements appear to vary at a greater scale.

Improved knowledge of the spatial scales at which these processes occur and the relative influence of harvest regulations on direct-recovery, recovery, and annual survival rates can be used to further develop harvest management strategies for MFGP geese. For example, the identification of region as the spatial scale that best explains variation of harvest parameters and annual survival rates suggests that these parameters are partially influenced by large-scale habitat quality and thus may be changed by habitat management. However, effective habitat management likely would have to be implemented at a large spatial scale, given the spatial extent of regions in Minnesota (>280,000 ha).

Our rankings of the relative importance of individual harvest regulations and the quantitative relationships between the most influential regulation and parameters of interest may be useful in the development of harvest management strategies. However, more work is needed to ascertain how individual regulations interact with each other.

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LITERATURE CITED

Abraham, K. F., J. O. Leafloor, and D. H. Rusch. 1999. Molt migrant Canada geese in northern Ontario and western James Bay. Journal of Wildlife Management 63:649–655.

Akaike, H. 1973. Information theory as an extension of the maximum likelihood principle. Proceedings of the International Symposium on Information Theory 2:267–281.

Anderson, D. R., K. P. Burnham, J. D. Nichols, and M. J. Conroy. 1987. The need for experiments to understand population dynamics of American black ducks. Wildlife Society Bulletin 15:282–284.

Balkcom, G. D. 2010. Demographic parameters of rural and urban adult resident Canada geese in Georgia. Journal of Wildlife Management 74:120–123.

Brownie, C., D. R. Anderson, K. P. Burnham, and D. S. Robson. 1985. Statistical inference from band recovery data–a handbook. Second edition. U. S. Fish and Wildlife Service Resource Publication 156, Washington, D. C.

Burnham, K. P., and D. R. Anderson. 2002. Model selection and multimodel inference. Second edition. Springer, New York, New York.

Caughley, G. 1985. Harvesting of wildlife: past, present, and future. Pages 3–14 *in* S. L. Beasom and S. F. Roberson, eds. Game Harvest Management. Caesar Kleberg Wildlife Research Institute, Kingsville, Texas.

Christman, M. C. 2008. Statistical modeling of observational data with spatial dependencies. Journal of Wildlife Management 72:23–33.

Coluccy, J. M., R. D. Drobney, R. M. Pace III, and D. A. Graber. 2002. Consequences of neckband and legband loss from giant Canada geese. Journal of Wildlife Management 66:353–360.

Cooch, G. 1953. Techniques for mass capture of flightless blue and lesser snow geese. Journal of Wildlife Management 17:460–465.

Costanzo, G. R., R. A. Williamson, and D. E. Hayes. 1995. An efficient method for capturing flightless geese. Wildlife Society Bulletin 23:201–203.

Cox, D. R., and E. J. Snell. 1989. Analysis of binary data. Second edition. Chapman and Hall, New York, New York.

Dill, H. H., and W. H. Thornsberry. 1950. A cannon-projected net trap for capturing waterfowl. Journal of Wildlife Management 14:132–137.

Dimmick, R. W., and M. R. Pelton. 1994. Criteria of sex and age. Pages 169–214 *in* T. A. Bookhout, ed. Research and management techniques for wildlife and habitats. Fifth edition. The Wildlife Society, Bethesda, Maryland.

Eichholz, M. W., and J. S. Sedinger. 2007. Survival and recovery rate of Canada geese staging in interior Alaska. Journal of Wildlife Management 71:36–42.

Fashingbauer, R. P. 1993. Molt migration chronology and phenology of the molt of Canada geese in Northeast Minnesota. M. S. thesis, St. Cloud State University, Minnesota.

Fronczak, D. 2009. Waterfowl harvest and population survey data: estimates of U. S. harvest, hunting activity, and success derived from the State-Federal Cooperative Harvest Information Program. U.S. Fish and Wildlife Service, Division of Migratory Bird Management, Fort Snelling, Minnesota.

Hanson, D. S., and B. Hargrave. 1996. Development of a multilevel ecological classification system for the state of Minnesota. Environmental Monitoring and Assessment 39:75–84.

Hanson, H. C. 1962. Characters of age, sex, and sexual maturity in Canada geese. Illinois Natural History Survey Biological Notes 49, Urbana.

Heller, B. 2010. Analysis of giant Canada goose band recovery data in Iowa and the Mississippi flyway. M. S. thesis, Iowa State University, Ames.

Henny, C. J. and K. P. Burnham. 1976. A reward band study of mallards to estimate band reporting rates. Journal of Wildlife Management 40:1–14.

Hestbeck, J. B. 1994. Survival of Canada geese banded in winter in the Atlantic Flyway. Journal of Wildlife Management 58:748–756.

Hestbeck, J. B., and R. A. Malecki. 1989. Estimated survival rates of Canada geese within the Atlantic Flyway. Journal of Wildlife Management 53:91–96.

Hill, M. R. J., R. T. Alisauskas, C. D. Ankney, J. O. Leafloor. 2003. Influence of body size and condition on harvest and survival of juvenile Canada geese. Journal of Wildlife Management 67:530–541.

Hoeting, J. A. 2009. The importance of accounting for spatial and temporal correlation in analyses of ecological data. Ecological Applications 19:574–577.

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

Lawrence, J. S. 1997. Canada geese in Minnesota–analysis of summer banding data. Pages 173-186 *in* B. Joselyn, ed. Summaries of wildlife research findings, 1997. Minnesota Department of Natural Resources, Wildlife Populations and Research Unit, Saint Paul.

Miller, D. A., J. B. Grand, T. F. Fondell, and R. M. Anthony. 2007. Optimizing nest survival and female survival: consequences of nest site selection for Canada geese. Condor 109:769–780.

Nichols, J. D., R. E. Reynolds, R. J. Blohm, R. E. Trost, J. E. Hines, and J. P. Bladen. 1995. Geographic variation in band reporting rates for mallards based on reward banding. Journal of Wildlife Management 59:697–708.

Otis, D. L. 2004. Mallard harvest distributions in the Atlantic and Mississippi Flyways during periods of restrictive and liberal hunting regulations. Journal of Wildlife Management 68:351–359.

Otis, D. L., and G. C. White. 2002. Re-analysis of a banding study to test the effects of an experimental increase in bag limits of mourning doves. Journal of Applied Statistics 29:479–495.

Pollock, K. H., and D. G. Raveling. 1982. Assumptions of modern band-recovery models, with emphasis on heterogeneous survival rates. Journal of Wildlife Management 46:88–98.

Powell, L. A. 2008. Approximating variance of demographic parameters using the delta method: a reference for avian biologists. Condor 109:949–954.

Rave, D. P., M. H. Dexter, and J. Giudice. 2009. Hunter activity and goose harvest during the September 2008 Canada Goose Hunt in Minnesota. Pages 172–175 *in* M. H. Dexter, ed. Status of wildlife populations, fall 2009. Minnesota Department of Natural Resources, Wildlife Research Unit, Saint Paul.

Rexstad, E. A. 1992. Effects of hunting on the survival of Canada geese in Utah. Journal of Wildlife Management 56:297–305.

Samuel, M. D., D. H. Rusch, and S. Craven. 1990. Influence of neck bands on recovery and survival rates of Canada geese. Journal of Wildlife Management 54:45–54.

Seber, G. A. F. 1982. The estimation of animal abundance and related parameters. Second Edition. MacMillan, New York, New York.

Shaffer, T. L., and D. H. Johnson. 2008. Ways of learning: observational studies versus experiments. Journal of Wildlife Management 72:4–13.

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.

Sheaffer, S. E., W. L. Kendall, and E. F. Bower. 2005. Impact of special early harvest seasons on subarctic-nesting and temperate-nesting Canada geese. Journal of Wildlife Management 69:1494–1507.

Van Horne, B. 1983. Density as a misleading indicator of habitat quality. Journal of Wildlife Management 47:893–901.

White, G. C., and K. P. Burnham. 1999. Program MARK: survival estimation from populations of marked animals. Bird Study 46 (Suppl):120–138.

Williams, B. K., J. D. Nichols, and M. J. Conroy. 2001. Analysis and management of animal populations. Academic Press, San Diego, California.

Zicus, M. C. 1981. Molt migration of Canada geese from Crex Meadows, Wisconsin. Journal of Wildlife Management 45:54–63.

Zimmerman, G. S., W. L. Kendall, T. J. Moser, G. C. White, and P. F. Doherty, Jr. 2009*a*. Temporal patterns of apparent leg band retention in North American geese. Journal of Wildlife Management 73:82–88.

Zimmerman, G. S., T. J. Moser, W. L. Kendall, P. F. Doherty, Jr., G. C. White, and D. F. Caswell. 2009*b*. Factors influencing reporting and harvest probabilities in North American geese. Journal of Wildlife Management 73:710–719.

Table 1. Model selection statistics for band recovery models used to estimate recovery (f) and annual survival (S) rates of large Canada geese leg-banded in Minnesota, 2002–2007.

Model[a]	K[b]	QAICc[c]	ΔQAICc[d]	ω[e]	-2 log likelihood
$S_{\text{age + region}}$ $f_{\text{age + time, time * region, age + region}}$	58	58,986.74	0	0.96	61,399.35
$S_{\text{age + hunting zone}}$ $f_{\text{age + time, time * hunting zone, age + hunting zone}}$	44	58,992.88	6.64	0.04	61,435.58
$S_{\text{age * region}}$ $f_{\text{age * region, L- time}}$	72	58,996.24	9.99	0.01	61,380.45
$S_{\text{age * region}}$ $f_{\text{age + time, time * region, age * region}}$	72	58,999.11	12.86	0	61,383.45
$S_{\text{age + region}}$ $f_{\text{age + region + time}}$	23	59,001.34	15.10	0	61,488.30
S_{age} $f_{\text{age * region, L-time}}$	58	59,002.68	16.44	0	61,416.50
$S_{\text{age + region + time}}$ $f_{\text{age + region + time}}$	27	59,006.38	20.14	0	61,485. 20
$S_{\text{age + region}}$ $f_{\text{age + region}}$	18	59,006.71	20.47	0	61,504.34
$S_{\text{age * region}}$ $f_{\text{age * region * time}}$	112	59,008.93	22.68	0	61,309.77
S_{age} $f_{\text{age * hunting zone * time}}$	74	59,010.72	24.48	0	61,391.37
$S_{\text{age * hunting zone}}$ $f_{\text{age * hunting zone * time}}$	84	59,011.89	25.64	0	61,371.62
$S_{\text{age * region, L-time}}$ $f_{\text{age * region, L-time}}$	104	59,012.35	26.10	0	61,330.14
$S_{\text{age * hunting zone}}$ $f_{\text{age + time, time * hunting zone, age * hunting zone}}$	55	59,012.68	26.44	0	61,433.21
$S_{\text{age * hunting zone}}$ $f_{\text{age * hunting zone, L-time}}$	54	59,014.42	28.17	0	61,437.11
$S_{\text{age + hunting zone}}$ $f_{\text{age + hunting zone + time}}$	19	59,015.84	29.59	0	61,511.77
$S_{\text{age * region}}$ $f_{\text{age * region}}$	32	59,016.40	30.16	0	61,485.20
$S_{\text{age * ecological province}}$ $f_{\text{age * ecological province, L-time}}$	36	59,017.06	30.82	0	61,477.52
S_{age} $f_{\text{age * region * time}}$	98	59,018.24	31.99	0	61,348.88
$S_{\text{age + hunting zone + time}}$ $f_{\text{age + hunting zone + time}}$	23	59,020.24	34.00	0	61,508.01
$S_{\text{age + hunting zone}}$ $f_{\text{age + hunting zone}}$	14	59,021.23	34.99	0	61,527.84
S_{age} $f_{\text{age * ecological province, L-time}}$	30	59,022.73	36.49	0	61,495.98
S_{age} $f_{\text{age * region}}$	18	59,023.56	37.32	0	61,521.92
$S_{\text{age * hunting zone * time}}$ $f_{\text{age * hunting zone * time}}$	132	59,025.07	38.82	0	61,284.57
$S_{\text{age * hunting zone}}$ $f_{\text{age * hunting zone}}$	24	59,025.60	39.36	0	61,511.51
$S_{\text{age * ecological province}}$ $f_{\text{age * ecological province * time}}$	56	59,027.78	41.53	0	61,446.86
$S_{\text{age + ecological province}}$ $f_{\text{age + ecological province + time}}$	15	59,028.19	41.95	0	61,533.01
S_{age} $f_{\text{age * ecological province * time}}$	50	59,028.28	42.03	0	61,459.94
$S_{\text{age * hunting zone, L-time;}}$ $f_{\text{age * hunting zone, L-time}}$	78	59,028.61	42.36	0	61,419.12
$S_{\text{age + ecological province}}$ $f_{\text{age + time, ecological province * time, age + ecological province}}$	30	59,028.96	42.72	0	61,502.48
$S_{\text{age + ecological province + time}}$ $f_{\text{age + ecological province + time}}$	19	59,032.21	45.97	0	61,528.85
$S_{\text{age * ecological province}}$ $f_{\text{age + time, ecological province * time, age * ecological province}}$	36	59,033.09	46.85	0	61,494.25

Table 1 (continued).

Model[a]	K[b]	QAICc[c]	ΔQAICc[d]	ω[e]	-2 log likelihood
$S_{\text{age + ecological province}}$ $f_{\text{age + ecological province}}$	10	59,033.27	47.02	0	61,548.74
$S_{\text{age * ecological province}}$ $f_{\text{age * ecological province}}$	16	59,035.51	49.27	0	61,538.56
S_{age} $f_{\text{age * hunting zone}}$	14	59,036.28	50.04	0	61,543.54
$S_{\text{age * region * time}}$ $f_{\text{age * region * time}}$	176	59,037.18	50.77	0	61,204.38
S_{age} $f_{\text{age + hunting zone + time}}$	14	59,037.11	50.87	0	61,544.40
S_{age} $f_{\text{age * hunting zone, L-time}}$	44	59,037.27	51.02	0	61,481.87
S_{age} $f_{\text{age + ecological province + time}}$	12	59,040.62	54.37	0	61,552.23
S_{age} $f_{\text{age * ecological province}}$	10	59,040.78	54.54	0	61,556.58
S_{age} $f_{\text{age + region + time}}$	16	59,041.46	55.21	0	61,544.76
S_{age} $f_{\text{age + hunting zone}}$	9	59,043.52	57.27	0	61,561.52
$S_{\text{age * ecological province * time}}$ $f_{\text{age * ecological province * time}}$	88	59,055.11	68.86	0	61,408.31
S_{age} $f_{\text{age + region}}$	11	59,060.77	74.53	0	61,575.34
S_{age} $f_{\text{age + ecological province}}$	7	59,071.58	85.33	0	61,594.96
S_{age} $f_{\text{age * time}}$	14	59,099.73	113.49	0	61,609.71
S_{age} $f_{\text{age + time}}$	9	59,102.17	115.93	0	61,622.70
$S_{\text{age + time}}$ $f_{\text{age + time}}$	13	59,104.87	118.63	0	61,617.16
$S_{\text{age * time}}$ $f_{\text{age * time}}$	22	59,105.16	118.91	0	61,598.67
S_{age} f_{age}	4	59,109.02	122.78	0	61,640.28
S_{region} f_{region}	16	59,109.24	123.00	0	61,615.46
$S_{\text{hunting zone}}$ $f_{\text{hunting zone}}$	12	59,122.03	136.62	0	61,638.02
$S_{\text{ecological province}}$ $f_{\text{ecological province}}$	8	59,140.23	153.99	0	61,664.48
S_{time} f_{time}	11	59,213.91	227.67	0	61,735.07
$S_{.}$ $f_{.}$	2	59,219.05	232.81	0	61,759.21
$S_{\text{age * ecological province, L-time}}$ $f_{\text{age * ecological province, L-time}}$	52	60,058.03	1071.79	0	62,529.79

[a] Structure of the recovery and annual survival parameterization as a function of age class (AHY and L), area of capture (3 aggregations: hunting zones, ecological provinces, and combination of hunting zones and ecological provinces), and time (year).

[b] Number of parameters in the model.

[c] QAICc is the Akaike Information Criterion value for a model that has been adjusted for overdispersion and small sample size.

[d] ΔQAICc is the difference between the QAICc value of the model under consideration and that of the model with the lowest QAICc value.

[e] Model weight.

Table 2. Model-averaged estimates and associated 95% confidence intervals of recovery ($\hat{f}$) and annual survival ($\hat{S}$) rates of large Canada geese leg-banded in Minnesota, 2002–2007.

Para-meter	Region[a]	Age class[b]	Time					
			2002–2003	2003–2004	2004–2005	2005–2006	2006–2007	2007–2008
$\hat{f}$	Northwest Zone–Aspen Parkland	L	0.058 (0.024–0.134)	0.073 (0.044–0.119)	0.074 (0.056–0.099)	0.113 (0.089–0.141)	0.132 (0.105–0.164)	0.106 (0.083–0.135)
		AHY	0.049 (0.020–0.115)	0.062 (0.037–0.102)	0.063 (0.047–0.085)	0.096 (0.076–0.121)	0.113 (0.089–0.141)	0.090 (0.071–0.115)
	Rest-of-State Zone–Forest	L	0.124 (0.099–0.154)	0.171 (0.152–0.192)	0.145 (0.129–0.162)	0.165 (0.150–0.181)	0.161 (0.147–0.176)	0.161 (0.147–0.177)
		AHY	0.106 (0.084–0.133)	0.148 (0.130–0.167)	0.124 (0.110–0.139)	0.142 (0.128–0.157)	0.138 (0.125–0.152)	0.139 (0.126–0.153)
	Rest-of-State Zone–Prairie	L	0.160 (0.134–0.190)	0.164 (0.145–0.184)	0.160 (0.146–0.176)	0.164 (0.149–0.180)	0.178 (0.154–0.204)	0.158 (0.143–0.173)
		AHY	0.138 (0.115–0.165)	0.141 (0.124–0.160)	0.138 (0.124–0.153)	0.141 (0.127–0.156)	0.136 (0.123–0.150)	0.136 (0.123–0.150)
	Rest-of-State Zone–Transition	L	0.171 (0.148–0.197)	0.208 (0.186–0.231)	0.158 (0.142–0.176)	0.191 (0.176–0.208)	0.189 (0.173–0.205)	0.170 (0.155–0.185)
		AHY	0.148 (0.127–0.171)	0.180 (0.159–0.202)	0.136 (0.121–0.153)	0.165 (0.150–0.181)	0.163 (0.149–0.178)	0.146 (0.133–0.161)
	Southeast Zone–Transition	L	0.205 (0.156–0.264)	0.189 (0.156–0.227)	0.148 (0.120–0.180)	0.173 (0.144–0.206)	0.132 (0.108–0.161)	0.204 (0.167–0.246)
		AHY	0.177 (0.134–0.230)	0.163 (0.134–0.197)	0.127 (0.103–0.156)	0.149 (0.124–0.179)	0.114 (0.091–0.140)	0.176 (0.144–0.214)
	West-central Zone–Prairie	L	0.186 (0.130–0.257)	0.125 (0.088–0.177)	0.125 (0.091–0.169)	0.136 (0.101–0.180)	0.110 (0.082–0.145)	0.200 (0.156–0.253)
		AHY	0.160 (0.111–0.225)	0.107 (0.074–0.153)	0.107 (0.077–0.147)	0.117 (0.086–0.156)	0.094 (0.069–0.127)	0.173 (0.134–0.221)
	West Zone–Prairie	L	0.138 (0.113–0.166)	0.184 (0.162–0.207)	0.181 (0.160–0.203)	0.164 (0.147–0.183)	0.167 (0.150–0.186)	0.193 (0.175–0.211)
		AHY	0.118 (0.096–0.145)	0.159 (0.139–0.181)	0.156 (0.137–0.176)	0.141 (0.125–0.159)	0.144 (0.128–0.161)	0.166 (0.150–0.184)
	Metro Zone–Transition	L	—[c]	0.109 (0.084–0.142)	0.151 (0.126–0.182)	0.152 (0.127–0.182)	0.129 (0.108–0.154)	0.156 (0.131–0.185)
		AHY	—[c]	0.093 (0.071–0.121)	0.130 (0.107–0.157)	0.131 (0.109–0.156)	0.110 (0.092–0.132)	0.134 (0.112–0.160)

Table 2 (continued)

Para-meter	Region[a]	Age class[b]	Time 2002–2003 to 2007–2008
$\hat{S}$	Northwest Zone–Aspen Parkland	L	0.875 (0.806–0.922)
		AHY	0.741 (0.650–0.815)
	Rest-of-State Zone–Forest	L	0.800 (0.753–0.840)
		AHY	0.622 (0.590–0.652)
	Rest-of-State Zone–Prairie	L	0.799 (0.752–0.839)
		AHY	0.621 (0.587–0.654)
	Rest-of-State Zone–Transition	L	0.775 (0.728–0.816)
		AHY	0.587 (0.558–0.615)
	Southeast Zone–Transition	L	0.811 (0.751–0.859)
		AHY	0.640 (0.584–0.692)
	West-central Zone–Prairie	L	0.836 (0.757–0.893)
		AHY	0.679 (0.595–0.753)
	West Zone–Prairie	L	0.769 (0.719–0.813)
		AHY	0.580 (0.547–0.612)
	Metro Zone–Transition	L	0.824 (0.758–0.874)
		AHY	0.656 (0.594–0.713)

[a] Regions are comprised of a combination of hunting zones and ecological provinces.
[b] Age classes: AHY (after hatch-year) and L (local or flightless young birds).
[c] Banding operations were not conducted in the Metro Zone–Transition region during 2002.

Table 3. Model selection statistics for band recovery models in which the influence of 10 harvest regulations on recovery rates (f) and annual survival (S) rates of large Canada geese leg-banded in Minnesota during 2002–2007 were examined.

Model[a]	K[b]	QAICc[c]	ΔQAICc[d]	ω[e]	-2 log likelihood
$S_{\text{age + \# Days1 + \# Days1}^2}\ f_{\text{age + \# Days1 + \# Days1}^2}$	8	59,048.69	0	1.00	61,569.01
$S_{\text{age + \# Days1}}\ f_{\text{age + \# Days1}}$	6	59,088.99	40.30	0	61,615.21
$S_{\text{age + Total \# Days + Total \# Days}^2}\ f_{\text{age + Total \# Days + Total \# Days}^2}$	8	59,089.21	40.52	0	61,611.27
$S_{\text{age * \# Days1}}\ f_{\text{age * \#Days1}}$	8	59,091.11	42.42	0	61,613.25
$S_{\text{age + Total \# Days}}\ f_{\text{age + Total \# Days}}$	6	59,094.79	46.10	0	61,621.26
$S_{\text{age * Bag1}}\ f_{\text{age * Bag1}}$	8	59,096.15	47.46	0	61,618.50
$S_{\text{age + \# Days2 + \# Days2}^2}\ f_{\text{age + \# Days2 + \# Days2}^2}$	8	59,097.58	48.88	0	61,619.99
$S_{\text{age * Total \# Days}}\ f_{\text{age * Total \# Days}}$	8	59,097.84	49.15	0	61,620.27
$S_{\text{age + Bag2}}\ f_{\text{age + Bag2}}$	6	59,097.89	49.19	0	61,624.49
$S_{\text{age + \# Days2}}\ f_{\text{age + \# Days2}}$	6	59,098.66	49.97	0	61,625.30
$S_{\text{age + Julian2 + Julian2}^2}\ f_{\text{age + Julian2 + Julian2}^2}$	8	59,098.87	50.18	0	61,621.34
$S_{\text{age + \#DaysSplit1,2 + \#DaysSplit1,2}^2}\ f_{\text{age + \#DaysSplit1,2 + \#DaysSplit1,2}^2}$	8	59,100.36	51.67	0	61,622.89
$S_{\text{age + \#DaysSplit1,2}}\ f_{\text{age + \#DaysSplit1,2}}$	6	59,101.50	52.81	0	61,628.26
$S_{\text{age * Bag2}}\ f_{\text{age * Bag2}}$	8	59,101.87	53.18	0	61,624.47
$S_{\text{age + Bag2 + Bag2}^2}\ f_{\text{age + Bag2 + Bag2}^2}$	8	59,101.89	53.20	0	61,624.49
$S_{\text{age * \# Days2}}\ f_{\text{age * \# Days2}}$	8	59,102.22	53.53	0	61,624.84
$S_{\text{age * Bag3}}\ f_{\text{age * Bag3}}$	8	59,102.65	53.95	0	61,625.28
$S_{\text{age + Bag1}}\ f_{\text{age + Bag1}}$	6	59,104.65	55.96	0	61,631.55
$S_{\text{age * \#DaysSplit1,2}}\ f_{\text{age * \#DaysSplit1,2}}$	8	59,105.14	56.45	0	61,627.88
$S_{\text{age + Bag3}}\ f_{\text{age + Bag3}}$	6	59,106.71	58.02	0	61,633.70
$S_{\text{age + \# Days3}}\ f_{\text{age + \# Days3}}$	6	59,106.88	58.18	0	61,633.87
$S_{\text{age * \# Days3}}\ f_{\text{age * \# Days3}}$	8	59,108.22	59.53	0	61,631.10
$S_{\text{age + Bag3 + Bag3}^2}\ f_{\text{age + Bag3 + Bag3}^2}$	8	59,108.30	59.61	0	61,631.18
$S_{\text{age + Bag1 + Bag1}^2}\ f_{\text{age + Bag1 + Bag1}^2}$	8	59,108.66	59.96	0	61,631.55
$S_{\text{age + Julian1}}\ f_{\text{age + Julian1}}$	6	59,108.79	60.10	0	61,635.86
$S_{\text{age + \# Days3 + \# Days3}^2}\ f_{\text{age + \# Days3 + \# Days3}^2}$	8	59,110.88	62.19	0	61,633.87
$S_{\text{age * Julian1}}\ f_{\text{age * Julian1}}$	8	59,111.30	62.61	0	61,634.30
$S_{\text{age + Julian1 + Julian1}^2}\ f_{\text{age + Julian1 + Julian1}^2}$	8	59,112.79	64.10	0	61,635.87
$S_{\text{age + Julian2}}\ f_{\text{age + Julian2}}$	6	59,112.84	64.15	0	61,640.09
$S_{\text{age * Julian2}}\ f_{\text{age * Julian2}}$	8	59,116.62	67.93	0	61,639.85

[a] Structure of the direct-recovery rate parameterization as a function of age class (AHY and L) and a harvest regulation associated with hunting zone and year of capture. Harvest regulations are: Julian1 = Julian date, first day of early goose hunting season; Julian2 = Julian date, first day of regular goose season; Bag1 = daily bag limit during early goose season; Bag2 = daily bag limit during regular goose season; Bag3 = daily bag limit during late goose season; # Days1 = number of hunting days permitted during early goose season; # Days2 = number of hunting days permitted during regular goose season; # Days3 = number of hunting days permitted during late goose season; Total # Days = total number of days that goose hunting was permitted; # DaysSplit1,2 = number of non-hunting days between the early and regular goose seasons.

[b] Number of parameters in the model.

[c] Akaike Information Criterion value adjusted for small sample size.

[d] Difference between the AICc value of the model under consideration and that of the model with the lowest AICc value.

[e] Model weight.

Table 4. Estimates and associated 95% confidence intervals of direct-recovery (f *), recovery (f), and annual survival rates (S) of large Canada geese leg-banded in Minnesota under early goose hunting seasons of different lengths (days) during 2002–2007.

	Parameter					
	f^*		f		S	
	Age class		Age class		Age class	
Length of early goose hunting season (days)	L	AHY	L	AHY	L	AHY
10	0.063 (0.046–0.086)	0.053 (0.038–0.073)	0.069 (0.054–0.088)	0.057 (0.044–0.074)	0.912 (0.840–0.954)	0.799 (0.684–0.880)
12	0.093 (0.077–0.112)	0.078 (0.064–0.095)	0.098 (0.085–0.114)	0.082 (0.070–0.096)	0.858 (0.797–0.903)	0.699 (0.624–0.765)
13	0.108 (0.094–0.124)	0.092 (0.079–0.107)	0.113 (0.010–0.126)	0.095 (0.084–0.107)	0.832 (0.777–0.876)	0.655 (0.598–0.707)
14	0.123 (0.111–0.136)	0.105 (0.093–0.118)	0.127 (0.117–0.138)	0.107 (0.098–0.117)	0.809 (0.758–0.851)	0.618 (0.574–0.661)
15	0.137 (0.127–0.148)	0.117 (0.107–0.128)	0.140 (0.132–0.149)	0.118 (0.110–0.127)	0.791 (0.742–0.833)	0.592 (0.554–0.628)
17	0.159 (0.152–0.167)	0.136 (0.127–0.146)	0.161 (0.154–0.167)	0.136 (0.129–0.143)	0.776 (0.729–0.817)	0.571 (0.539–0.602)
19	0.169 (0.163–0.175)	0.147 (0.136–0.154)	0.170 (0.164–0.176)	0.144 (0.137–0.151)	0.793 (0.750–0.830)	0.594 (0.575–0.613)
20	0.169 (0.164–0.174)	0.145 (0.136–0.153)	0.170 (0.165–0.175)	0.144 (0.137–0.151)	0.811(0.769–0.847)	0.622 (0.603–0.641)
21	0.165 (0.160–0.170)	0.141 (0.133–0.150)	0.166 (0.161–0.172)	0.141 (0.134–0.148)	0.835 (0.790–0.871)	0.659 (0.626–0.690)
22	0.159 (0.152–0.166)	0.136 (0.127–0.145)	0.160 (0.154–0.167)	0.136 (0.128–0.143)	0.861 (0.811–0.900)	0.704 (0.650–0.752)

Table 5. Model selection statistics for band recovery models used to estimate direct-recovery rates ($f*$) of large Canada geese leg-banded in Minnesota, 2002–2007.

Model[a]	K[b]	AICc [c]	ΔAICc [d]	ω [e]	-2 log likelihood
$f*_{\text{age + time, time * region, age + region}}$	49	28,375.26	0	0.74	28,277.11
$f*_{\text{age + time, time * region, age * region}}$	56	28,377.36	2.09	0.26	28,265.16
$f*_{\text{age + time, time * hunting zone, age * hunting zone}}$	37	23,386.85	11.59	0	28,312.77
$f*_{\text{age * region, L-time}}$	55	28,386.93	11.66	0	28,276.74
$f*_{\text{age * hunting zone * time}}$	70	28,397.18	21.92	0	28,256.88
$f*_{\text{age * region * time}}$	94	28,399.50	24.24	0	28,210.96
$f*_{\text{age + region + time}}$	14	28,401.58	26.32	0	28,373.57
$f*_{\text{age + region}}$	9	28,401.60	26.33	0	28,383.59
$f*_{\text{age + hunting zone + time}}$	12	28,402.51	27.24	0	28,378.50
$f*_{\text{age * hunting zone, L-time}}$	41	28,402.65	27.39	0	28,320.55
$f*_{\text{age * region}}$	16	28,408.24	32.98	0	28,376.22
$f*_{\text{age + hunting zone}}$	7	28,411.96	36.70	0	28,397.96
$f*_{\text{age * hunting zone}}$	12	28,415.43	40.17	0	28,391.43
$f*_{\text{age * ecological province * time}}$	48	28,419.56	44.29	0	28,323.41
$f*_{\text{age + time, time * ecological province, age * ecological province}}$	28	28,420.10	44.84	0	28,364.05
$f*_{\text{age * ecological province, L-time}}$	32	28,420.40	45.14	0	28,356.34
$f*_{\text{age + time, time * ecological province, age + ecological province}}$	25	28,420.66	45.40	0	28,420.66
$f*_{\text{region}}$	8	28,422.06	46.80	0	28,406.05
$f*_{\text{hunting zone}}$	6	28,432.04	56.78	0	28,420.04
$f*_{\text{age + ecological province}}$	5	28,432.28	57.02	0	28,422.28
$f*_{\text{age * ecological province}}$	8	28,433.82	58.56	0	28,417.81
$f*_{\text{ecological province}}$	4	28,454.81	79.55	0	28,446.81
$f*_{\text{age + ecological province + time}}$	10	28,475.06	99.80	0	28,455.06
$f*_{\text{age + time}}$	7	28,492.33	117.07	0	28,478.33
$f*_{\text{age * time}}$	12	28,492.66	117.40	0	28,468.65
$f*_{\text{age}}$	2	28,504.19	128.93	0	28,500.19
$f*_{\text{time}}$	6	28,516.14	140.88	0	28,504.14
$f*_{.}$	1	28,528.36	153.10	0	28,526.36

[a] Structure of the direct-recovery rate parameterization as a function of age class (AHY and L), time (year), and area of capture (3 aggregations: hunting zones, ecological provinces, and combination of hunting zones and ecological provinces).

[b] Number of parameters in the model.

[c] Akaike Information Criterion value adjusted for small sample size.

[d] Difference between the AICc value of the model under consideration and that of the model with the lowest AICc value.

[e] Model weight.

Table 6. Model-averaged estimates of direct-recovery rates and associated 95% confidence intervals of large Canada geese leg-banded in Minnesota, 2002–2007.

		Time					
Region [a]	Age class[b]	2002–2003	2003–2004	2004–2005	2005–2006	2006–2007	2007–2008
Northwest Zone–Aspen Parkland	L	0.058 (0.024–0.132)	0.074 (0.042–0.129)	0.067 (0.047–0.094)	0.125 (0.089–0.173)	0.125 (0.088–0.175)	0.071 (0.047–0.107)
	AHY	0.050 (0.021–0.115)	0.061 (0.034–0.109)	0.056 (0.038–0.080)	0.106 (0.074–0.151)	0.104 (0.071–0.148)	0.060 (0.039–0.092)
Rest-of-State Zone–Forest	L	0.116 (0.092–0.145)	0.169 (0.148–0.193)	0.143 (0.123–0.165)	0.163 (0.143–0.184)	0.158 (0.140–0.178)	0.165 (0.145–0.188)
	AHY	0.106 (0.079–0.142)	0.147 (0.125–0.171)	0.119 (0.099–0.144)	0.137 (0.116–0.162)	0.135 (0.116–0.157)	0.138 (0.116–0.163)
Rest-of-State Zone–Prairie	L	0.155 (0.130–0.185)	0.156 (0.135–0.180)	0.156 (0.139–0.175)	0.174 (0.151–0.198)	0.173 (0.151–0.197)	0.161 (0.138–0.186)
	AHY	0.133 (0.108–0.162)	0.143 (0.111–0.182)	0.134 (0.114–0.156)	0.151 (0.127–0.178)	0.145 (0.121–0.173)	0.135 (0.112–0.163)
Rest-of-State Zone–Transition	L	0.171 (0.147–0.198)	0.215 (0.187–0.245)	0.155 (0.135–0.178)	0.195 (0.175–0.217)	0.196 (0.176–0.217)	0.144 (0.126–0.163)
	AHY	0.147 (0.124–0.173)	0.181 (0.150–0.216)	0.132 (0.111–0.158)	0.180 (0.146–0.220)	0.169 (0.147–0.193)	0.124 (0.107–0.144)
Southeast Zone–Transition	L	0.193 (0.145–0.251)	0.189 (0.151–0.234)	0.157 (0.119–0.203)	0.161 (0.123–0.207)	0.130 (0.097–0.173)	0.238 (0.174–0.317)
	AHY	0.163 (0.122–0.214)	0.160 (0.124–0.204)	0.134 (0.100–0.177)	0.134 (0.098–0.180)	0.111 (0.082–0.149)	0.220 (0.153–0.306)
West-central Zone–Prairie	L	0.201 (0.142–0.277)	0.141 (0.093–0.209)	0.124 (0.081–0.187)	0.113 (0.069–0.179)	0.093 (0.061–0.139)	0.224 (0.149–0.322)
	AHY	0.173 (0.120–0.245)	0.122 (0.079–0.185)	0.104 (0.066–0.160)	0.094 (0.056–0.154)	0.078 (0.050–0.120)	0.189 (0.121–0.282)
West Zone–Prairie	L	0.142 (0.116–0.171)	0.188 (0.161–0.217)	0.168 (0.143–0.197)	0.178 (0.157–0.202)	0.149 (0.127–0.174)	0.209 (0.186–0.233)
	AHY	0.121 (0.097–0.150)	0.173 (0.135–0.218)	0.144 (0.119–0.174)	0.155 (0.132–0.181)	0.125 (0.102–0.152)	0.178 (0.153–0.206)
Metro Zone [c] –Transition	L	—	0.108 (0.083–0.140)	0.138 (0.107–0.177)	0.153 (0.120–0.193)	0.112 (0.085–0.145)	0.144 (0.112–0.185)
	AHY	—	0.092 (0.070–0.120)	0.115 (0.086–0.151)	0.130 (0.100–0.168)	0.102 (0.075–0.137)	0.123 (0.094–0.161)

[a] Regions are comprised of a combination of hunting zones and ecological provinces.

[b] Age classes: AHY (after hatch-year) and L (local or flightless young birds).

[c] Banding operations were not conducted in the Metro Zone–Transition region during 2002.

Table 7. Model selection statistics for band-recovery models in which we examined the influence of 10 harvest regulations on the direct-recovery rates (f *****) of large Canada geese leg-banded in Minnesota, 2002–2007.

Model [a]	K [b]	AICc [c]	ΔAICc [d]	ω [e]	-2 log likelihood
$f*_{\text{age + \# Days1 + \# Days1}^2}$	4	28,462.25	0	1.0	28,454.25
$f*_{\text{age + Julian2 + \# Julian2}^2}$	4	28,488.07	25.81	0	28,480.07
$f*_{\text{age + Total \# Days + Total \# Days}^2}$	4	28,492.85	30.60	0	28,484.85
$f*_{\text{age * \# Days1}}$	4	28,492.90	30.65	0	28,484.90
$f*_{\text{age + \# Days1}}$	3	28,493.21	30.96	0	28,487.21
$f*_{\text{age * Total \# Days}}$	4	28,495.83	33.58	0	28,487.83
$f*_{\text{age + \#DaysSplit1,2+ \#DaysSplit1,2}^2}$	4	28,497.15	34.90	0	28,489.15
$f*_{\text{age * Bag1}}$	4	28,497.42	35.17	0	28,489.42
$f*_{\text{age + Bag1}}$	3	28,497.59	35.34	0	28,491.59
$f*_{\text{age + Total \# Days}}$	3	28,497.87	35.61	0	28,491.87
$f*_{\text{age * \# Days2}}$	4	28,499.25	37.00	0	28,491.25
$f*_{\text{age + Bag1+ Bag1}^2}$	4	28,499.59	37.34	0	28,491.59
$f*_{\text{age + \#DaysSplit1,2}}$	3	28,500.73	38.48	0	28,494.73
$f*_{\text{age + \# Days2}}$	3	28,500.79	38.54	0	28,494.79
$f*_{\text{age * Bag2}}$	4	28,501.27	39.02	0	28,493.27
$f*_{\text{age + Bag2}}$	3	28.501.82	39.56	0	28,495.82
$f*_{\text{age * Julian1}}$	4	28,502.13	39.88	0	28,494.13
$f*_{\text{age * \#DaysSplit1,2}}$	4	28,502.47	40.21	0	28,494.47
$f*_{\text{age + \# Days2 + \# Days2}^2}$	4	28,502.50	40.25	0	28,494.50
$f*_{\text{age + \# Days3}}$	3	28,502.89	40.64	0	28,496.89
$f*_{\text{age * Bag3}}$	4	28,503.21	40.95	0	28,495.20
$f*_{\text{age * \# Days3}}$	4	28,503.74	41.48	0	28,495.74
$f*_{\text{age + Bag2+ Bag2}^2}$	4	28,503.82	41.56	0	28,495.82
$f*_{\text{age + Bag3+ Bag3}^2}$	4	28,503.88	41.62	0	28,495.88
$f*_{\text{age + \# Days3 + \# Days3}^2}$	4	28,504.89	42.64	0	28,496.89
$f*_{\text{age + Bag3}}$	3	28,505.67	43.42	0	28,499.67
$f*_{\text{age + Julian1}}$	3	28,505.93	43.68	0	28,499.93
$f*_{\text{age + Julian2}}$	3	28,506.19	43.94	0	28,506.19
$f*_{\text{age + Julian1+ Julian1}^2}$	4	28,507.93	45.67	0	28,499.93
$f*_{\text{age * Julian2}}$	4	28,508.09	45.84	0	28,500.09

[a] Structure of the direct-recovery rate parameterization as a function of age class (AHY and L) and a harvest regulation associated with the hunting zone and year of capture. Harvest regulations are: Julian1 = Julian date of the first day of early goose hunting season, Julian2 = Julian date of the first day of regular goose season, Bag1 = daily bag limit during the early goose season, Bag2 = daily bag limit during the regular goose season, Bag3 = daily bag limit during the late goose season, # Days1 = the number of hunting days permitted during the early goose season, # Days2 = the number of hunting days permitted during the regular goose season, # Days3 = the number of hunting days permitted during the late goose season, Total # Days = the total number of days that goose hunting was permitted, # DaysSplit1,2 = the number of non-hunting days between the early and regular goose seasons.

[b] Number of parameters in the model.

[c] Akaike Information Criterion value adjusted for small sample size.

[d] Difference between the AICc value of the model under consideration and that of the model with the lowest AICc value.

[e] Model weight.

Table 8. Harvest rate estimates and associated 95% confidence intervals of large Canada geese banded in Minnesota, 2002–2007.

Region [a]	Age class[b]	Time					
		2002–2003	2003–2004	2004–2005	2005–2006	2006–2007	2007–2008
Northwest Zone–Aspen Parkland	L	0.077 (0.011–0.143)	0.099 (0.042–0.156)	0.089 (0.057–0.121)	0.166 (0.108–0.224)	0.166 (0.107–0.225)	0.095 (0.054–0.135)
	AHY	0.066 (0.009–0.123)	0.082 (0.033–0.130)	0.074 (0.046–0.101)	0.141 (0.089–0.193)	0.137 (0.086–0.188)	0.080 (0.045–0.115)
Rest-of-State Zone–Forest	L	0.154 (0.116–0.192)	0.224 (0.187–0.262)	0.189 (0.155–0.223)	0.216 (0.181–0.251)	0.210 (0.177–0.243)	0.219 (0.184–0.255)
	AHY	0.141 (0.104–0.177)	0.195 (0.159–0.231)	0.158 (0.126–0.191)	0.182 (0.148–0.216)	0.180 (0.147–0.212)	0.183 (0.150–0.216)
Rest-of-State Zone–Prairie	L	0.206 (0.164–0.248)	0.207 (0.170–0.243)	0.207 (0.175–0.239)	0.230 (0.191–0.269)	0.229 (0.191–0.268)	0.213 (0.174–0.252)
	AHY	0.176 (0.136–0.216)	0.190 (0.153–0.227)	0.177 (0.145–0.210)	0.200 (0.161–0.239)	0.193 (0.156–0.230)	0.180 (0.142–0.217)
Rest-of-State Zone–Transition	L	0.227 (0.186–0.268)	0.285 (0.237–0.333)	0.206 (0.170–0.241)	0.259 (0.222–0.296)	0.259 (0.221–0.298)	0.191 (0.159–0.222)
	AHY	0.195 (0.156–0.233)	0.240 (0.194–0.285)	0.176 (0.140–0.211)	0.238 (0.200–0.277)	0.224 (0.186–0.261)	0.165 (0.135–0.195)
Southeast Zone–Transition	L	0.256 (0.182–0.329)	0.250 (0.189–0.311)	0.208 (0.148–0.267)	0.213 (0.153–0.273)	0.173 (0.120–0.226)	0.316 (0.216–0.415)
	AHY	0.216 (0.152–0.280)	0.212 (0.156–0.268)	0.178 (0.124–0.231)	0.177 (0.123–0.231)	0.147 (0.100–0.194)	0.292 (0.196–0.388)
West-central Zone–Prairie	L	0.266 (0.172–0.360)	0.187 (0.109–0.266)	0.165 (0.094–0.236)	0.150 (0.077–0.223)	0.123 (0.070–0.176)	0.297 (0.179–0.416)
	AHY	0.230 (0.144–0.316)	0.162 (0.091–0.233)	0.138 (0.075–0.200)	0.125 (0.061–0.189)	0.104 (0.057–0.151)	0.250 (0.144–0.357)
West Zone–Prairie	L	0.188 (0.147–0.229)	0.249 (0.205–0.293)	0.223 (0.181–0.266)	0.236 (0.198–0.275)	0.198 (0.160–0.235)	0.277 (0.235–0.319)
	AHY	0.160 (0.122–0.198)	0.229 (0.184–0.274)	0.192 (0.151–0.233)	0.205 (0.167–0.244)	0.166 (0.130–0.201)	0.236 (0.195–0.277)
Metro Zone[c] –Transition	L	—	0.144 (0.103–0.184)	0.183 (0.134–0.233)	0.203 (0.150–0.255)	0.148 (0.108–0.189)	0.191 (0.139–0.244)
	AHY	—	0.122 (0.086–0.158)	0.152 (0.108–0.196)	0.173 (0.125–0.221)	0.135 (0.097–0.173)	0.164 (0.116–0.211)

[a] Regions are comprised of a combination of hunting zones and ecological provinces.

[b] Age classes: AHY (after hatch-year) and L (local or flightless young birds).

[c] Banding operations were not conducted in the Metro Zone–Transition region during 2002.

Table 9. Distribution of direct recoveries [a] (n = 5,131) of large Canada geese banded in each region of Minnesota during the 2002–2003 to 2007–2008 hunting seasons (all years pooled).

		Region of recovery [b]								
Region of banding [b]	n [c]	Northwest Zone–Aspen Parkland	Rest-of-State Zone–Forest	Rest-of-State Zone–Prairie	Rest-of-State Zone–Transition	Southeast Zone–Transition	West-central Zone-Prairie	West Zone-Prairie	Metro Zone-Transition	Out-of-State
Northwest Zone–Aspen Parkland	128	0.461	0.039	0.157	0	0	0	0.039	0.031	0.273
Rest-of-State Zone–Forest	1103	0.004	0.660	0.015	0.057	0.006	0	0.005	0.018	0.235
Rest-of-State Zone–Prairie	925	0.001	0.010	0.563	0.147	0.005	0.001	0.029	0.029	0.214
Rest-of-State Zone–Transition	1372	0	0.034	0.089	0.611	0.013	0.004	0.015	0.055	0.180
Southeast Zone–Transition	280	0	0	0.004	0.004	0.639	0	0.004	0	0.350
West-central Zone–Prairie	120	0	0	0	0.017	0	0.483	0.083	0	0.417
West Zone–Prairie	928	0	0.006	0.061	0.017	0	0.080	0.481	0.002	0.352
Metro Zone–Transition	275	0	0	0	0.153	0.025	0	0	0.542	0.280

[a] Distribution of direct recoveries has not been adjusted for spatiotemporal differences in band reporting rates (see Otis 2004).

[b] Regions are comprised of a combination of hunting zones and ecological provinces.

[c] Number of direct recoveries.

Table 10. Distribution of direct recoveries [a] (n = 5,191) of large Canada geese (all ages and sexes combined) leg-banded in each region of Minnesota during the 2002–2003 to 2007–2008 hunting seasons.

Region [b]	Year of banding	n [c]	Proportion						
			MN	IA	IL	MO	ND	SD	Other
Northwest Zone–Aspen Parkland	2002	5	1.000	0	0	0	0	0	0
	2003	11	0.545	0.091	0.273	0.091	0	0	0
	2004	30	0.833	0.033	0.033	0.067	0.033	0	0
	2005	30	0.600	0.067	0	0.300	0	0	0.033
	2006	29	0.690	0.103	0.034	0.103	0.034	0	0.034
	2007	21	0.857	0.048	0	0.095	0	0	0
Weighted $\bar{x}$		126	0.730	0.063	0.040	0.135	0.016	0	0.016
Rest-of-State Zone–Forest	2002	71	0.915	0	0.014	0.070	0	0	0
	2003	190	0.747	0.042	0.158	0.032	0	0	0.021
	2004	160	0.688	0.025	0.194	0.025	0	0	0.069
	2005	211	0.787	0.009	0.100	0.071	0	0	0.033
	2006	238	0.756	0.029	0.151	0.013	0	0	0.050
	2007	217	0.788	0.028	0.101	0.060	0	0	0.023
Weighted $\bar{x}$		1087	0.767	0.025	0.130	0.042	0	0	0.036
Rest-of-State Zone–Prairie	2002	106	0.868	0.019	0	0.038	0.067	0	0.009
	2003	168	0.774	0.113	0.012	0.054	0.030	0	0.018
	2004	239	0.833	0.025	0.008	0.038	0.038	0.021	0.038
	2005	175	0.737	0.063	0.023	0.097	0.069	0.011	0
	2006	186	0.833	0.038	0	0.059	0.027	0.016	0.027
	2007	143	0.706	0.105	0.007	0.098	0.042	0	0.042
Weighted $\bar{x}$		1017	0.793	0.059	0.009	0.063	0.043	0.010	0.024
Rest-of-State Zone–Transition	2002	155	0.877	0.052	0.006	0.032	0	0.013	0.019
	2003	173	0.872	0.035	0	0.087	0	0	0.006
	2004	170	0.906	0.006	0.024	0.006	0.041	0.006	0.012
	2005	328	0.841	0.030	0.027	0.055	0.030	0.003	0.012
	2006	293	0.785	0.061	0.017	0.051	0.061	0.007	0.017
	2007	211	0.668	0.047	0.076	0.104	0.043	0.028	0.033
Weighted $\bar{x}$		1330	0.818	0.040	0.026	0.057	0.033	0.009	0.017

Table 10 (continued).

Region [b]	Year of banding	*n* [c]	Proportion						
			MN	IA	IL	MO	ND	SD	Other
Southeast Zone–Transition	2002	44	0.955	0.045	0	0	0	0	0
	2003	65	0.569	0	0.200	0.046	0	0	0.185
	2004	47	0.617	0.021	0.106	0	0	0	0.255
	2005	48	0.604	0.021	0.042	0	0	0	0.333
	2006	43	0.674	0.023	0.093	0.047	0	0	0.163
	2007	33	0.485	0	0.242	0	0	0.030	0.242
Weighted $\bar{x}$		280	0.650	0.018	0.114	0.018	0	0.004	0.196
West-central Zone –Prairie	2002	27	0.667	0	0	0.037	0.185	0.111	0
	2003	20	0.650	0	0	0.100	0.050	0.100	0.100
	2004	20	0.650	0	0	0.050	0	0.300	0
	2005	16	0.438	0.125	0	0.063	0	0.188	0.188
	2006	20	0.550	0	0	0	0	0.450	0
	2007	20	0.450	0	0	0	0.150	0.400	0
Weighted $\bar{x}$		123	0.577	0.016	0	0.041	0.073	0.252	0.041
West Zone–Prairie	2002	92	0.663	0.011	0	0.065	0.152	0.076	0.033
	2003	153	0.634	0.039	0.007	0.111	0.052	0.137	0.020
	2004	128	0.727	0.031	0.008	0.016	0.070	0.102	0.047
	2005	201	0.557	0.024	0.010	0.075	0.090	0.199	0.045
	2006	131	0.756	0.084	0	0.031	0.031	0.069	0.031
	2007	250	0.660	0.012	0.004	0.068	0.052	0.168	0.036
Weighted $\bar{x}$		955	0.657	0.031	0.005	0.064	0.069	0.138	0.036
Metro Zone–Transition	2003	52	0.615	0.115	0.135	0.096	0	0	0.038
	2004	55	0.782	0	0.164	0.055	0	0	0
	2005	59	0.661	0.018	0.119	0.203	0	0	0
	2006	54	0.852	0	0.093	0.019	0	0	0.037
	2007	53	0.660	0.018	0.226	0.094	0	0	0
Weighted $\bar{x}$		273	0.714	0.029	0.147	0.095	0	0	0.015
Statewide weighted $\bar{x}$			0.750	0.037	0.051	0.058	0.032	0.036	0.036

[a] Distribution of direct recoveries have not been adjusted for spatiotemporal differences in band reporting rates (see Otis 2004).
[b] Regions are comprised of a combination of hunting zones and ecological provinces.
[c] Number of direct recoveries.

Table 11. The proportion of leg-banded large Canada geese that were captured in each region of Minnesota and directly-recovered in-state during 3 time periods (1–22 September, 23 September–30 November, 1 December–21 February) of the 2002–2003 to 2007–2008 hunting seasons.

		Proportion directly-recovered in-state by time period		
Region [a]	*n* [b]	1–22 September	23 September–30 November	1 December–21 February
Northwest Zone–Aspen Parkland	126	0.932	0.918	0.182
Rest-of-State Zone–Forest	1079	0.992	0.863	0.113
Rest-of-State Zone–Prairie	1014	0.964	0.769	0.276
Rest-of-State Zone–Transition	1321	0.958	0.867	0.288
Southeast Zone–Transition	280	0.663	0.815	0.403
West-central Zone–Prairie[c]	120	0.814	0.447	0.071
West Zone–Prairie	951	0.800	0.609	0.203
Metro Zone–Transition	272	0.988	0.958	0.242
Weighted $\bar{x}$		0.918	0.797	0.230

[a] Regions are comprised of a combination of hunting zones and ecological provinces.

[b] Number of direct recoveries (overall *n* = 5,163).

[c] No December hunting season.

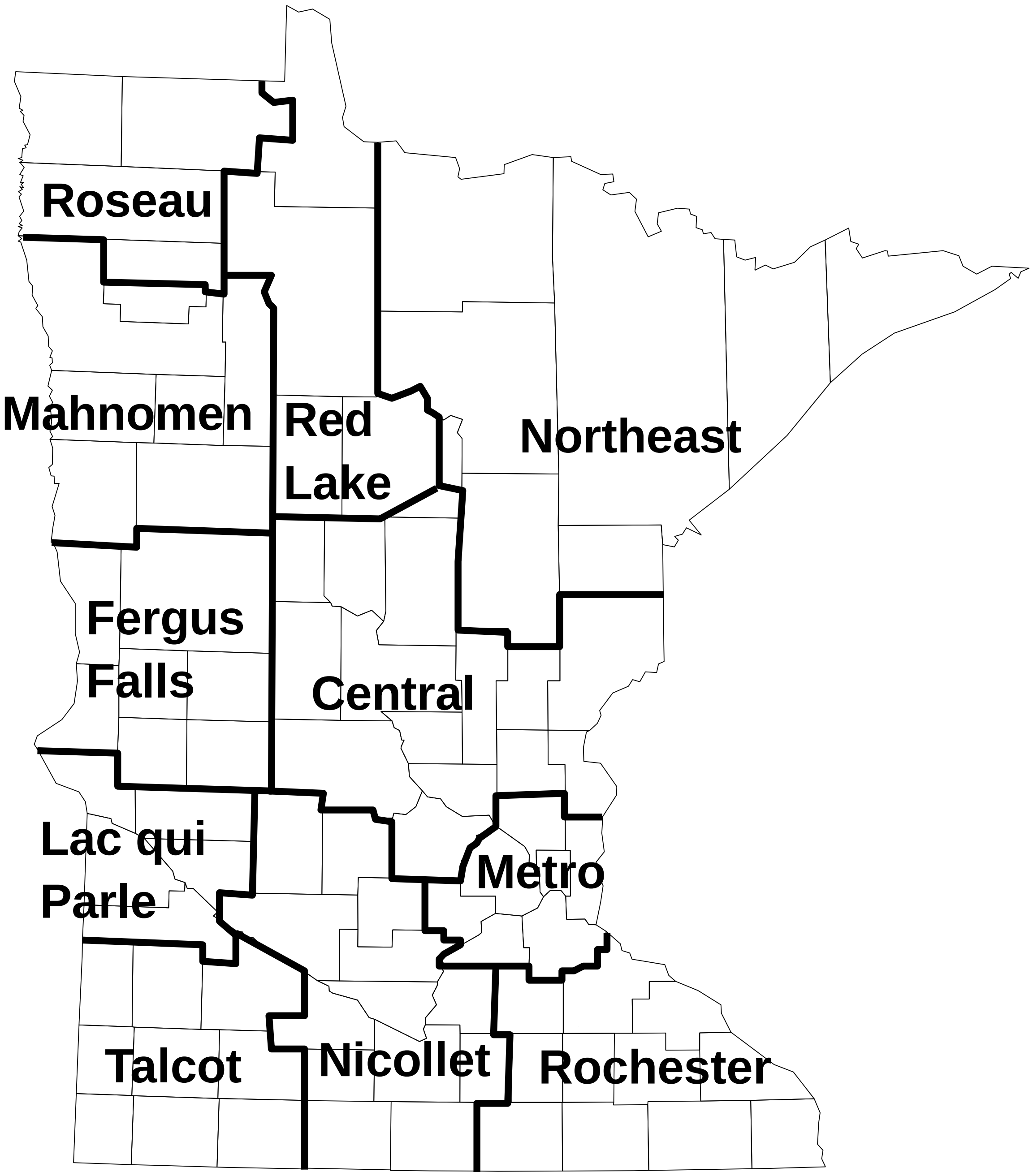


Figure 1. Goose Management Blocks of Minnesota in which large Canada geese were leg-banded during 2002-2007.

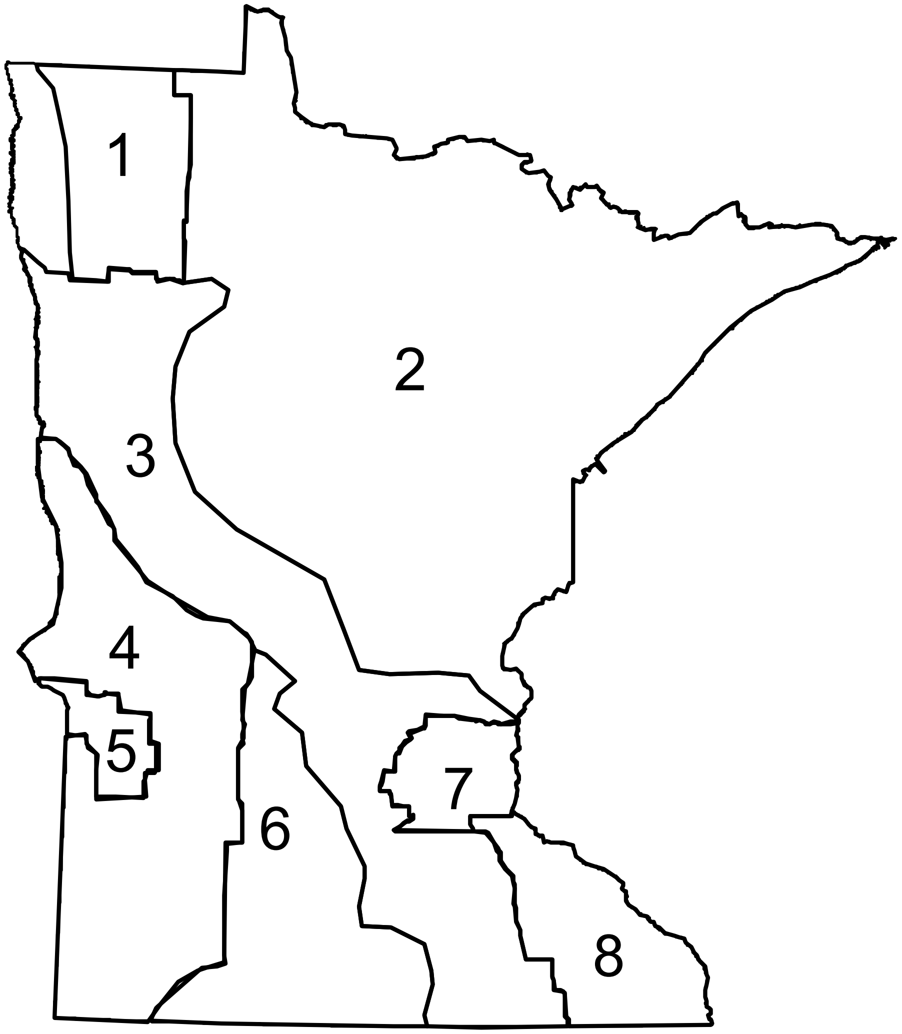


Figure 2. The regions (combinations of ecological provinces and hunting zones) of Minnesota in which large Canada geese were leg-banded during 2002–2007. The regions are: 1 = Northwest Zone–Aspen Parkland, 2 = Rest-of-State Zone–Forest, 3 = Rest-of-State Zone–Transition, 4 = West Zone–Prairie, 5 = Westcentral Zone–Prairie, 6 = Rest-of-State Zone–Prairie, 7 = Metro Zone–Transition, 8 = Southeast Zone–Transition.

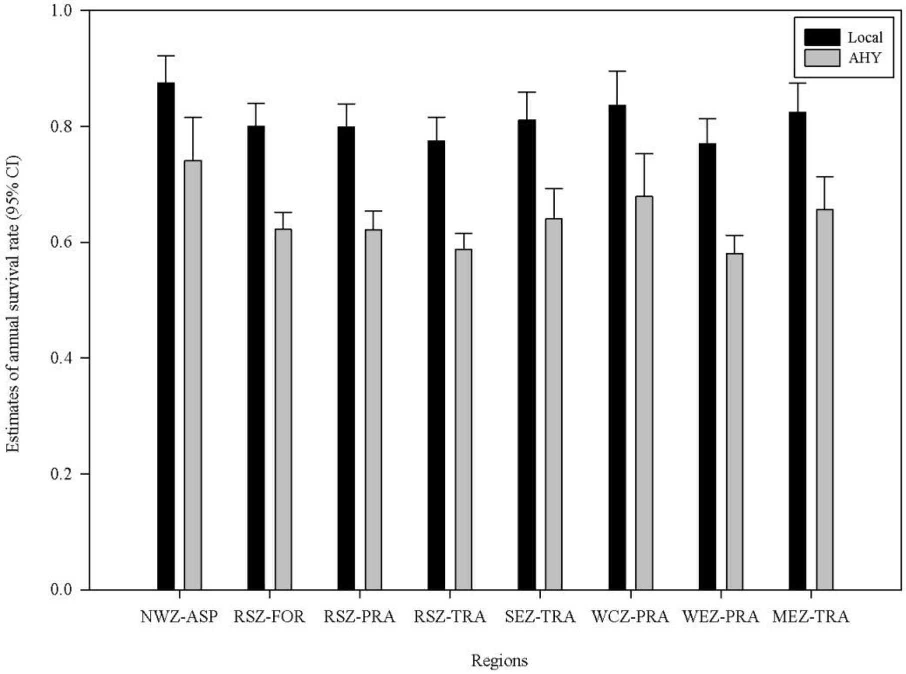


Figure 3. The age-specific annual survival rate estimates of large Canada geese captured in 8 regions (NWZ-ASP [Northwest Zone–Aspen Parkland], RSZ-FOR [Rest-of-State Zone–Forest], RSZ-PRA [Rest-of-State Zone–Prairie], RSZ-TRA [Rest-of-State Zone–Transition], SEZ-TRA [Southeast Zone–Transition], WCZ-PRA [West-central Zone–Prairie], WEZ-PRA [West Zone–Prairie], MEZ-TRA [Metro Zone–Transition]) of Minnesota during 2002–2007. Regions are a combination of hunting zones and ecological provinces.

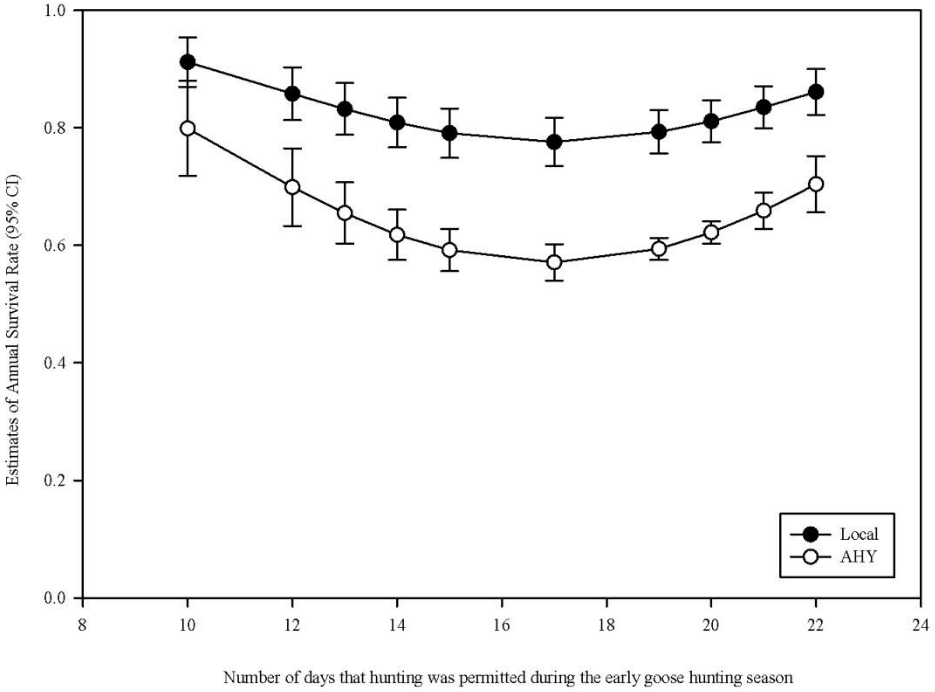


Figure 4. The influence of the number of days that hunting was permitted during the early goose hunting season on age-specific annual survival rates of large Canada geese captured in Minnesota during 2002–2007.

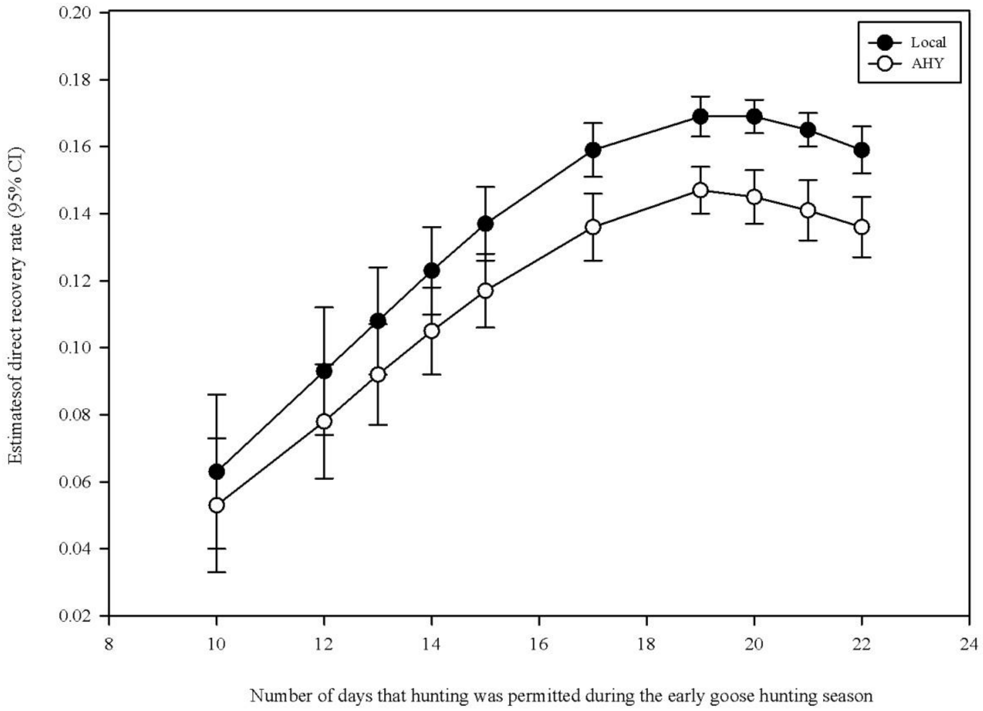


Figure 5. The influence of the number of days that hunting was permitted during the early goose hunting season on the direct recovery rates of large Canada geese captured in Minnesota during 2002–2007.

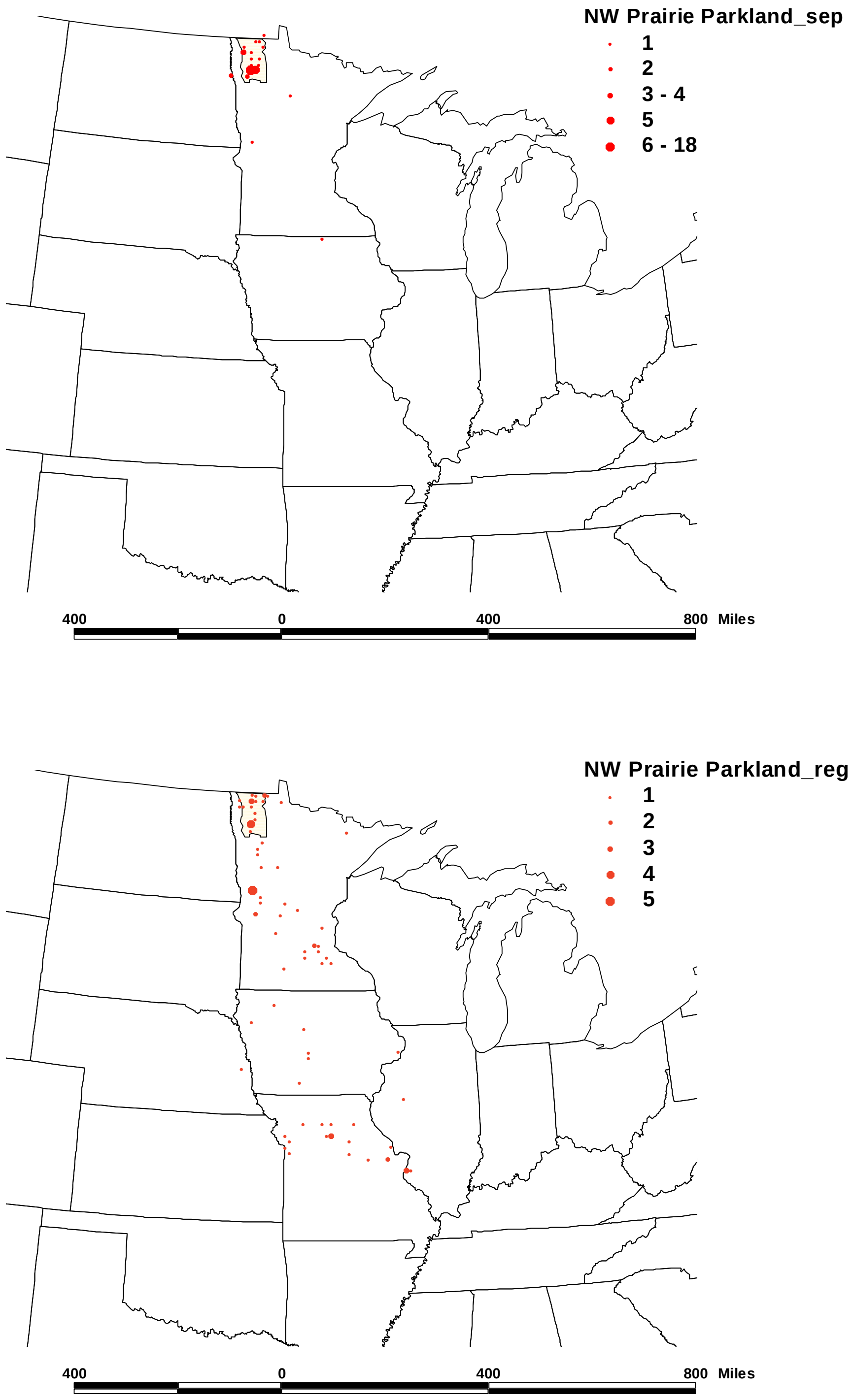


Figure 6. Distribution of direct recoveries of large Canada geese banded in the Northwest Zone –Aspen Parkland region and recovered during the 1–22 September (top) and 23 September–21 February (bottom) portions of the 2002–2003 to 2007–2008 hunting seasons. Figure excludes one regular season recovery in Manitoba.

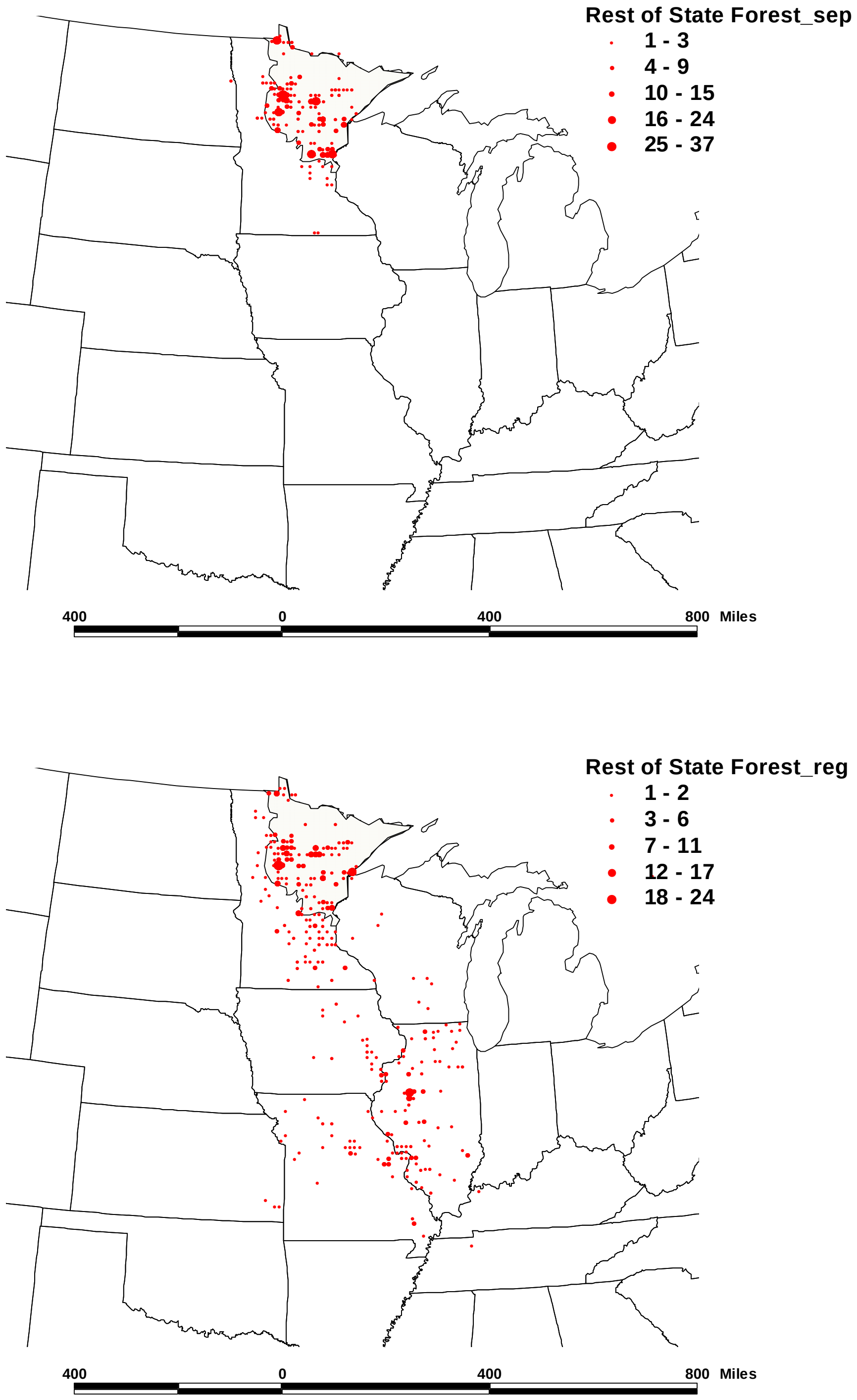


Figure 7. Distribution of direct recoveries of large Canada geese banded in the Rest-of-State Zone–Forest region and recovered during the 1–22 September (top) and 23 September–21 February (bottom) portions of the 2002–2003 to 2007–2008 hunting seasons. Figure excludes 1 early season recovery in New Jersey, 1 regular season recovery in Manitoba, and 1 regular season recovery in Ontario.

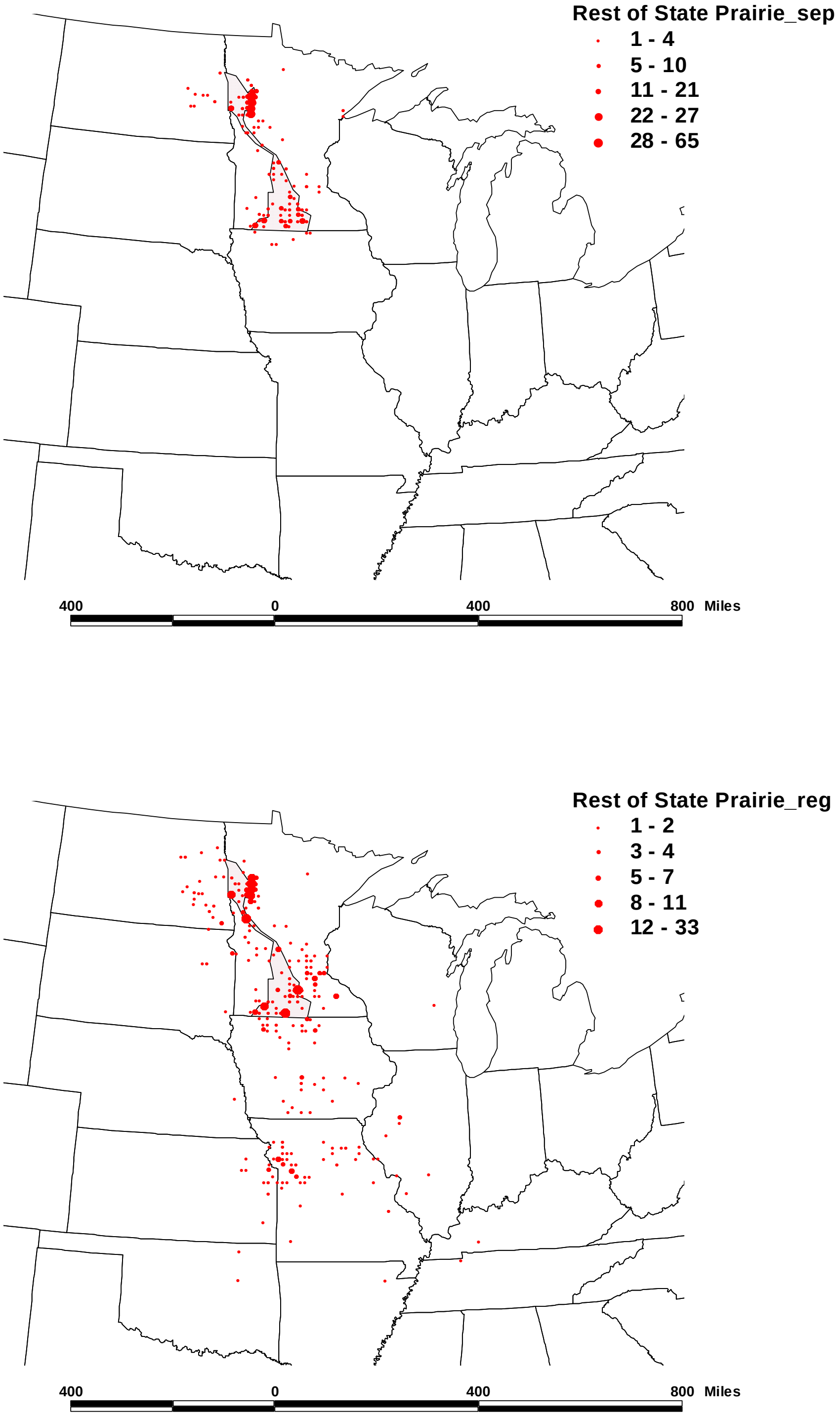


Figure 8. Distribution of direct recoveries of large Canada geese banded in the Rest-of-State Zone–Prairie region and recovered during the 1–22 September (top) and 23 September–21 February (bottom) portions of the 2002–2003 to 2007–2008 hunting seasons. Figure excludes 1 early season recovery in Montana and 1 early season recovery in Virginia.

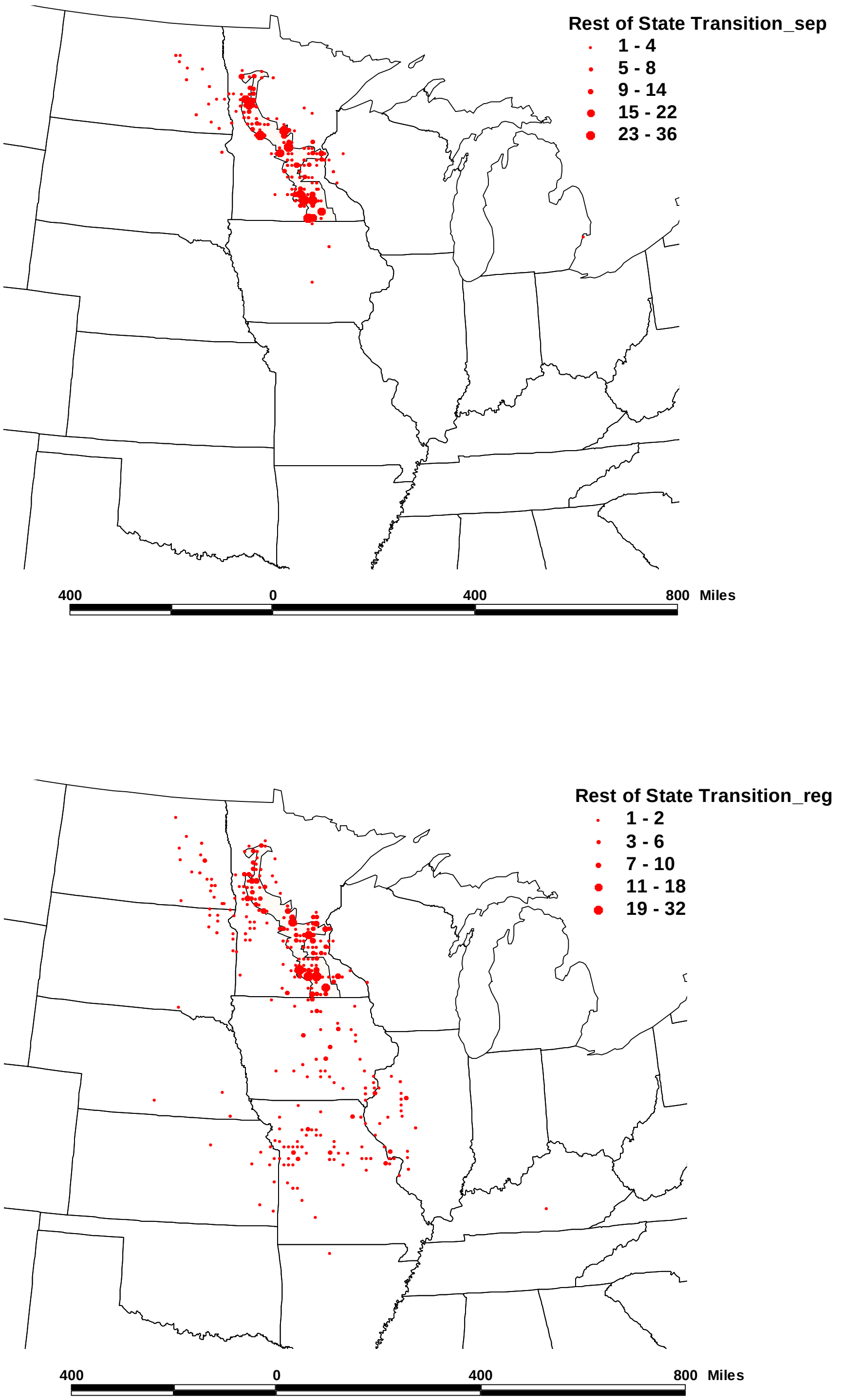


Figure 9. Distribution of direct recoveries of large Canada geese banded in the Rest-of-State Zone–Transition region and recovered during the 1–22 September (top) and 23 September–21 February (bottom) portions of the 2002–2003 to 2007–2008 hunting seasons.

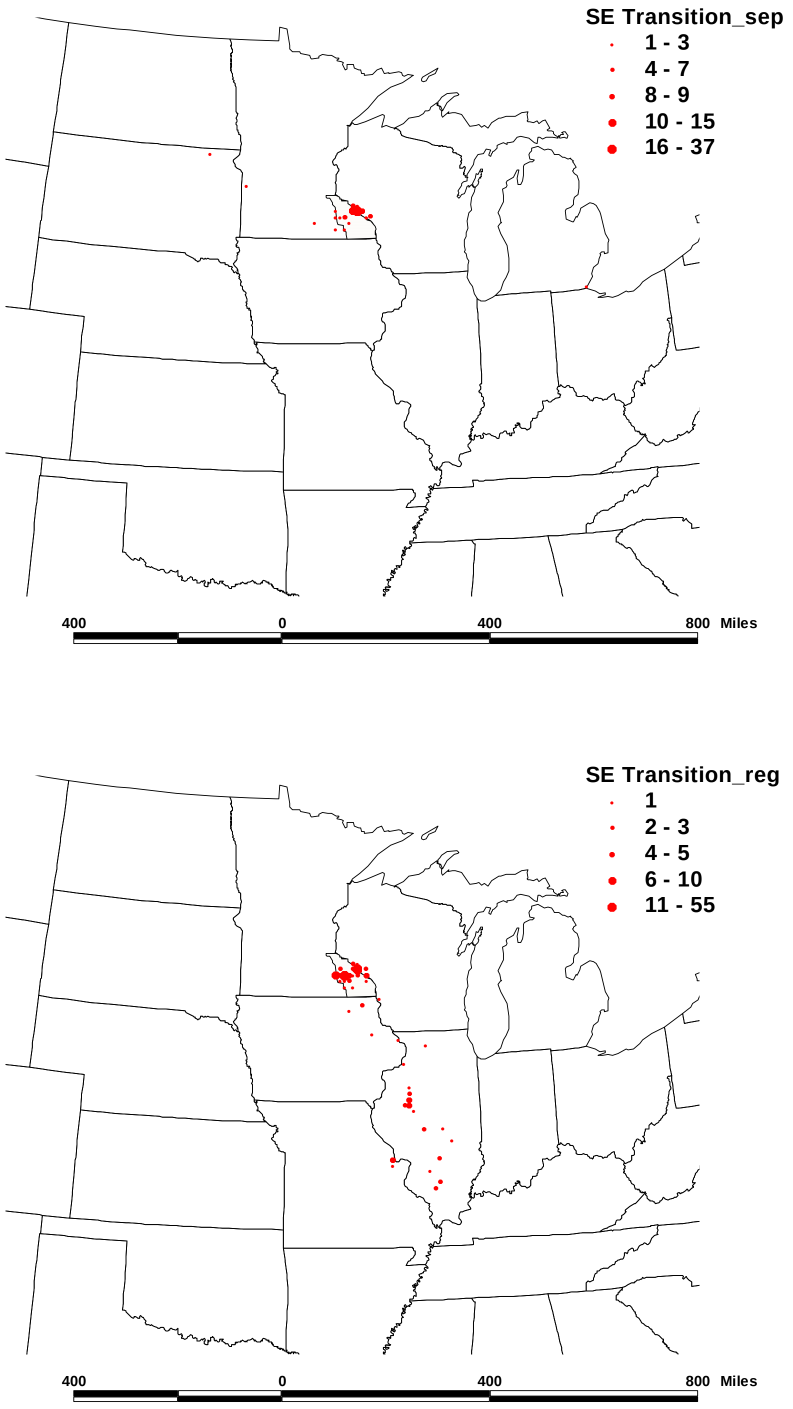
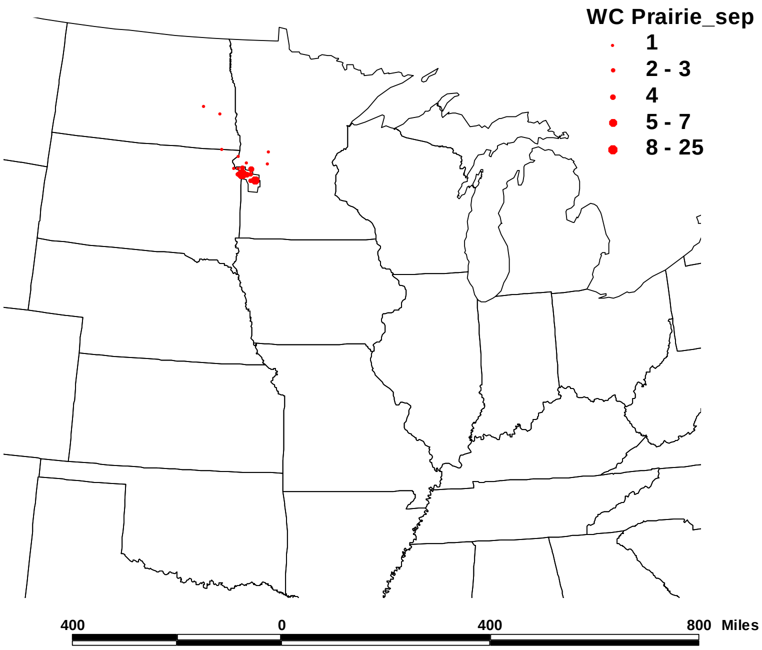


Figure 10. Distribution of direct recoveries of large Canada geese banded in the Southeast Zone–Transition region and recovered during the 1–22 September (top) and 23 September–21 February (bottom) portions of the 2002–2003 to 2007–2008 hunting seasons.



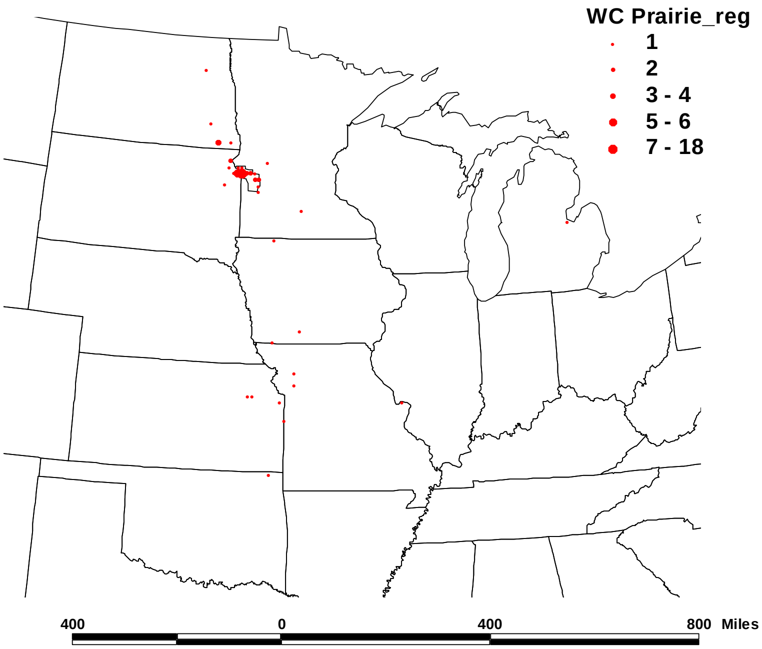


Figure 11. Distribution of direct recoveries of large Canada geese banded in the West-central Zone–Prairie region and recovered during the 1–22 September (top) and 23 September–21 February (bottom) portions of the 2002–2003 to 2007–2008 hunting seasons.

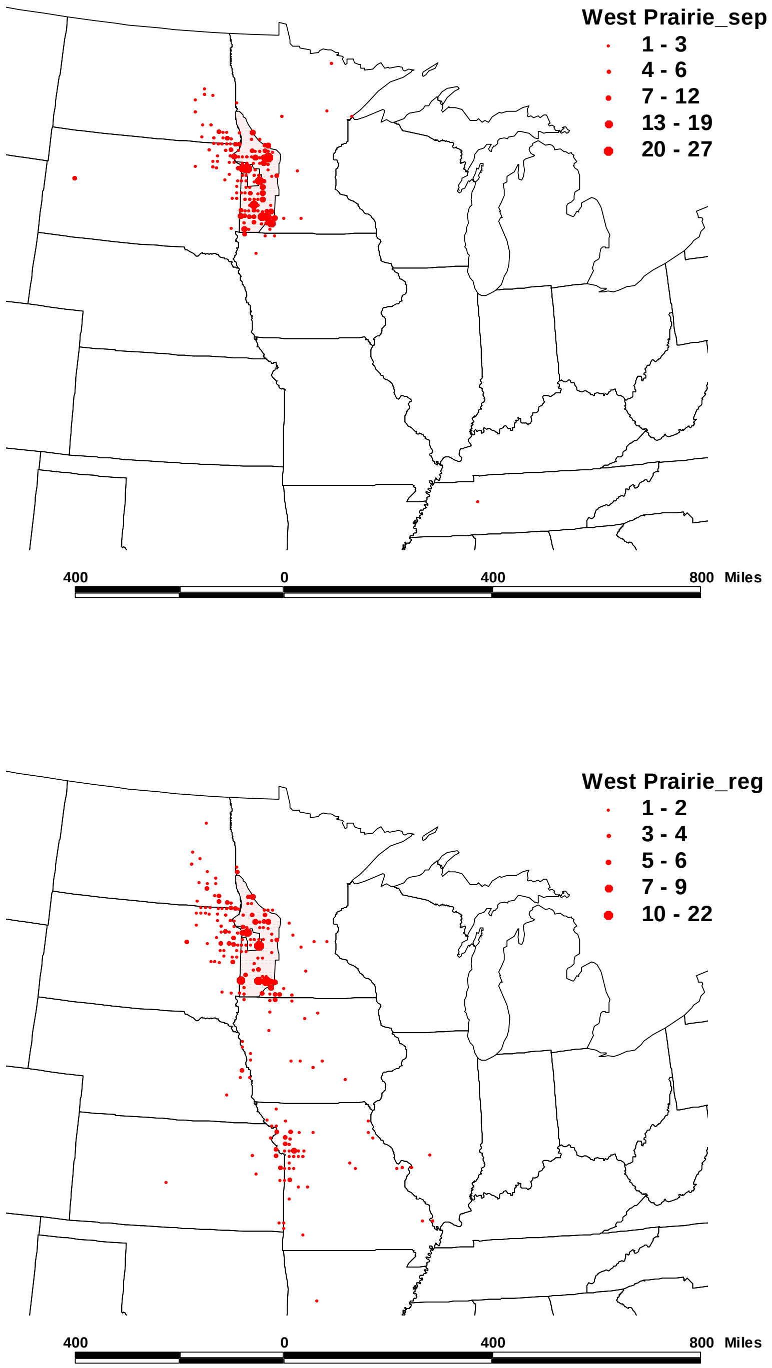


Figure 12. Distribution of direct recoveries of large Canada geese banded in the West Zone–Prairie region and recovered during the 1–22 September (top) and 23 September–21 February (bottom) portions of the 2002–2003 to 2007–2008 hunting seasons. Figure excludes 1 early season recovery in Utah, 1 regular season recovery in Manitoba, 1 regular season recovery in Mississippi, 1 regular season recovery in Nevada, and 1 regular season recovery in New Brunswick.

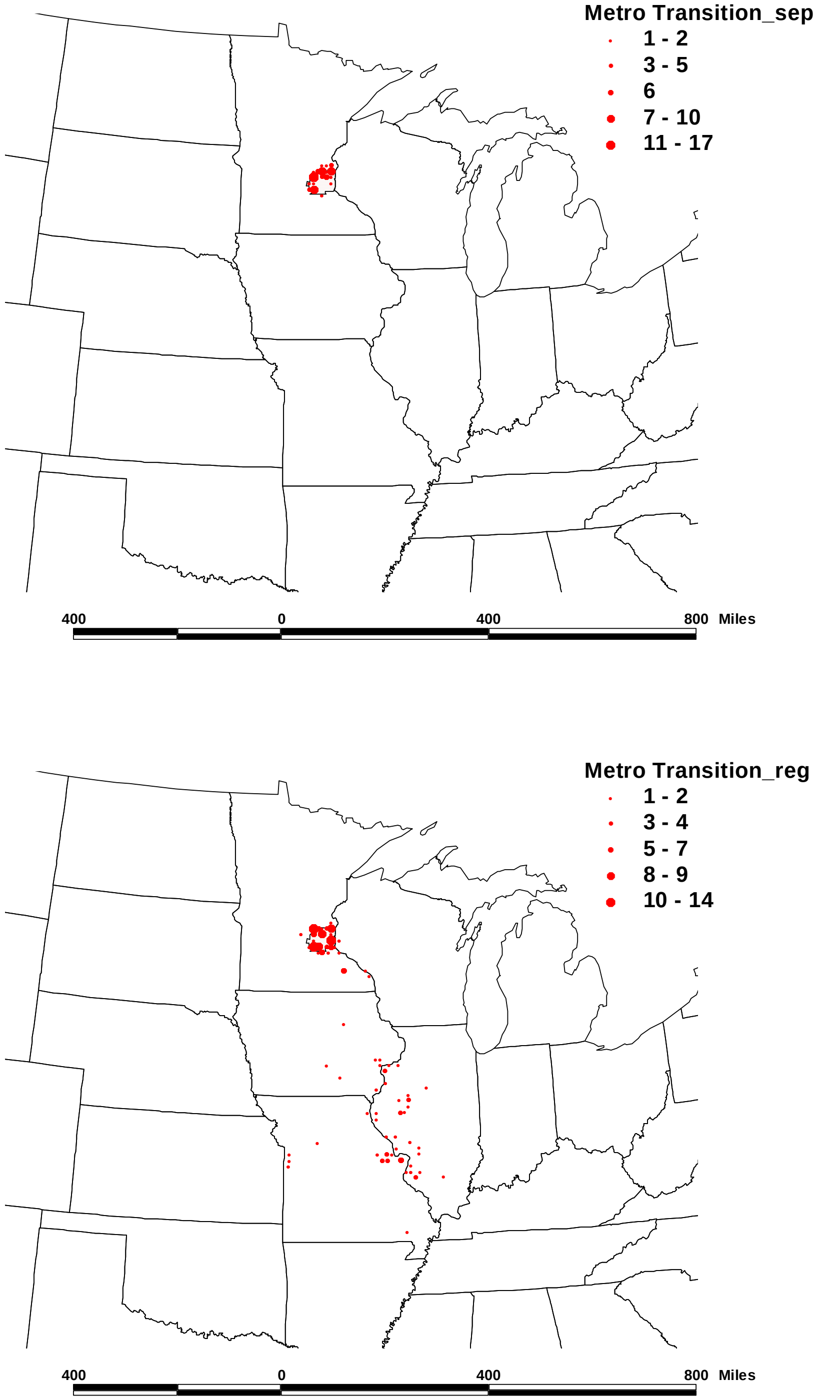


Figure 13. Distribution of direct recoveries of large Canada geese banded in the Metro Zone–Transition region and recovered during the 1–22 Sep (top) and 23 Sep–21 Feb (bottom) portions of the 2003–2004 to 2007–2008 hunting seasons. Figure excludes 1 early season recovery in Manitoba.

Appendix 1. Zone-specific large Canada geese harvest regulations in Minnesota during 2002–2007.

		Dates of hunting seasons			Number of days of large Canada goose hunting				Daily bag limit		
Hunting Zone	Year	Early	Regular	Late	Early	Regular	Late	Total	Early	Regular	Late
Northwest	2002	9/1– 9/15	9/28 – 11/6	12/7 – 12/16	15	40	10	65	2	1	5
	2003	9/6 – 9/15	9/27 – 11/5	12/6 – 12/15	10	40	10	60	2	1	5
	2004	9/4 – 9/15	9/25 – 11/3	12/4 – 12/13	12	40	10	62	2	1	5
	2005	9/3 – 9/15	10/1 – 11/9	12/10 – 12/19	13	40	10	63	5	1	5
	2006	9/2 – 9/15	9/30 – 12/8	12/9 – 12/18	14	70	10	94	5	2	5
	2007	9/1 – 9/22	9/29 – 12/7	12/8 – 12/17	22	70	10	102	5	2	5
Rest-of-state	2002	9/1 – 9/22	9/28 – 12/6	12/7 – 12/16	22	70	10	102	5	2	5
	2003	9/6 – 9/22	9/27 – 12/5	12/6 –12/15	17	70	10	97	5	2	5
	2004	9/4 – 9/22	9/25 – 11/23	12/4 – 12/13	19	60	10	89	5	2	5
	2005	9/3 – 9/22	10/1 – 12/9	12/10 – 12/19	20	70	10	100	5	2	5
	2006	9/2 – 9/22	9/30 – 12/8	12/9 – 12/18	21	70	10	101	5	2	5
	2007	9/1 – 9/22	9/29 – 12/7	12/8 – 12/17	22	70	10	102	5	2	5
Southeast	2002	9/1 – 9/22	9/28 – 12/6	12/13 – 12/22	22	70	10	102	2	2	2
	2003	9/6 – 9/22	9/27 – 12/5	12/12 – 12/21	17	70	10	97	2	2	2
	2004	9/4 – 9/22	9/25 – 11/23	12/10 – 12/19	19	60	10	89	2	2	2
	2005	9/3 – 9/22	10/1 – 12/9	12/15 – 12/24	20	70	10	100	2	2	2
	2006	9/2 – 9/22	9/30 – 12/8	12/15 – 12/24	21	70	10	101	2	2	2
	2007	9/1 – 9/22	9/29 – 12/7	12/14 – 12/23	22	70	10	102	2	2	2
West-central	2002	9/1 – 9/22	10/5 – 11/13	–	22	40	0	62	5	1	0
	2003	9/6 – 9/22	10/11 – 11/19	–	17	40	0	57	5	1	0
	2004	9/4 – 9/22	10/21 – 11/14	–	19	25	0	44	5	1	0
	2005	9/3 – 9/22	10/20 – 11/28	–	20	40	0	60	5	1	0
	2006	9/2 – 9/22	10/19 – 11/27	–	21	40	0	61	5	2	0
	2007	9/1 – 9/22	10/18 – 11/27	–	22	41	0	63	5	2	0
West	2002	9/1 – 9/22	9/28 – 11/6	12/7 – 12/16	22	40	10	72	5	1	5
	2003	9/6 – 9/22	9/27 – 11/5	12/6 – 12/15	17	40	10	67	5	1	5
	2004	9/4 – 9/22	9/25 – 10/29	12/4 – 12/13	19	35	10	64	5	1	5
	2005	9/3 – 9/22	10/1 – 11/9	12/10 – 12/19	20	40	10	70	5	1	5
	2006	9/2 – 9/22	9/30 – 11/28	12/9 – 12/18	21	70	10	101	5	2	5
	2007	9/1 – 9/22	9/29 – 11/27	12/8 – 12/17	22	60	10	92	5	2	5
Metro	2003	9/6 – 9/22	9/27 – 12/5	12/6 – 12/15	17	70	10	97	5	2	5
	2004	9/4 – 9/22	9/25 – 11/23	12/4 – 12/13	19	60	10	89	5	2	5
	2005	9/3 – 9/22	10/1 – 12/9	12/10 – 12/19	20	40	10	100	5	2	5
	2006	9/2 – 9/22	9/30 – 12/8	12/9 – 12/18	21	70	10	101	5	2	5
	2007	9/1 – 9/22	9/29 – 12/7	12/8 – 12/17	22	70	10	102	5	2	5

CHARACTERISTICS OF SHALLOW LAKES IN MINNESOTA: REGIONAL COMPARISONS AND RELATIONSHIPS AMONG LANDSCAPE SETTING, AMBIENT NUTRIENTS, AND FISH COMMUNITIES

Mark A. Hanson Shane Bowe[1], Jim Cotner[2], , Brian R. Herwig[3], Sean R. Vaughn[4], Patrick G. Welle[5], Robert W. Wright[6], Jerry A. Younk[3], and Kyle D. Zimmer[7]

SUMMARY OF FINDINGS

Minnesota's shallow lakes provide numerous direct human benefits such as clean water, hydrologic storage to limit flooding, recreational opportunities, and access to unique wild areas. They also contribute valuable ecosystem services including carbon sequestration and habitat for native species. Unfortunately, water and habitat quality of Minnesota's shallow lakes have deteriorated dramatically during the past century. Our research is aimed at identifying factors influencing key ecological features and causes for deterioration of these sites, comparing costs of possible rehabilitation strategies, and synthesizing results to provide guidance for future shallow lake management. To accomplish these objectives, we are studying approximately 136 shallow lakes in 5 ecological regions of Minnesota. Our efforts include extensive sampling of shallow lakes to identify direct and indirect causes of deterioration, evaluation of responses of 8 lakes currently undergoing rehabilitation, and an economic analysis to determine which enhancement strategies are likely to produce the greatest improvements in water quality per unit cost. Ultimately, our results will provide guidance to municipalities, state, county, local governments, and private organizations in identifying cost-effective approaches for maintaining and restoring ecological integrity of shallow lakes throughout Minnesota. Special attention will be directed towards development of regionally-specific recommendations for sustainable lake management.

BACKGROUND

Minnesota has approximately 4,000 lakes characterized by mean depth $\leq$ 5 m (15 feet) and mean surface area >16 hectares (40 acres) (Nicole Hansel-Welch, personal communication) and many thousands of smaller waters technically classified as "prairie wetlands"; the latter are functionally indistinguishable from the larger analogues (Potthoff et al. 2008). Collectively, these shallow lakes represent an international resource, providing critical waterfowl habitat and ecological benefits within Minnesota and the Mississippi Flyway. Currently, only about 40 of these lakes >16 hectares (40 acres) are formally designated for wildlife management; however, many others are focus areas for various wildlife habitat and conservation practices. Due to concerns over shallow lake water quality, seasonal duck abundance and habitat use, and hunter satisfaction, the Minnesota Department of Natural Resources (MNDNR) recently proposed a collaborative plan to Recover Ducks, Wetlands, and Shallow Lakes (http://files.dnr.state.mn.us/outdoor_activities/hunting/waterfowl/duck_plan_highlights.pdf). This plan targets restoration of 1,800 shallow lakes in Minnesota. Rehabilitation

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[1] Red Lake Department of Natural Resources, 15761 High School Drive, Red Lake, MN 56671

[2,7] Department of Ecology, Evolution, and Behavior, University of Minnesota, 100 Ecology Building, 1987 Upper Buford Circle, St. Paul, MN 55108

[3] Fisheries Research, Populations and Community Ecology Group, Minnesota Department of Natural Resources, 2114 Bemidji Avenue, Bemidji, MN 56601

[4] Division of Waters, Minnesota Department of Natural Resources, 800 Oak Savanna Lane SW, Cambridge, MN 55008

[5] Department of Economics, Bemidji State University, 1500 Birchmont Drive NE # 30, Bemidji, MN 56601

[6] Wildlife GIS/IT Unit, Minnesota Department of Natural Resources, Carlo-Avery Game Farm, 5463-C West Broadway, Forest Lake, MN 55025

[7] Department of Biology, University of St. Thomas, 2115 Summit Avenue, St. Paul, MN 55105

strategies available to shallow lake managers remain limited and often ineffective; in addition, reliable data on baseline conditions of shallow lake characteristics and regional patterns of variability are often unavailable, especially for northern areas. This means that lake and wildlife managers are frequently unsure of the current status of lakes they manage, and whether ecological characteristics of these areas may be limiting use by waterfowl and other wildlife. Generally, managers receive little useful technical guidance for management and restoration of these lakes, or for implementation of rules for managing increased development and other anthropogenic influences in these areas.

Ecological characteristics of shallow lakes, along with their suitability for ducks and other wetland wildlife species, result from integrated influences of within-site and landscape-mediated processes. Effects of key variables operate at multiple spatial scales, sometimes result from off-site influences, and no doubt vary regionally throughout the state. Ecologists have long held that prairie wetlands (including our "shallow lakes") are strongly influenced by gradients of hydrology (or hydrogeomorphic setting) and climate (especially precipitation) (Euliss et al. 2004). However, within boundaries established by hydrology and climate, as well as biological interactions, especially wetland fish communities, also exert major structuring influences on communities and characteristics of shallow lakes (Hanson et al. 2005). This is not surprising given robust improvements known to follow removal of undesirable fishes from shallow Minnesota lakes such as Christina (Hanson and Butler 1994), and smaller "prairie pothole" wetlands (Zimmer et al. 2001).

As evidenced by whole-lake fish removals such as those summarized above, shallow lake food webs often differ dramatically in response to density and community structure of associated fish populations. Fish-mediated influences on invertebrate community structure and water transparency are often pronounced (Bendell and McNicol 1987; Zimmer et al. 2000, 2001). Recent studies in Minnesota's Prairie Pothole Region (PPR) documented the strong negative influences of fathead minnows on invertebrate populations (Zimmer et al. 2000, 2001, 2002). Consequent reductions in herbivorous zooplankton (resulting from fish predation) allowed increases in phytoplankton densities and turbidity consistent with predictions of the models of Scheffer et al. (1993) and Scheffer (1998). These models propose that shallow-water ecosystems exist in 1 of 2 alternative conditions, either a clear-water, macrophyte-dominated state or a turbid-water, phytoplankton-dominated state (Scheffer et al. 1993). Minnesota PPR wetlands largely conform to a binomial distribution (clear or turbid), rather than a normal distribution of features along a theoretical continuum (Zimmer et al. 2001, Herwig et al. 2004, Zimmer et al. 2009).

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

Many lake and wetland studies have reported that landscape setting directly influences characteristics of embedded waters. For example, the watershed position sets boundaries on a variety of physical, chemical, and biological attributes of both deep lakes (Kratz et al. 1997) and prairie wetlands (Euliss et al. 2004). These lake properties include potential responses to drought, predominant groundwater interactions, water chemistry and concentrations of dissolved constituents, and biological communities. Other landscape features that have been found to influence lake water quality are wetland extent in the lake watershed (Detenbeck et al. 1993, Prepas et al. 2001) and extent of agricultural land use, the latter being correlated with higher trophic state index in associated lakes (Detenbeck et al. 1993). In many cases, off-site influences probably interact with site-level wetland features and processes so that observed community characteristics reflect simultaneous influences operating within the local context of lake nutrient status (Scheffer et al. 1993, Bayley and Prather 2003, Jackson 2003), surface area (Hobæk et al. 2002), depth (Scheffer et al. 1993), and biological properties such as abundance of macrophytes (Scheffer et. al. 1993, Paukert and Willis 2003, Zimmer et al. 2003).

Our previous work (2005-2006) confirmed that landscape characteristics can influence lake communities, interact with within-basin processes, and may be important determinants of shallow lake characteristics in Minnesota. These landscape effects are direct and indirect. For example, both presence of downstream fish sources and depth were useful for predicting fish presence/absence (Herwig et al. 2010), and landscape control on distribution of fish species limited the ability of predatory fish to control prey fish and improve water quality conditions (Friederichs et al. 2010). Extent of agriculture in upstream lake watersheds interacted with fish mass in our best models, and together these attributes were useful for predicting algal biomass in adjacent shallow lakes (Gorman et al. In preparation), and fish variables were always included in best models for predicting amphibian site occupancy and abundance in shallow lakes (Herwig et al. In preparation). In addition, results from our previous study helped elucidate mechanisms associated with important in-lake processes, such as identifying thresholds at which shallow lakes shift from turbid- to clear-water regimes, and clarifying roles of benthivorous fish in these well-known lake dynamics (Zimmer et al. 2009). Preliminary results from earlier work indicate that fish abundance and community structure exert major influences on shallow lake invertebrates, yet this relationship varies widely across ecological regions. We also are comparing relative influences of within-site and landscape-scale characteristics on shallow lake invertebrate communities. Contributions from S. Vaughn (Division of Waters, MDNR) and R. Wright (Section of Wildlife, MNDNR) provided new spatial analysis tools (e.g., delineating lake watershed boundaries, spatial analysis) that were not only critical for the recently-completed study, but will have direct application to questions and hypotheses posed in this current effort.

We plan to develop conceptual and empirical models linking landscape features, environmental influences and wetland fish assemblages, to assess influences of these factors on the community characteristics in shallow lakes, and to clarify specific influences of within-lake processes that modify ecological characteristics of shallow lakes. An overarching finding of the prior work was that regional differences often constituted the largest source of variance in characteristics of shallow Minnesota lakes. This is not unexpected given findings of others studying deeper lakes (Carpenter et al. 2007), or perceptions of staff from the MDNR Shallow Lakes Program indicating that baseline characteristics of shallow lakes differ dramatically across regions of the state (Nicole Hansel-Welch, personal communication). Regional differences not only contribute to major variability in obvious lake characteristics such as water clarity, but they probably influence extent and nature of lake responses to landscape constraints such as surface-water connectivity, as well as within-lake processes in regime responses to thresholds of phytoplankton and fish mass. For example, it is likely that combinations of increased benthivorous fish mass or decreased macrophytes will often induce regime shifts to turbid-water states in prairie lakes. However, we speculate that increased fish mass is much less likely to induce turbid-states in north-central Minnesota lakes, and turbid states may not even be possible in northern lakes where low ambient nutrient levels prevail. Additional work is needed to document extent and patterns of regional variation, and to assess how it influences key structuring mechanisms such as surface connectivity, fish community characteristics, stability of phytoplankton- and macrophyte-dominated states, and proportion of lakes in clear- versus turbid-water states.

Working Hypotheses

Our general working hypothesis is that 6 fundamental "drivers" are ultimately responsible for most of the variation in ecosystem characteristics of Minnesota's shallow lakes: climate, ambient nutrient levels, fish abundance and community type, landscape features, land use, and morphometric features of individual lakes. These 6 factors, in turn, induce strong, predictable spatial gradients in shallow lake characteristics across Minnesota. Thus, we expect shallow lakes will exhibit wide ranges of features (and responses to lake management) at a statewide scale as the influence of some drivers increase while others decrease. Additionally, inter-annual and regional variability in precipitation and temperature will have strong influences on shallow lakes. Thus, we hypothesize these drivers generate predictable spatial and temporal patterns in shallow lakes across the state of Minnesota. Overall, we believe that understanding and predicting ecosystem characteristics of shallow lakes (e.g., fish, plant and invertebrate communities, water quality, carbon cycling), along with lake responses to rehabilitation efforts, requires understanding influence of these drivers, as well as synergistic combinations of 2 or more drivers. Within-lake interactions, such as those associated with fish, have strong influences on shallow lakes (Scheffer et al. 2006, Verant et al. 2007, Potthoff et al. 2008). However, we hypothesize that strengths of these interactions also are a function of our main drivers, such that within-lake interactions also will contribute to observed spatial and temporal patterns.

We also believe it is especially important to test further hypotheses regarding stability regimes in shallow lakes. Previous work (Hanson and Butler 1994) suggests that shallow lakes in Minnesota conform to general models of alternative states developed for European lakes (Scheffer et al. 1993, Scheffer 1998), and these relationships have recently been confirmed from our prior work on Minnesota lakes (Zimmer et al. 2009). However, in Minnesota, it is likely that regime dynamics and stability thresholds will vary along regional gradients. We expect that companion models may need to be developed that extend concepts of lake regimes to include patterns of variance in invertebrate communities and other lake characteristics. Results from all study lakes will be used to estimate the magnitude of major factors responsible for deterioration of shallow lakes within the 6 study regions. Comparisons among management outcomes on 8 Intensive lakes will allow generalizations about relative usefulness of these lake rehabilitation approaches. Using a combination of data and outcomes from Extensive and Intensive lakes, our economic analysis will compare cost-effectiveness of various management approaches and should provide guidelines useful for maximizing future lake restoration and management decisions, including suggestions for cost-effective approaches in different regions of the state.

APPROACH AND METHODS

Our research has 3 broad objectives; general approaches and more specific methods for each objective are summarized below.

Objective 1: Extensive Lakes - Identify and estimate major factors responsible for deterioration of shallow lakes in 6 areas of Minnesota (hereafter Extensive Lakes).

We selected study lakes in 6 areas distributed among 5 Ecoregions of Minnesota. The following numbers of lakes were sampled during 2010 (Figure 1): Twin Cities 22, Windom 22, Alexandria 23, Itasca 22, Chippewa 15, and Red Lake 23. We sampled a total of 127 lakes for this extensive aspect of our study. We had planned to sample approximately 17 more lakes during 2010, but this was not practical due to low-water conditions, unexpected characteristics of lakes (such as alteration due to damming by beaver [*Castor canadensis*], extreme depth, or other features not noted until field visits), and because in at least 1case, landownership changed before onset of our study.

We sampled lakes to assess general ecological features and to determine whether basins exhibit characteristics of clear- or turbid-water regimes. Lake watershed characteristics associated with each study lake also will be determined. Resulting data will be used to develop models to identify combinations of variables that explain most of the variability in shallow lake characteristics, especially water quality features and lake regime status (turbid or clear). Special attention will be given to assessing influences of resident fish populations, extent of surface-water connectivity associated with study lakes, and proportion of agriculture in lake watersheds, because these are believed to be major determinants of water quality in Minnesota's shallow lakes. Resulting data will help identify and estimate magnitude of major factors responsible for deterioration of water quality and ecological characteristics in our regional subsets of study lakes.

All sites were visited during July-early August 2010. At each study lake, we sampled fish populations, abundance of submerged aquatic plants, aquatic invertebrates, water transparency, and a suite of chemical constituents in lake waters (Table 1). Water samples were collected in the field and are being tested for turbidity and concentrations of dissolved inorganic and total nitrogen, dissolved and total phosphorus, dissolved inorganic and organic carbon, and chlorophyll *a* (as a proxy for phytoplankton biomass). Additional laboratory analyses are being conducted on water column particulate matter (seston) to determine concentrations of carbon, nitrogen, and phosphorus suspended in lake water columns. Field crews collected approximately 1,260 samples of aquatic invertebrates from study lakes. Samples are currently being processed, and we expect that resulting electronic data sets will be developed by summer 2013.

We also are investigating influences of earthworms on lake productivity and soil properties in 10 small watersheds within or near Itasca State Park, Minnesota. Worms were extracted from soils to measure biomass, and soil and lake properties were characterized.

Objective 2: Intensive Lakes - evaluate and refine specific strategies for improving water quality and ecological characteristics of shallow lakes across Minnesota (hereafter Intensive Lakes).

In 2010, we met with project partners (Minnesota Ducks Unlimited, MNDNR Shallow Lakes Program staff), local MNDNR Area Wildlife Managers, and U. S. Fish and Wildlife Service staff to discuss recent shallow lake restoration (hereafter enhancement) projects in Minnesota and specific lakes for possible inclusion as case studies. Collectively, we identified 28 candidate lakes. After subsequent discussions, we selected a group of final study sites that included Nora, Sedan and Wilts lakes in the "Alexandria" study area, and Augusta, Hjermstad, Maria, Spellman and Teal lakes in the "Windom" study area. Table 2 summarizes specific enhancement activities previously implemented at each of the Intensive study lakes. Treatments and timing varied, but generally included combinations of either partial or full drawdown, rotenone additions, and in some cases, stocking of piscivorous fish (e.g., walleye).

We are evaluating responses of 8 shallow lakes (hereafter Intensive lakes) currently undergoing lake restoration treatments such as drawdowns or fish community manipulation. Ecological characteristics of Intensive lakes were sampled from June to August 2010, including all components measured in the 128 Extensive sites. Identical landscape-level analyses are being conducted on these areas to determine upland cover and surface-water connectivity in lake watersheds using Geographic Information System (GIS) analysis and interpretation of aerial photographs. Combining results and data from Intensive and Extensive lakes, we will estimate water quality improvements in response to various combinations of rehabilitation treatments, including upland restoration and within-lake-basin measures, such as fish community manipulation. Specific efforts will be directed to evaluating responses of the Intensive lakes to management efforts applied on each lake.

All Intensive lakes were sampled once monthly during June, July, and August. At each monthly visit, we sampled aquatic invertebrates, phytoplankton abundance, water transparency, and chemical constituents in lake waters (see Table 1). In July only, we also sampled fish populations and abundance of submerged aquatic plants. Samples of aquatic invertebrates collected by field crews are currently being processed and we expect that electronic data sets will be developed by summer 2013.

Objective 3: Assess cost-effectiveness of alternative lake management methods - develop region-specific guidelines useful for identifying cost-effective reclamation approaches.

An economic analysis will be conducted using empirical data from Extensive and Intensive lakes to assess costs of water quality improvements (such as cost per unit of algae reduced [µg/L chlorophyll *a*]) resulting from application of various management options being used in Minnesota. We plan to contrast costs of applying various combinations of upland vegetation restoration (e.g., conversion of agriculture to grass) and in-lake habitat enhancements (e.g., fish removal, installation of barriers) to achieve a given measure of lake water quality improvement. We expect that costs of management options will vary widely among ecological regions due to regional variability in lake characteristics, lakesheds, upland easement costs, property values, and other attributes of lakes and adjacent uplands.

Comparison of restoration costs will be informative and will help elucidate trade-offs on temporal and spatial scales. Some options may generate quick results, but may need to be repeated frequently, so that variations in long-term costs (i.e., over multiple decades) will be important to consider. Easement costs for land to be restored to vegetative buffers are known to vary across regions of the state. Cost data for the management options being studied are known to be currently available or obtainable.

Discussions among project investigators have occurred to clarify personnel and data needs, and a sub-contract with Dr. Welle (Bemidji State University) has been executed. Dr. Welle's duties for summer-fall 2011 include additional work on the conceptual framework for cost-effectiveness analysis and evaluation of preliminary data useful for relating lake rehabilitation methods to water quality improvements of shallow lakes. B. Nelson (a graduate assistant) has been hired to work with Dr. Welle; Nelson began collaborating with researchers during winter 2010-2011.

RESULTS AND DISCUSSION

Data presented here are from 5 of 6 study areas and were collected during 2010; we have not yet summarized data from our Red Lake sites, although we expect to receive summaries in the near future. Also, all data reported here are from Extensive lakes; results from Intensive lakes, and from the cost-effectiveness (economic) portions of our study, are not yet available. Finally, note that data and summaries below are preliminary; results and interpretation may change with additional data collection and analyses.

Regional patterns in total phosphorus (TP) – TP values in study lakes showed a weak increasing trend along a general north-south gradient from Itasca to Windom areas, with highest median values recorded for lakes within the Windom core. Smallest variation in TP among lakes was observed within the Itasca study area, where mean and range values were < 5 um L^{-1}. Median values in the Metro were comparable to those observed in other areas, but showed greater variability due to 1record of extremely high TP (>20 um L^{-1}).

Submerged aquatic plants, phytoplankton, and regime implications – Generally, phytoplankton was more abundant in lakes in west-central and southern study areas, especially in Windom lakes, where values for chlorophyll *a* often exceeded 100 ug L^{-1}. Other recent research on shallow Minnesota lakes suggests that sites < 22 and >31 ug L^{-1} were most often characterized as clear- or turbid-regime sites, respectively (Zimmer et al. 2009). Comparing our current lakes to those threshold values suggests that lakes in the Itasca, Chippewa, and Metro study areas were most often characterized by phytoplankton levels falling within the range expected for clear-regimes. In contrast, many Alexandria and Windom lakes showed phytoplankton levels in excess of thresholds expected for turbid-regime lakes (Figure 3). This indicates higher probability for lakes in Alexandria and Windom to show characteristics of turbid regimes.

Submerged aquatic plants are key ecosystem components of lakes, and dense plant communities are known to favor clear-water regimes in Minnesota's shallow lakes. Our preliminary data show that abundance of submerged plants varies widely from lake-to-lake and among study areas in Minnesota. Lakes supporting high mass of submerged plants showed relatively low abundance of phytoplankton (chlorophyll *a*); alternatively no lakes with chlorophyll *a* concentrations above 30 *u*g l^{-1} showed high abundance of submerged macrophytes (Figure 4). Preliminary data also suggested that these patterns vary considerably among study areas. For example, most Windom, Alexandria, and Metro-area lakes exhibited high abundance of either macrophytes or phytoplankton (but not both). In contrast, lakes in our Itasca and Chippewa study areas were dominated by macrophytes, but abundance of macrophytes and phytoplankton were much lower than in other ecoregions. This seems to suggest that primary producers in Itasca and Chippewa lakes were limited by factors such as nutrient availability.

Regional patterns in fish communities – Data gathered during 2010 indicated presence of complex fish communities in lakes of all study areas summarized thus far. We sampled no fishless lakes in our Chippewa study area, whereas a large number of fishless sites were found in our Itasca study area; a small number of fishless sites were observed in all other areas (Figure 5). Highest fish species richness was observed in Metro, Windom, and Alexandria study areas, where we sampled lakes containing up to 8, 10, and 8 species, respectively. Common carp (*Cyprinus carpio*) were less widely distributed among our lakes than we expected, with carp occurring in 23, 14, and 14 % of lakes in Windom, Metro, and Alexandria areas, respectively; carp were not collected from lakes in Itasca and Chippewa study areas. Bullheads were collected from lakes in all 5 study areas, and were found most frequently in lakes in Windom, Metro, Alexandria, and Chippewa study areas (>46% in all regions). In contrast, bullheads were collected in only 9% of Itasca lakes.

We also summarized relative abundance (mean total mass sampled) of predominant fish feeding guilds (planktivores [e.g., fathead minnows, shiners, yellow perch], benthivores [e.g., bullheads, common carp], piscivores [e.g., northern pike, walleye, largemouth bass]) for lakes within each study area (Figure 6). Generally, fish mass was roughly comparable among Windom, Alexandria, and Chippewa lakes, but showed a trend toward lower abundance in Metro and Itasca areas. Highest mass of planktivorous fishes was collected from lakes in Chippewa and Windom areas; piscivores were usually collected in lower numbers than other guilds, but were sampled at higher levels than benthivorous species in Itasca lakes. Piscivore abundance was also relatively high in Chippewa lakes. Dense populations of planktivores (shiners, yellow perch) and benthivores (bullheads) were evident in Chippewa lakes, although conventional wisdom suggests that these species are more abundant in prairie regions.

Nutrient dynamics due to earthworms - We observed a wide range of worm influences among the sampled lakes in the Itasca study area. Dissolved organic phosphorus (DOP), dissolved organic carbon (DOC), and total nitrogen (TN) in lake waters were highest at intermediate worm biomass in the surrounding soils, suggesting that uplands with intermediate biomass sites are active invasion zones where worms are releasing high amounts of nutrients. In the soils, we measured wet bulk density, percent water, total organic matter and organic phosphorus (P), and found that at intermediate worm biomass, organic P levels decreased in the soil, whereas bulk density increased. We suspect that extremes in earthworm biomass represent different degrees of invasion, with fewer anecic worms (e.g., the soil dwelling, vertical burrowing common nightcrawler *Lumbricus terrestris*) at the low extreme. At extreme earthworm abundance, more of the nutrients may have already been removed from the soil. This implies that earthworms have potential to contribute most to aquatic eutrophication as anecic worms are actively invading.

SYNTHESIS AND EXPECTED RESEARCH PRODUCTS

We will use data from 8 Intensive and 127 Extensive lakes and from characterization of associated watersheds to address our working hypotheses. Along with results from our economic analysis, we will suggest management guidelines for shallow lakes based on data and outcomes from specific ecological regions of the state. Study results will be synthesized and distributed in the form of several peer-reviewed manuscripts and a project summary, the latter to be developed specifically for shallow lake managers in Minnesota.

LITERATURE CITED

Bayley, S. E., and C. M. Prather. 2003. Do wetland lakes exhibit alternative stable states? Submersed aquatic vegetation and chlorophyll in western boreal shallow lakes. Limnology and Oceanography 48:2335-2345.

Bendell, B.E., and D. K. McNicol. 1987. Fish predation, lake acidity and the composition of aquatic insect assemblages. Hydrobiologia 150:193-202.

Braig, E. C., and D. L. Johnson. 2003. Impact of black bullhead (*Ameiurus melas*) on turbidity in a diked wetland. Hydrobiologia 490:11-21.

Carpenter, S. R., B. J. Benson, R. Biggs, J. W. Chipman, J. A. Foley, S. A. Golding, R. B. Hammer, P. C. Hanson, P. T. J. Johnson, A. M. Kamarainen, T. K. Kratz, R. C. Lathrop, K. D. McMahon, B. Povencher, J. A. Rusak, C. T. Solomon, E. H. Stanley, M. G. Turner, M. J. Vander Zanden, C. -H. Wu, and H. Yuan. 2007. Understanding regional change: a comparison of two lake districts. BioScience 57:323-335.

Detenbeck, N. E., C. A. Johnston, and G. J. Niemi. 1993. Wetland effects on lake water quality in the Minneapolis/St. Paul metropolitan area. Landscape Ecology 8:39-61.

Euliss, N. H., Jr., J. W. LaBaugh, L. H. Fredrickson, D. M. Mushet, M. K> Laubhan, G. A. Swanson, T. C. Winter, D. O. Rosenberry, and R. D. Nelson. 2004. The wetland continuum: a conceptual framework for interpreting biological studies in the prairie pothole region. Wetlands 24: 448-458.

Friederichs, S. J., K. D. Zimmer, B. R. Herwig, M. A. Hanson, and J. R. Fieberg. 2010. Total phosphorus and piscivore mass as drivers of food web characteristics in shallow lakes. Oikos: In press.

Gorman, M. W., K. D. Zimmer, B. R. Herwig, M. A. Hanson, R. G. Wright, S. R. Vaughn, and J. A. Younk. 2011. Ambient nutrient levels, fish communities, and watershed characteristics as drivers of phytoplankton abundance in shallow lakes. Ecological Applications: In preparation.

Hanson, M. A., and M. G. Butler. 1994. Responses of plankton, turbidity, and macrophytes to biomanipulation in a shallow prairie lake. Canadian Journal of Fisheries and Aquatic Sciences 51:1180-1188.

Hanson, M. A., K. D. Zimmer, M. G. Butler, B. A. Tangen,B. R. Herwig, and N. H. Euliss, Jr. 2005. Biotic interactions as determinants of ecosystem structure in prairie wetlands: an example using fish. Wetlands 25:764-775.

Herwig, B. R., M. A. Hanson, J. R. Reed, B. G. Parsons, A. J. Potthoff, M. C. Ward, K. D. Zimmer, M. G. Butler, D. W. Willis, and V. A. Snook. 2004. 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. Minnesota Department of Natural Resources, Division of Fisheries and Wildlife, Special Publication 159, St. Paul.

Herwig, B. R., L. W. Schroeder, K. D. Zimmer, M. A. Hanson, D. F. Staples, R. W. Wright, and J. A. Younk. 2011. Factors influencing amphibian site occupancy and abundance patterns in permanent wetlands in prairie and parkland landscapes of Minnesota, U.S.A.: roles of fish, basin characteristics, and adjacent landscape configuration. Biological Conservation: In preparation.

Herwig, B. R., K. D. Zimmer, M. A. Hanson, M. L. Konsti, J. A. Younk, R. W. Wright, S. R. Vaughn, and M. D. Haustein. 2010. Factors influencing fish distributions and community structure in shallow lakes in prairie and prairie-parkland regions of Minnesota, USA. Wetlands 30:609-619.

Hobaek, A., M. Manca, and T. Andersen. 2002. Factors influencing species richness in lacustrine zooplankton. Acta Oecologia 23:155-163.

Jackson, L. J. 2003. Macrophyte-dominated and turbid states of shallow lakes: evidence from Alberta lakes. Ecosystems 6:213-223.

Kratz, T. K., K. E. Webster, C. J. Bowser, J. J. Magnuson, and B. J. Benson. 1997. The influence of landscape position on lakes in northern Wisconsin. Freshwater Biology 37:209-217.

Parkos III, J. J., V. J. Santucci, Jr., and D. H. Wahl. 2003. Effects of adult common carp (*Cyprinus carpio*) on multiple trophic levels in shallow mesocosms. Canadian Journal of Fisheries and Aquatic Sciences 60:182-192.

Paukert, C. P., and D. W. Willis. 2003. Aquatic invertebrate assemblages in shallow prairie lakes: fish and environmental influences. Journal of Freshwater Ecology 18:523-536.

Potthoff, A. J., B. R. Herwig, M. A. Hanson, K. D. Zimmer, M. G. Butler, J. R. Reed, B. G. Parsons, and M. C. Ward. 2008. Cascading food-web effects of piscivore introductions in shallow lakes. Journal of Applied Ecology 45:1170-1179.

Prepas, E. E., D. Planas, J. J. Gibson, D. H. Vitt, T. D. Prowse, W. P. Dinsmore, L. A. Halsey, P. M. McEachern, S. Paquet, G. J. Scrimgeour, W. M. Tonn, C. A. Paszkowski, and K. Wolfstein. 2001. Landscape variables influencing nutrients and phytoplankton communities in Boreal Plain lakes of northern Alberta: a comparison of wetland- and upland-dominated catchments. Canadian Journal of Fisheries and Aquatic Sciences 58:1286-1299.

Rahel, F. J. 1984. Factors structuring fish assemblages along a bog lake successional gradient. Ecology 65:1276-1289.

Scheffer, M. 1998. Ecology of shallow lakes. Champman and Hall, London, United Kingdom.

Scheffer, M., S. H. Hosper, M. -L. Meijer, B. Moss, and E. Jeppesen. 1993. Alternative equilibria in shallow lakes. Trends in Ecology and Evolution 8:275-279.

Scheffer, M., G. J. vanGeest, K. Zimmer, E. Jeppesen, M. Søndergaard, M. G. Butler, M. A. Hanson, S. Declerck, L. De Meester. 2006. Small habitat size and isolation can promote species richness: second-order effects on biodiversity in shallow lakes and ponds. Oikos 112:227–231.

Spencer, C. N., and D. L. King. 1984. Role of fish in regulation of plant and animal communities in eutrophic ponds. Canadian Journal of Fisheries and Aquatic Sciences 41:1851-1855.

Tonn, W. M., and J. J. Magnuson. 1982. Patterns in the species composition and richness of fish assemblages in northern Wisconsin lakes. Ecology 63:1149-1166.

Verant, M. L., M. L. Konsti, K. D. Zimmer, and C. A. Deans. 2007. Factors influencing nitrogen and phosphotus excretion rates of fish in a shallow lake. Freshwater Biology 52:1968-1981.

Walker, R. E., and R. L. Applegate. 1976. Growth, food, and possible ecological effects of young-of-the-year walleyes in a South Dakota prairie pothole. The Progressive Fish-Culturist 38:217-220.

Zimmer, K. D., M. A. Hanson, and M. G. Butler. 2000. Factors influencing invertebrate communities in prairie wetlands: a multivariate approach. Canadian Journal of Fisheries and Aquatic Sciences 57:76-85.

Zimmer, K. D., M. A. Hanson, and M. G. Butler. 2001. Effects of fathead minnow colonization and removal on a prairie wetland ecosystem. Ecosystems 4:346-357.

Zimmer, K. D., M. A. Hanson, and M. G. Butler. 2002. Effects of fathead minnows and restoration on prairie wetland ecosystems. Freshwater Biology 47:2071-2086.

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, B. R. Herwig, and M. L. Konsti. 2009. Thresholds and stability of alternative regimes in shallow prairie-parkland lakes of central North America. Ecosystems 12:843-852.

Table 1. Summary of lake variables sampled during summer 2010. Similar data were gathered from Extensive (*n*=127) and Intensive (*n*=8) lakes, except that Intensive lakes were sampled once monthly during June, July, and August. Extensive lakes were sampled a single time during July.

Biological	**Physical**	**Chemical**
Fish abundance (gill and trap nets)	Turbidity	Total Nitrogen
Submerged aquatic plants (rake and mass methods)	Specific Conductivity	Dissolved Inorganic Nitrogen
Aquatic invertebrates (sweep nets, column samplers)		Total Phosphorus
Phytoplankton abundance (chlorophyll *a*)		Dissolved Phosphorus
		Dissolved Inorganic Carbon
		Dissolved Organic Carbon
		Seston Phosphorus
		Seston Carbon
		Seston Nitrogen

Table 2. Narrative describing shallow lake enhancement strategies implemented on selected case study lakes.

Lake	**County**	**Size (acres)**	**Enhancement strategy**	**Years post-treatment in 2010**
Nora	Pope	60	Full drawdown implemented in 2007. Began to refill in 2008, 40-50% open water by 2009. Metal half-riser structure with stop-logs functions as a fish barrier.	3
Sedan	Pope	62	Partial drawdown began in 2007, with a full drawdown occurring in 2008. Began to refill in 2009. Concrete variable crust structure with stop-logs regulates water level.	2
Wilts	Grant	55	Water levels were low in 2008 and lake is isolated, thus a decision was made to rotenone-treat the lake in fall 2008. Isolated basin.	2
Augusta	Cottonwood	499	This lake has a long history of drawdown to achieve wildlife benefits (pre-2004), but the most recent full drawdown occurred in 2008. Lake was re-flooded in 2009. Water control structure exists on lake outlet; control structures and high-velocity fish barrier installed on other adjacent waters within immediate watershed.	2
Hjermstad	Murray	60	Partial drawdown implemented in 2008, and lake was rotenone-treated under the ice during 2008-09. Fathead minnows persisted, so the lake was stocked with piscivores (walleye fry) in 2009 to attempt to suppress antecedent minnow populations. Water control via weir with stop logs; hanging finger fish barrier in place.	2
Maria	Murray	425	Full drawdown implemented from fall 2006 through fall 2007. Electric barrier was placed at lake outlet, but fish remain in the basin. Lake was rotenone-treated under the ice in February 2007. As of 2010, water levels remain low, and much of lake remains covered with very dense stands of emergent cattail. Water control via weir with stop-logs; electric fish barrier in place.	3
Spellman	Yellow Medicine	300	A managed drawdown occurred on this basin from 2006-08. 2009 was the first year with full water in the south basin. Box inlet culvert, outlet pipe, and finger-gate fish barrier in place.	2
Teal	Jackson	91	Partial drawdown implemented in 2008, and lake was rotenone-treated under the ice during winter 2008-09. Water control structure allows partial drawdown; no fish barrier in place at present.	2

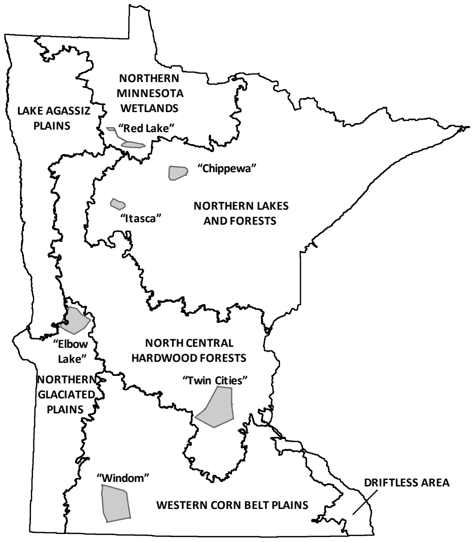

Figure 1. Map showing locations of proposed study areas (shaded gray) in relationship to Minnesota's aquatic ecoregions (thick black lines).

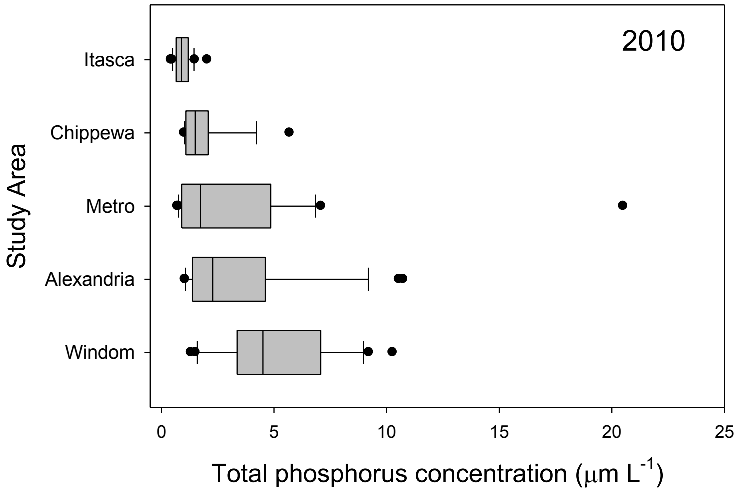

Figure 2. Box plots showing mean abundance of total phosphorus (TP) for 127 shallow lakes sampled within 5 study areas during 2010. Vertical lines within boxes depict median TP values for each study area; boxes depict 25th and 75th percentiles. Whiskers show 10th and 90th percentiles, with dots indicating more extreme values.

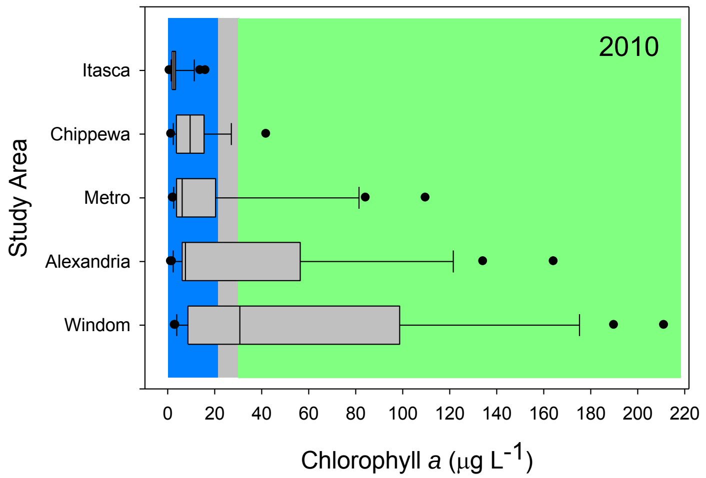

Figure 3. Box plots showing mean abundance of phytoplankton (chlorophyll *a* concentration) for 127 shallow lakes sampled within 5 study areas during 2010. Background colors depict expected chlorophyll *a* regions for clear- (blue), unstable (grey), and turbid-regimes (green) based on threshold values of Zimmer et al. (2009).

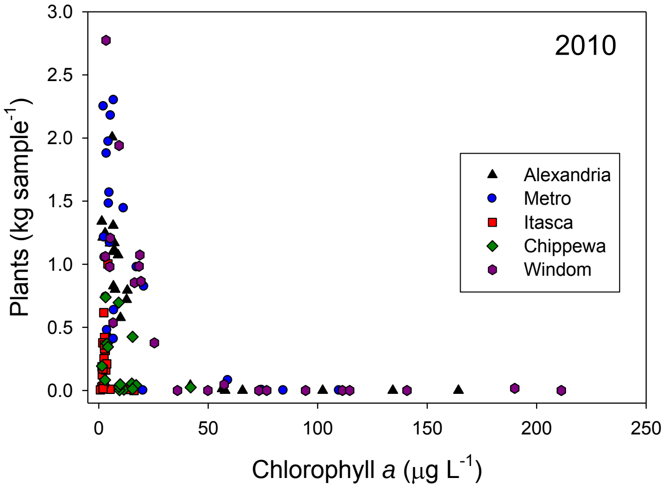

Figure 4. Abundance of phytoplankton (chlorophyll *a* concentration) and submerged macrophyte biomass for 127 shallow lakes in 5 study areas during 2010. Plant mass indicates average wet weight of plants collected on rake casts in each lake. Colors depict study area as indicated in legend.

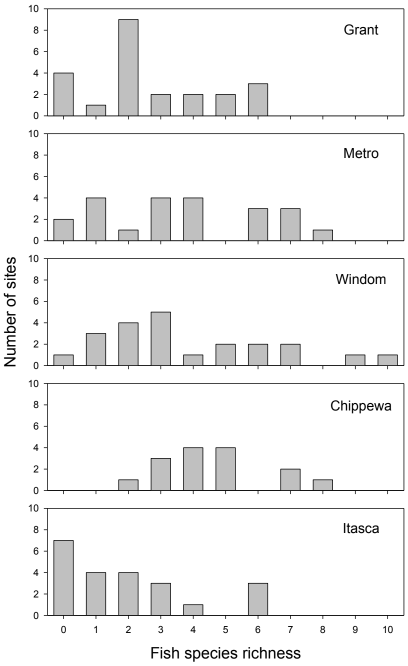

Figure 5. Fish species richness for 127 shallow lakes in 5 study areas during 2010. Height of bars on *x*-axis depicts number of lakes in which corresponding number of fish species were collected.

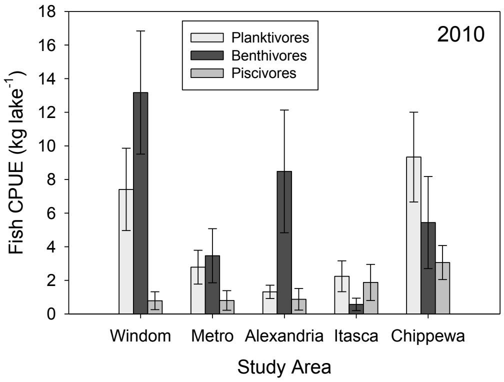

Figure 6. Summary of fish relative abundance for 127 shallow lakes in 5 study areas during 2010. Height of bars on *x*-axis depicts average weight (mass) for each of 3 major fish feeding guilds common in these lakes (planktivores, benthivores, piscivores).

Forest Wildlife Populations and Research Group

1201 East Highway 2
Grand Rapids, Minnesota 55744
(218) 327-4432

ECOLOGY AND POPULATION DYNAMICS OF BLACK BEARS IN MINNESOTA

David L. Garshelis, Karen V. Noyce, and Mark A. Ditmer[1]

SUMMARY OF FINDINGS

During April 2010–March 2011, we monitored 33 radiocollared black bears (*Ursus americanus)* at 4 study sites representing contrasting portions of the bear's geographic range in Minnesota: Voyageurs National Park (VNP, northern extreme), Chippewa National Forest (CNF; central), Camp Ripley (southern fringe), and a site at the northwestern (NW) edge of the range. This marks the 30th year of our study. Hunting has been the primary source (~80%) of mortality in all areas, even though, for the last 10 years, hunters were asked not to shoot radiocollared bears and bears cannot be legally hunted in 2 of the areas (but can be hunted when they wander outside). Reproduction was highest at the fringes of the bear range (NW and Camp Ripley), due largely to an abundance of oaks and hazelnuts in these areas. Data from Global Positioning System (GPS)-radiocollars indicated that males in the NW made significant use of cropfields (corn and sunflowers) from August–October (25% of fall locations). Females in this area rarely used crops, but instead spent much of their time in aspen woodlands and shrublands. Continuation of this work will aim to explain this sex-related disparity in habitat use and predict whether further expansion of the bear range is possible.

INTRODUCTION

Thirty years ago, the Minnesota Department of Natural Resources (MNDNR) initiated research on black bears, spurred by concerns of low population size, and commensurate with management programs to restrict the harvest and enable the population to grow. For the first 10 years, the study was limited to the Chippewa National Forest (CNF), near the center of the Minnesota bear range (Figure 1). Later, we started satellite bear projects in other study sites with different habitat conditions. Each of these began as a graduate student project, supported in part by the MNDNR. After completion of these student projects, we continued studies of bears at Camp Ripley Military Reserve, near the southern fringe of the Minnesota bear range, and in Voyageurs National Park (VNP), on the Canadian border (Figure 1).

These study sites differ enormously. The CNF is one of the most heavily hunted areas of the state, with large, easily-accessible tracts of public (national, state, and county) forests dominated by aspen (*Populus tremuloides, P. grandidentata)* of varying ages. Camp Ripley is unhunted, but bears may be killed by hunters when they range outside, which they often do in the fall, as the reserve is only 6–10 km wide. Oaks (*Quercus* sp.) are far more plentiful here than in the 2 study sites farther north, and cornfields border the reserve. VNP, being a national park, is also unhunted, but again bears may be hunted when they range outside. Soils are shallow and rocky in this area, and foods are generally less plentiful than in the other sites.

In 2007 we initiated work in a fourth study site at the northwestern edge of the Minnesota bear range (henceforth NW; Figure 1). This area differs from the other 3 areas in a number of key respects: (1) it is largely agricultural (including cropfields, like corn and sunflowers, that bears consume), (2) most of the land, including various small woodlots, is privately-owned, with some larger blocks of forest contained within MNDNR Wildlife Management Areas (WMAs) and a National Wildlife Refuge (NWR); (3) the bear range in this area appears to be expanding and bear numbers have been increasing, whereas most other parts of the bear range are stable or declining in bear numbers; and (4) hunting pressure in this area is unregulated (it is within the no-quota zone, so there is no restriction on numbers of hunting licenses, and each hunter is allowed to kill 2 bears).

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[1] Department of Fisheries, Wildlife, and Conservation Biology, University of Minnesota, St. Paul

OBJECTIVES

1. Quantify temporal and spatial variation in cub production and survival;
2. Quantify rates and sources of mortality;
3. Assess body condition indices across sites and years (not covered in this report);
4. Determine habitat use (including crop use) for bears in an agricultural fringe area; and
5. Predict range expansion of bears in northwestern Minnesota (not covered in this report).

METHODS

We attached radiocollars with breakaway and/or expandable devices to bears either when they were captured during the summer or when they were handled as yearlings in the den with their radiocollared mother. We trapped bears this year only in the NW study site, using barrel traps baited with raw bacon, and anesthetized them with ketamine-xylazine. In this area, we used principally GPS-collars, programmed to collect locations at designated intervals (varying from 6 hr to 20 min, depending on time of year). Most GPS collars used this year were “pods” (Telemetry Solutions, Concord, California) that were bolted onto normal VHF collars.

During December–March, we visited all radio-instrumented bears once or twice at their den-site. We immobilized bears in dens with an intramuscular injection of Telazol, administered with a jab stick or Dan-Inject dart gun. Bears were then removed from the den for processing, which included changing or refitting the collar, attaching a first collar on yearlings, measuring, weighing, and obtaining blood and hair samples. All GPS data were downloaded from collars of denned bears (GPS pods were either removed from the bear or downloaded at the densite). We also measured biolelectrical impedance (to calculate percent body fat) and vital rates of immobilized bears. Additionally, collaborators from the University of Minnesota (Dr. Paul Iaizzo) and Medtronic (Dr. Tim Laske) measured heart condition with a 12-lead EKG and ultrasound on a select sample of bears, and implanted (subcutaneously) a miniature heart monitoring device (developed for humans) that will record heart rate, body temperature, and activity throughout the year. Bears were returned to their dens after processing.

We assessed reproduction by observing cubs in dens of radiocollared mothers. We sexed and weighed cubs without drugging them. We evaluated cub mortality by examining dens of radiocollared mothers the following year: cubs that were not present as yearlings with their mother were presumed to have died.

During the non-denning period we monitored survival of radio-instrumented bears periodically through the summer. We listened to their radio signals, and if a pulse rate was in mortality mode (no movement of the collar in >4 hours), we tracked the collar on the ground to locate the dead animal or the shed radiocollar. During the hunting season (1 September–mid-October), hunters typically (but not always) reported collared bears that they killed.

We plotted GPS locations downloaded from collars on bears in the NW study site. We used a Geographic Information System (GIS) overlay to categorize the covertypes of GPS locations, and then grouped these into broad habitat types. We calculated percent use of these types by season for each bear, and then averaged among bears of each sex.

We quantified food production and other site characteristics in representative habitats that bears used in the NW study site. We did this in 2 ways: (1) sampling GPS locations of bears from previous years (sampling them at the same time of year as the bear’s location), and (2) sampling random plots. Within each sampling plot, we separately estimated the percent areal coverage and productivity of all principal fruiting species that bears consume. We visually rated fruit production on a 0–4 scale (0 = no fruit, 2 = average fruiting, 4 = bumper crop). We also collected and counted fruits from a sample of bushes to enable conversion of our subjective ratings to estimates of biomass. For GPS bear plots we also quantified herbaceous, shrub, and canopy layers, soil moisture, and horizontal visibility.

RESULTS AND DISCUSSION

Radiocollaring and Monitoring

Since 1981 we have handled >800 individual bears and radiocollared >500. As of April 2010, the start of the current year's work, we monitored 33 collared bears: 5 in the CNF, 8 at Camp Ripley, 3 in VNP, and 17 in the NW. We captured 7 more bears in the NW study site during June (4 males, 3 females), and collared them, 6 with GPS-collars. However, 2 GPS collars put on bears in dens failed, 2 GPS collars put on in summer failed, 5 collars were dropped, and 1 dispersing yearling bear apparently traveled beyond the search area and could not be located. We collared 3 yearling bears during March, 2011.

Mortality

Legal hunting has been the dominant cause of mortality among radiocollared bears from all study sites; over the 30 years of our study, 77% of mortalities that we observed were due, or likely due to hunting. For the 2 sites where hunting was legal (CNF, NW), 81% of bears died from this cause (Table 1). In earlier years of this study, 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 shifted to reproduction and habitat use 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 and colorful eartags so hunters could more easily see them. However, the mortality rate for collared bears has remained high even though some hunters reported avoiding them, and most of those who shot them said they saw the tags and were aware of our request to not shoot them. Ironically, on the CNF prior to asking hunters not to shoot collared bears (1981–2000), 81% died due to hunting, whereas since the request not to shoot collared bears (2001–2010), 92% died due to hunting.

This year hunters legally killed 3 collared NW bears, and we surmised that 1 other was killed by a hunter who cut off the collar and left it in the woods. Two of 5 collared bears were shot by hunters in the CNF. A number of other collars were found in the woods, but with no indication that the bears had been shot.

Although nuisance kills have been the second-most common cause of bear mortality overall, across all study areas and years (Table 1), few collared bears have been killed as nuisances in recent years (most of the nuisance-related mortalities among collared bears occurred in the 1980s). This year, 1 NW bear was killed as a nuisance; however this was an unusual case. This bear did not come from this area: it was orphaned as a cub, raised for a few months in captivity (at a rehab facility), and released in late fall with 3 other orphaned cubs in the Thief Lake WMA. In the spring, it traveled 26 km west and entered a small town, where it could not be scared away, so was shot. Another of this group of released orphans was a nuisance elsewhere, but not killed; it subsequently dropped its collar and was shot by a hunter. This was our first attempt to release orphaned, captive-raised cubs in the NW study site. We have had better success doing so within the CNF study site (1 was released in the CNF in late fall, 2010).

One other mortality of a collared bear occurred this year: a bear from Camp Ripley denned in a cornfield outside the Camp and was run over by the farmer's combine; its yearling offspring also died.

Reproduction

We visited 8 dens of females with cubs during March, 2011 (including 1 outside our 4 study sites). On the 4 study sites, since 1982, we have checked 251 litters with 644 cubs (2.6 cubs/litter), of which 52% were male (Tables 2–5). Overall, first-year mortality averaged 21%, and mortality of male cubs (26%) exceeded that of females (15%). However, there appears to have been a change in these rates through time: during the most recent 5 years (2007–2011), litters have been slightly larger (2.7 cubs/litter) with increased numbers of females (50.9% male), but also increased female cub mortality (26%, versus only 19% for males). The timing and causes of cub mortality are unknown.

Reproductive rates were highest in the 2 study areas at the periphery of the bear range (NW, Camp Ripley) and lowest in VNP (Figure 2). The reproductive rate (cubs/female 4+ years old) combines litter size, litter frequency, and age of first reproduction into a single parameter. Reproductive rate was higher for 7+-year-old bears than 4–6-year-old bears because many bears in this younger age group either had not yet reproduced or had their first litter, which tended to be smaller. Regional variation in reproductive rates of older bears relates to effects of food supply on litter size and litter interval.

Habitat Use of NW Bears

During spring and summer, both male and female GPS-collared bears in the NW study site most frequently used forests dominated by aspen (Figure 3); they spent, on average 33% to nearly 50% of their time there, whereas this forest type comprised only 13% of the area of the region, in a patchy distribution (e.g., small WMAs, Agassiz NWR, and private lands). Lowland shrub was the second-most used habitat (~25% frequency of use) during these seasons. In fall, females continued to use primarily these same 2 habitat types; surprisingly, they used oak forests (primarily bur oak [*Q. macrocarpa*], which comprised ~2% of the landscape) only 8% of the time. Also, females rarely used agricultural crops (Figures 3–4). Their high reproductive rate (Figure 2), which should be reflective of high food availability, especially in fall, is thus somewhat of an enigma; however, there appeared to be an abundance of hazelnuts (*Corylus americana, C. cornuta*) and dogwood berries (mainly *Cornus racemosa* and *C. sericea*) in these habitats (quantification of the food abundance data by habitat is ongoing). Males, in contrast, were frequently found in croplands during fall, on average spending about 25% of their time there. Although about half the landscape was comprised of agricultural crops, the crops that bears consumed (mainly corn and sunflowers) represented a small areal coverage, equivalent to oak forests (2%); male bears spent more time feeding on crops than on acorns, even in years when acorns were plentiful. Both sexes were also periodically found in soybean and wheat fields (Figure 4); we are uncertain whether they fed on wheat, but have observed them occasionally feeding on soybeans.

We highlight an interesting example of a male bear that fed in a sunflower field because it exemplifies several notable issues with management implications. This adult male, collared since 2008, has lived principally in Agassiz NWR. In 2010 he rarely moved outside the refuge throughout the year. In fall he found a sunflower field just outside the western edge of the refuge (Figure 5). He first entered this field on August 1, but did not use it on a daily basis until August 13, at which time he used it every day through September 24, almost entirely at night (Figure 6). He then left, but returned again on September 30 and used the field daily until October 6, at which point he traveled 10 km to immediately den (8 October) in a wilderness area in central Agassiz (he has denned in this vicinity each year). Clearly this bear was entirely reliant on anthropogenic food in the fall, feeding in the cropfield daily for 7 weeks; notably, there are very few oaks in Agassiz NWR, so this sunflower field was probably the closest concentrated food source available. We visited his den, an excavation into a peat “island”, on March 10, at which time he weighed 185 kg (406 lbs).

FUTURE DIRECTIONS

We plan to continue monitoring bears on these 4 study sites, although sample sizes have been greatly diminished by the exceedingly high harvest of collared bears in the past few years. We will continue to collect GPS-collar data in the NW study site. In addition to gaining information from radiocollars, we have been and will continue to interview farmers to collect additional data on bear use of crops. This will yield a historical perspective on crop use, and provide insights into specific varieties of corn and sunflowers used by bears. Moreover, we have obtained a collection of hair samples from hunter-killed bears in the NW for stable isotope analysis to ascertain the importance of corn in the diet, relative to gender and location. Ultimately we aim to create a habitat suitability map and thereby predict how far the bear population is likely to expand in this part of the state.

ACKNOWLEDGMENTS

We thank the collaborators in this study: B. Dirks at Camp Ripley, Dr. P. Iaizzo at the University of Minnesota, and Dr. T. Laske at Medtronic, Inc. A. Tri and M. Lyons assisted with fieldwork in the NW.

Table 1. Causes of mortality of radiocollared black bears ≥1 year old from the Chippewa National Forest (CNF), Camp Ripley, Voyageurs National Park (VNP), and northwestern (NW) Minnesota, 1981–2010. Bears did not necessarily die in the area where they usually lived (e.g., hunting was not permitted within Camp Ripley or VNP, but bears were killed by hunters when they traveled outside these areas).

	CNF	Camp Ripley	VNP	NW	All combined
Shot by hunter	223	11	15	10	259
Likely shot by hunter[a]	8	1	0	3	12
Shot as nuisance	22	2	1	1	26
Vehicle collision	12	8	1	1	22
Other human-caused death	9	1	0	0	10
Natural mortality	7	3	4	0	14
Died from unknown causes	4	2	0	3	9
Total deaths	285	28	21	18	352

[a] Lost track of during the hunting season, or collar seemingly removed by a hunter.

Table 2. Black bear cubs examined in dens of radiocollared mothers in or near the Chippewa National Forest during March, 1982–2011. High hunting mortality of radiocollared bears has severely reduced the sample size in recent years.

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%
1998	2	6	3.0	67%	0%
1999	7	15	2.1	47%	9%
2000	2	6	3.0	50%	17%
2001	5	17	3.4	76%	15%
2002	0	0	—	—	—
2003	4	9	2.3	22%	0%
2004	5	13	2.6	46%	33%
2005	6	18	3.0	33%	28%
2006	2	6	3.0	83%	33%
2007	2	6	3.0	67%	17%
2008	1	3	3.0	100%	33%
2009	1	3	3.0	33%	33%
2010	1	4	4.0	100%	50%
2011	1	4	4.0	25%	
Overall	176	463	2.6	52%	19%

[a] Cubs that were absent from their mother's den as yearlings were considered dead. Blanks indicate no cubs were born to collared females.

Table 3. Black bear cubs examined in dens in or near Camp Ripley Military Reserve during March, 1992–2011.

Year	Litters checked	No. of cubs	Mean cubs/litter	% Male cubs	Mortality after 1 yr[a]
1992	1	3	3.0	67%	0%
1993	3	7	2.3	57%	43%
1994	1	1	1.0	100%	—
1995	1	2	2.0	50%	0%
1996	0	0	—	—	—
1997	1	3	3.0	100%	33%
1998	0	0	—	—	—
1999	2	5	2.5	60%	20%
2000	1	2	2.0	0%	0%
2001	1	3	3.0	0%	33%
2002	0	0	—	—	—
2003	3	8	2.7	63%	33%
2004	1	2	2.0	50%	—
2005	3	6	2.0	33%	33%
2006	2	5	2.5	60%	—
2007	3	7	2.3	43%	0%
2008	2	5	2.5	60%	0%
2009	3	7	2.3	29%	29%
2010	2	4	2.0	75%	25%
2011	3	8	2.7	50%	
Overall	33	78	2.4	51%	22%

[a] Cubs that were absent from their mother's den as yearlings were considered dead. Blanks indicate no cubs were born to collared females or collared mothers with cubs died before the subsequent den visit to assess cub survival.

Table 4. Black bear cubs examined in dens in Voyageurs National Park during March, 1999–2011. All adult collared females were killed by hunters in fall 2007, so no reproductive data were obtained during 2008–2009.

Year	Litters checked	No. of cubs	Mean cubs/litter	% Male cubs	Mortality after 1 yr[a]
1999	5	8	1.6	63%	20%
2000	2	5	2.5	60%	80%
2001	3	4	1.3	50%	75%
2002	0		—	—	—
2003	5	13	2.6	54%	8%
2004	0		—	—	—
2005	5	13	2.6	46%	20%
2006	1	2	2.0	50%	0%
2007	3	9	3.0	44%	—
2008	0				
2009	0				
2010	1	2	2.0	50%	0%
2011	1	2	2.0	0%	
Overall	26	58	2.2	50%	27%

[a] Cubs that were absent from their mother's den as yearlings were considered dead. Blanks indicate no cub mortality data, because no cubs were born to collared females.

Table 5. Black bear cubs examined in dens in northwestern Minnesota during March, 2007–2011.

Year	Litters checked	No. of cubs	Mean cubs/litter	% Male cubs	Mortality after 1 yr[a]
2007	2	6	3.0	33%	100%[b]
2008	5	15	3.0	67%	22%
2009	1	3	3.0	33%	33%
2010	6	17	2.8	41%	13%
2011	2	4	2.0	75%	
Overall	16	45	2.8	50%	20%[c]

[a] Cubs that were absent from their mother's den as yearlings were considered dead.

[b] Only one 5-cub litter was monitored, and all the cubs died (mother produced a litter of 4 cubs the next year).

[c] Excludes the total loss of the single 5-cub litter (which was not within the designated study area).

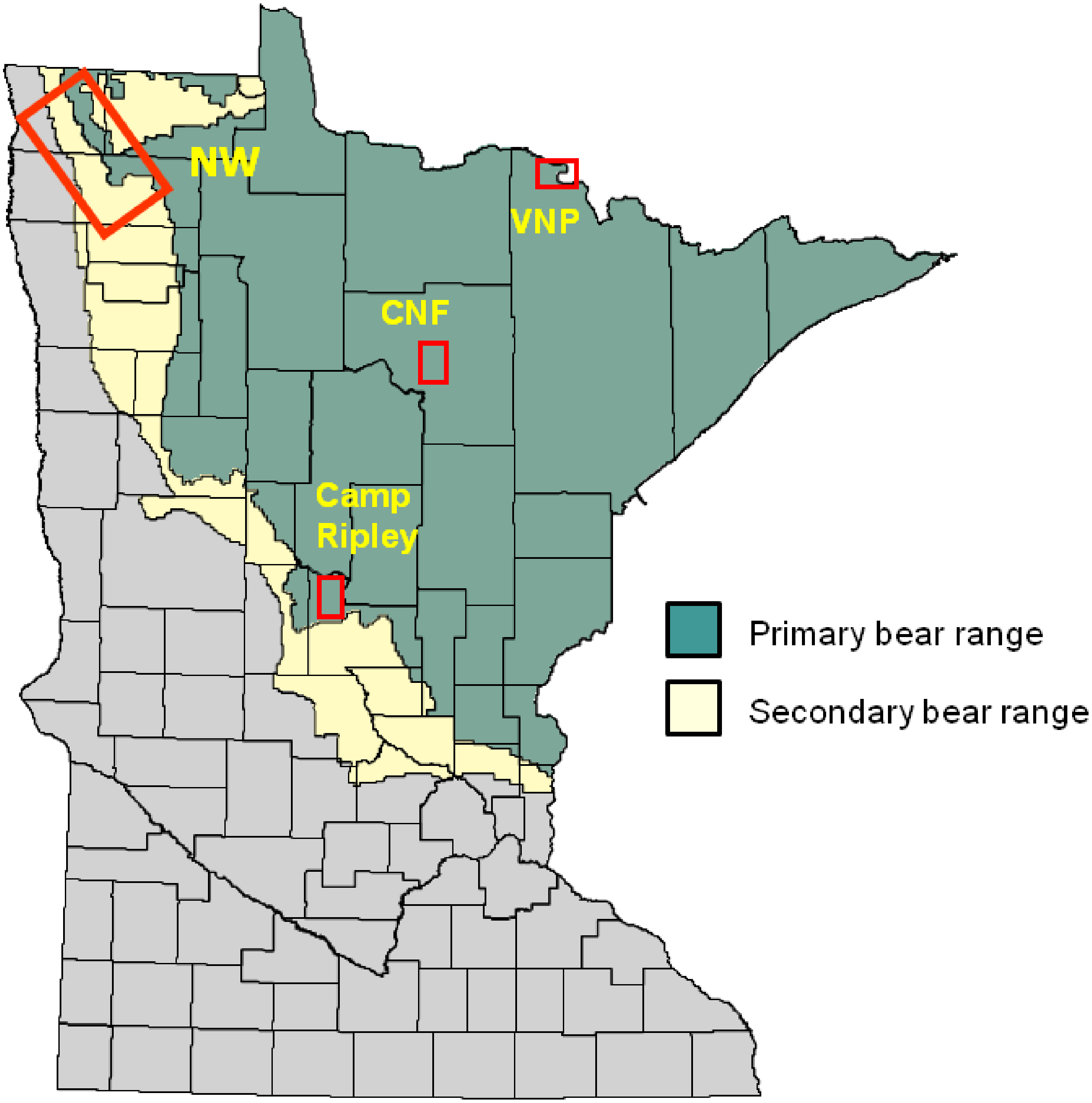


Figure 1. Location of 4 study sites within Minnesota's black bear range: CNF (Chippewa National Forest, central bear range; 1981–2011); VNP (Voyageurs National Park, northern fringe of range; 1997–2011); Camp Ripley Military Reserve (near southern edge of range; 1991–2011); and NW (northwestern fringe of range; 2007–2011).

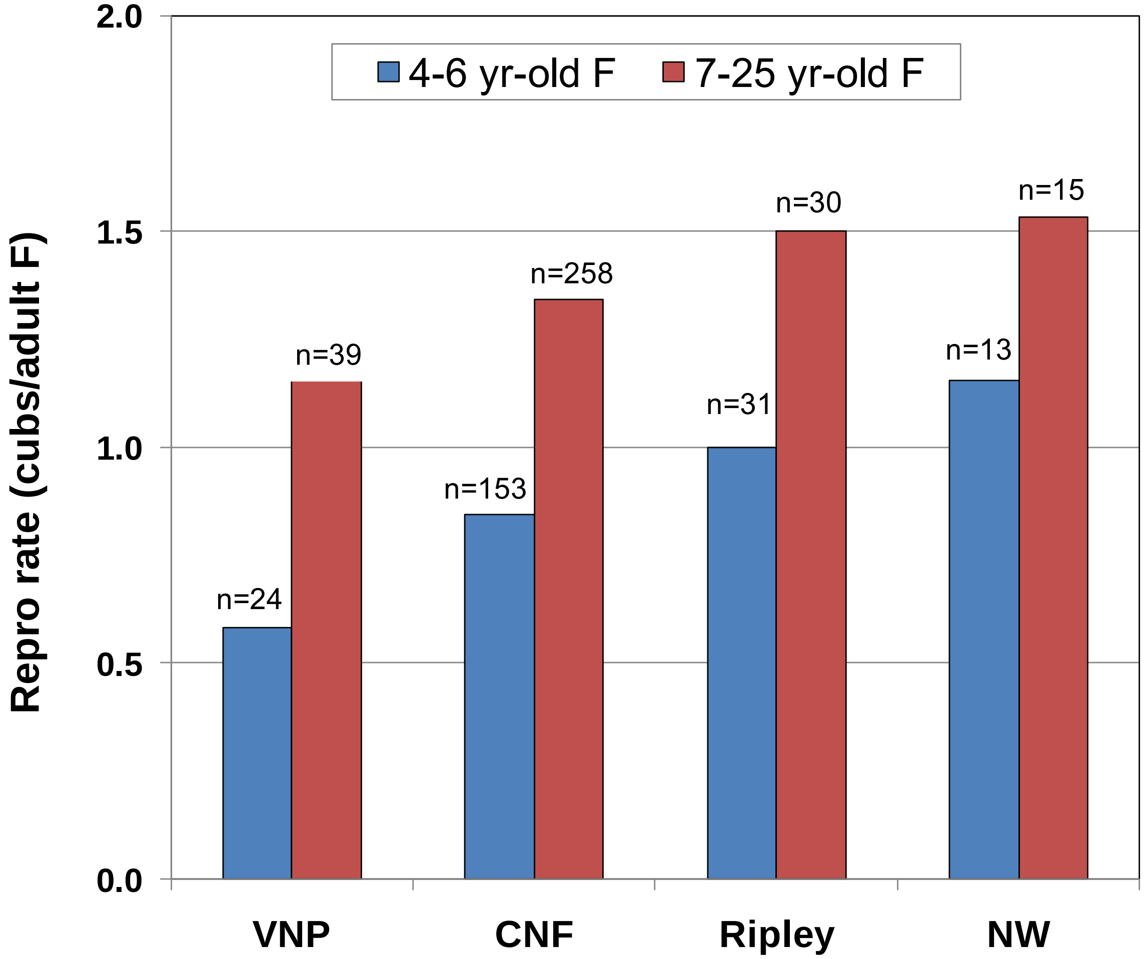
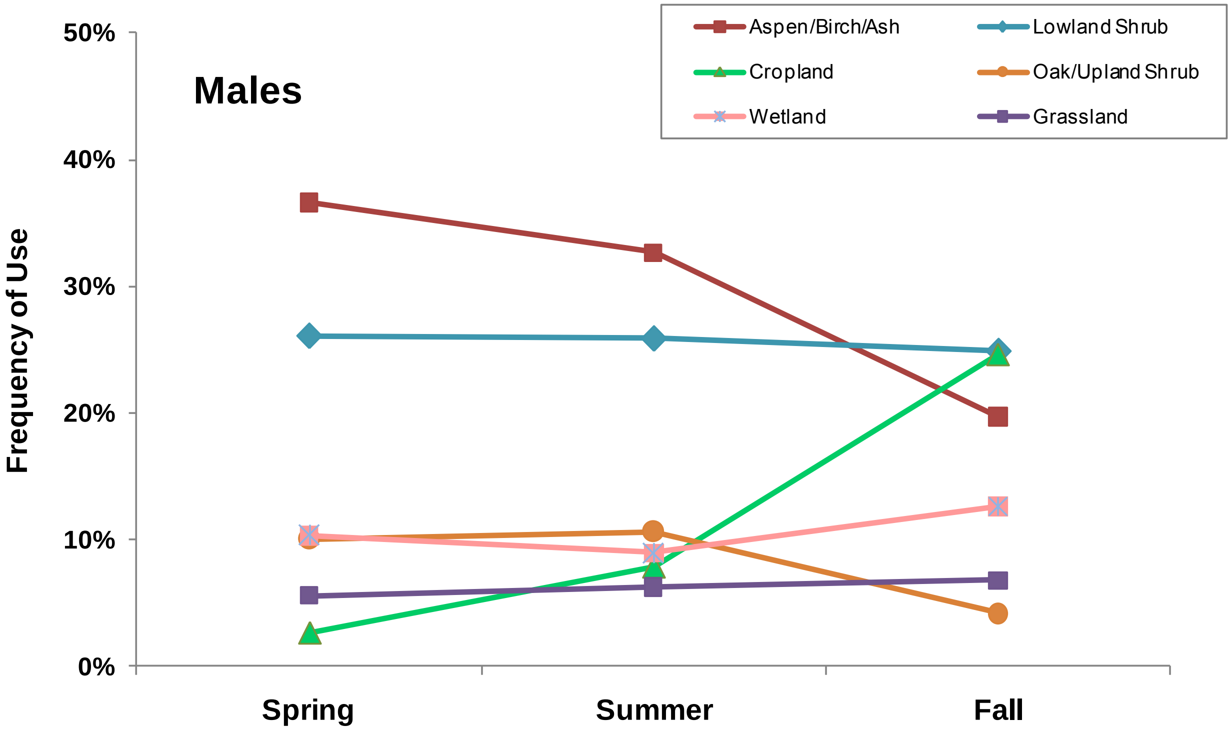


Figure 2. Reproductive rates of radiocollared black bears within 4 study sites (see Figure 1). Sample sizes refer to the number of female bear-years of monitoring in each area for each age group. Data include only litters that survived 1 year (even if some cubs in the litter died).



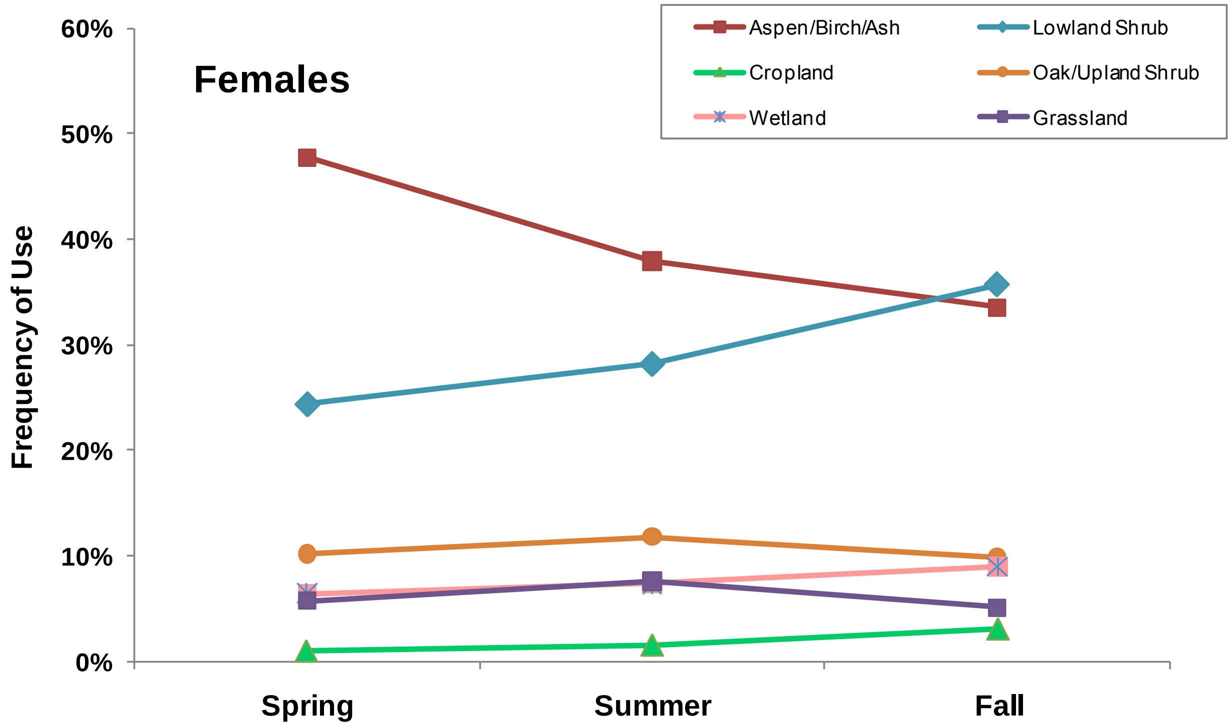


Figure 3. Seasonal habitat use of GPS-collared black bears in northwestern Minnesota, 2007–2010. Values for each season do not sum to 100%, because bears also infrequently used habitat types other than those graphed. Seasons were defined as follows: spring (April [den exit]–mid-June); summer (mid-June–August); fall (September–denning).

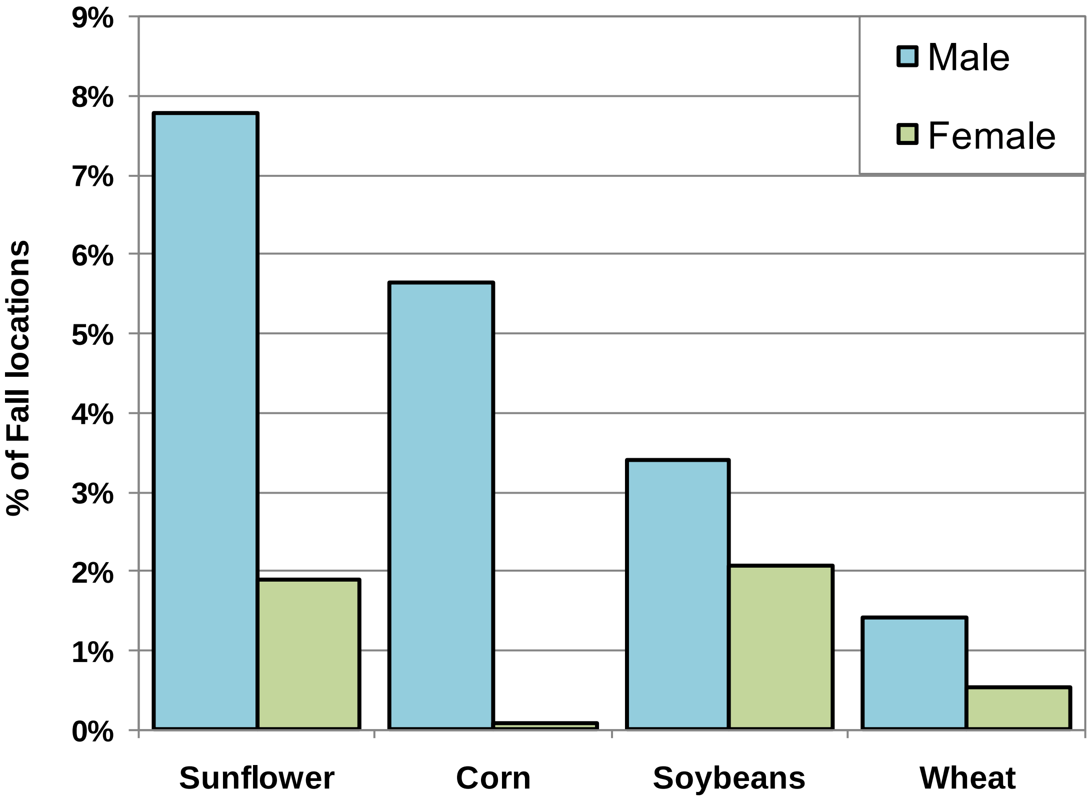


Figure 4. Use of crops by GPS-collared black bears in northwestern Minnesota during fall 2010. Values for each sex do not sum to the total cropland use shown in Figure 3, because the data graphed here are only for 1 year.

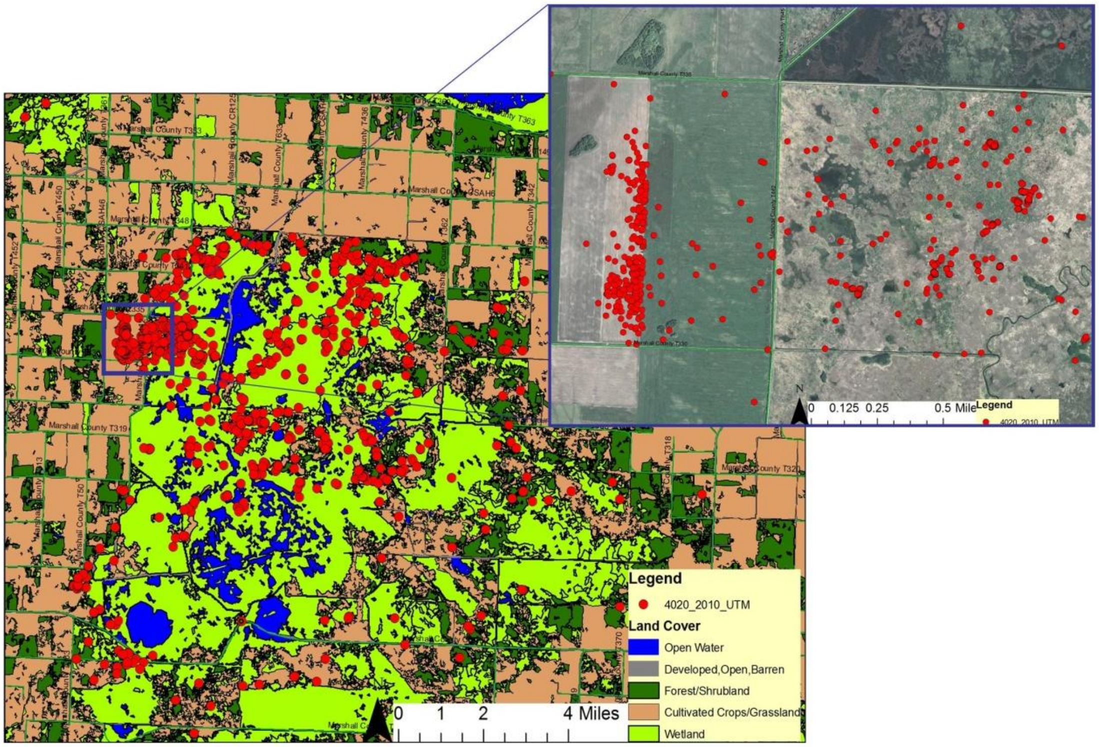


Figure 5. Locations of a GPS-collared male black bear in and near Agassiz NWR (most of the light-green colored area is inside the refuge) during 2010. The enlarged block (satellite photo) shows an area of concentrated use in a sunflower field during August–September, before he denned in central Agassiz in early October.

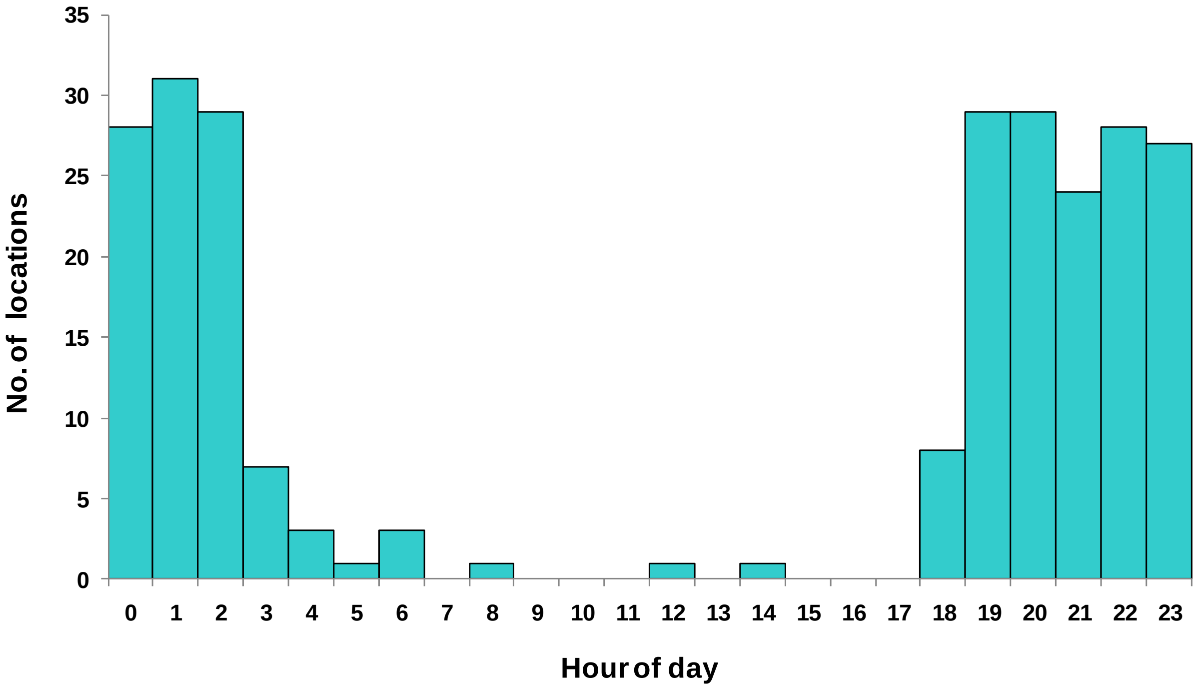


Figure 6. Hourly use of sunflower field by the male black bear shown in Figure 5.

MOOSE POPULATION DYNAMICS IN NORTHEASTERN MINNESOTA

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

SUMMARY OF FINDINGS

We captured and radiocollared a total of 150 adult moose (55 adult males and 95 adult females) between 2002 and 2008. As of 1 April 2011, 114 collared moose (50 adult males and 64 adult females) have died. Annual mortality rates varied among years, and generally were higher than found elsewhere in North America. Estimates of fertility for this population were also low compared with other North American moose populations. Data analyses from this research are progressing and 3 manuscripts are published, 1 manuscript is in press, and 2 other manuscripts have been submitted to peer-reviewed journals.

INTRODUCTION

Moose (*Alces alces*) formerly occurred throughout much of the forested zone of northern Minnesota. Today they are restricted to the northeastern-most counties, including all of Lake and Cook counties, and most of northern St. Louis County. We initiated a research project in 2002 to better understand the dynamics of this population. Fieldwork, including aerial mortality checks and necropsy of dead animals, continued through 1 April 2011. We are in the process of analyzing data and preparing manuscripts. The following report will discuss preliminary findings.

The project was a partnership between the Minnesota Department of Natural Resources (MNDNR), Fond du Lac Band of Lake Superior Chippewa, 1854 Treaty Authority and U. S. Geological Survey. A second phase of the research project was initiated in February 2008 with funding secured by the Fond du Lac Band. The MNDNR and 1854 Treaty Authority provide in-kind support and limited funding for this second phase of research.

METHODS

We captured a total of 150 moose in southern Lake County and southwestern Cook County between 2002 and 2008, attached radiocollars, and collected blood, hair, fecal and tooth samples. See Lenarz et al. (2009) for greater detail on the study area and research methods. We monitored a sample of up to 78 radiocollared moose weekly to determine when mortality occurred. We calculated annual non-hunting mortality rates (1 – survival) using the Kaplan-Meier procedure (Kaplan and Meier 1958) modified for staggered-entry (Pollock et al. 1989) and censored all moose killed by hunters, that died from capture mortality, that had emigrated from the study area, or experienced apparent transmitter failure. We used a Cox Proportional Hazard (CPH) model (Cox 1972, SAS PROC PHREG, SAS Institute 2008) to test for a difference in annual survival between sexes. Beginning in 2004, we used helicopter surveys in late May–early June (MJ) to estimate fertility of radiocollared females and a survey the following year in late April–early May (AM) to estimate survival of calves born during the previous spring.

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[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, Northern Prairie Wildlife Research Center, Jamestown, North Dakota 58401

RESULTS AND DISCUSSION

As of 1 April 2011, 114 collared moose (50 adult males and 64 adult females) have died. In addition, 1 moose slipped its collar, 1 moose moved out of the study area, and we lost contact (apparent transmitter failure) with 2 moose (1 of the moose with a dead collar was re-captured in January 2011 as part of a new research project). Moose that died within 2 weeks of capture (6) were designated as capture mortality. Hunters killed 17 moose, 2 were poached, and 12 were killed in collisions with vehicles (cars, trucks, or trains). The remaining mortality (77) was considered to be non-anthropogenic, and causes included wolf predation (10), bacterial meningitis (1), and unknown (66).

The unknown mortality appeared to be largely non-traumatic. In 51% of the cases, the intact carcass was found with only minor scavenging by small mammals or birds. Wolves (*Canis lupus*) and black bears (*Ursus americanus*) were the primary scavengers in 34% of the cases. We were unwilling to attribute predation as the cause of death in these cases, because there was little evidence that a struggle had preceded death. In 15% of the cases, we were unable to examine the carcasses or only found a collar with tooth-marks.

Annual non-hunting mortality rates (1 June–31 May) for adult moose averaged 20% for males (SE = 5, 0–40%, n = 8) and 21% for females (SE = 3, 5–30%, n = 8; Table 1). Sex did not contribute to the prediction of survival (χ^2 = 0.001, P = 0.98), which implies that there was no difference in survival rates (non-hunting) between adult male and female moose. Non-hunting mortality was substantially higher than documented for populations outside of Minnesota (generally 8 to 12%; Peterson 1977, Mytton and Keith 1981, Bangs 1989, Larsen et al. 1989, Ballard 1991, Kufeld and Bowden 1996, Bertram and Vivion 2002,) and similar to that observed for adult moose in northwestern Minnesota (21%; Murray et al. 2006).

Serum samples from 91 radiocollared adult female moose were collected between 2002 and 2008 and analyzed by radioimmunoassay for concentrations of progesterone. Using a pregnancy threshold of 2.0 ng/ml progesterone, annual pregnancy rate varied from 55 to 100% ($\bar{x}$ = 80%, SE = 8, n = 5). Boer (1992), in his review of moose reproduction in North America found that adult pregnancy rates across North America averaged 84%. Although the pregnancy rates of yearlings tend to be lower than for adult moose (Schwartz 1997), our sample included only 1 yearling. Our estimates may be biased low, because 4 cows that tested negative in 2003 (55% pregnancy rate) were subsequently observed with a calf.

Between 2004 and 2010, 222 radiocollared adult females gave birth to a minimum of 196 calves (115 singles, 39 twins, and 1 set of triplets; M. W. Schrage, Fond du Lac Resources Management Division, unpublished data). The annual ratio of calves:radiocollared females ranged from 0.53 to 1.13 ($\bar{x}$ = 0.87, SE = 0.07, n = 7). These estimates were biased low, because in 6 of 7 years, radiocollared females not observed with calves during the late MJ survey were subsequently observed to be accompanied by a single calf ($\bar{x}$ = 3, SE = 0.6, n = 7). It is also possible that post-natal mortality occurred prior to the MJ survey. Nonetheless, these estimates are low compared with other locations in North America. Boer (1992), for example, reported estimates ranging from 0.88 to 1.24 calves/adult female in moose populations above and below K-carrying capacity, respectively.

During the past 3 years, 6 manuscripts discussing the results of this research have been prepared for publication. Most recently, a paper entitled "Spending degrees of freedom in a poor economy: a case study of building a sightability model for moose in northeastern Minnesota" was accepted for publication in the *Journal of Wildlife Management*. Two additional manuscripts evaluating habitat utilization by northeastern moose have been submitted to the journal Alces. Finally, data from our research served as the basis for a chapter in a Master of Science thesis on characteristics of post-partum areas for moose in northeastern Minnesota.

LITERATURE CITED

Ballard, W. B., J. S. Whitman, and D. J. Reed. 1991. Population dynamics of moose in south-central Alaska. Wildlife Monograph 114.

Bangs, E. E., T. N. Bailey, and M. F. Portner. 1989. Survival rates of adult female moose on the Kenai Peninsula, Alaska. Journal of Wildlife Management 53:557-563.

Bertram, M. R., and M. T. Vivion. 2002. Moose mortality in eastern interior Alaska. Journal of Wildlife Management 66:747-756.

Boer, A. H. 1992. Fecundity of North American moose (*Alces alces*): a review. Alces (Supplement) 1:1-10.

Cox, D. R. 1972. Regression models and life tables. Journal of the Royal Statistical Society, Series B 20:187-220.

Kaplan, E. L., and O. Meier. 1958. Non-parametric estimation from incomplete observations. Journal of the American Statistical Association 53:457-481.

Kufeld, R. C., and D. C. Bowden. 1996. Survival rates of Shiras moose (*Alces alces shirasi*) in Colorado. Alces 32:9-13.

Larsen, D. G., D. A. Gauthier, and R. L. Markel. 1989. Cause and rate of moose mortality in the southwest Yukon. Journal of Wildlife Management 53:548-557.

Lenarz, M. S., M. E. Nelson, M. W. Schrage, and A. J. Edwards. 2009. Temperature mediated survival in northeastern Minnesota moose. Journal of Wildlife Management 73:503-510.

Murray, D. L., E. W. Cox, W. B. Ballard, H. A. Witlaw, M. S. Lenarz, T. W. Custer, T. Barnett, and T. K. Fuller. 2006. Pathogens, nutritional deficiency, and climate influences on a declining moose population. Wildlife Monographs 166.

Mytton, W. R., and L. B. Keith. 1981. Dynamics of moose populations near Rochester, Alberta, 1975-1978. Canadian Field-Naturalist 95:39-49.

Peterson, R. O. 1977. Wolf ecology and prey relationships on Isle Royale. National Park Service Scientific Monograph. 88. Washington, D.C.

Pollock, K. H., S. R. Winterstein, C. M. Bunck, and P.D. Curtis. 1989. Survival analysis in telemetry studies: the staggered entry design. Journal of Wildlife Management 53:7-15.

Schwartz, C. C. 1997. Reproduction, natality and growth. Pages 141-172 *in* A. W. Franzmann and C. C. Schwartz, eds. Ecology and management of North American moose. Smithsonian Institution Press, Washington, D. C.

SAS Institute. 2008. Version 9.1, SAS Institute, Cary, North Carolina.

Table 1. Annual adult mortality of moose in northeastern Minnesota, USA, 2002–2009. Estimates censored for hunting, capture mortality, and apparent transmitter failure.

Year[1]	Male	Female	Combined
2002	7% (25)[2]	30% (29)	23% (54)
2003	25% (21)	20% (34)	21% (55)
2004	8% (32)	5% (42)	6% (74)
2005	24% (21)	29% (30)	26% (51)
2006	40% (10)	27% (22)	31% (32)
2007	20% (8)	19% (49)	18% (57)
2008	0% (7)	21% (38)	16% (45)
2009	33% (4)	13% (33)	16% (37)
Mean	18%	21%	20%

[1] Period: 1 June–31 May.

[2] Sample size as of 31 May.

SURVIVAL AND CAUSES OF MORTALITY FOR FISHER AND MARTEN IN MINNESOTA

John Erb, Barry Sampson, and Pam Coy

SUMMARY OF FINDINGS

As part of a larger project on *Martes* ecology in Minnesota, we began monitoring survival of radiocollared fishers (*Martes pennanti*) and martens (*Martes americana*) during winter 2007–2008. Including the pilot year of the study, a total of 128 martens (58 females, 70 males) and 65 fishers (36 females, 29 males) have been radiocollared. An additional 6 animals (3 martens, 3 fishers) were ear-tagged only. Of the 128 martens radiocollared, 51 are still actively monitored (18 females, 33 males), radio-contact has been lost on 23 (8 slipped collars, 15 missing), and 54 deaths have occurred. Of the 54 known marten deaths (26 females, 28 males), most have been from regulated fur trapping (*n* = 16; 13 males, 3 females) and predation (*n* = 29; 19 females, 10 males). Of the 29 predation events, 21 marten were killed by mammalian predators, 7 by raptors, and 1 by an unknown predator. While predation mortality of marten has occurred in most seasons, the majority has occurred during late winter and spring. While total marten mortality has not been noticeably sex-biased, predation mortality has been very female-biased (~2 females:1 males), while harvest mortality of marten is significantly male-biased (~4 males:1 females). The combination of male-biased harvest mortality and female-biased non-harvest mortality may produce offsetting effects on the population sex ratio. Of the 65 fishers radiocollared, 24 are still being monitored (14 females, 10 males), radio-contact was lost on 17 (12 belting hardware failures, 4 missing, 1 collar removed), and 24 deaths (12 females, 12 males) have occurred (12 [8 females, 4 males] were killed by other predators [scavenging by an eagle can't be ruled out in 1 case], 4 [1 female, 3 males] died from unknown but apparently natural causes, 4 [1 female, 3 males] were legally trapped, 2 [1 male, 1 female] were struck by vehicles (both while apparently dispersing in the fall), 1 male was accidentally trapped out of season, and 1 female was illegally trapped). Although sample size is small, 10 of the 12 predation deaths of fishers took place from late winter through spring. Seven of the 8 female fisher predation mortalities were attributed to other mammalian carnivores, while 3 of the 4 male fisher predation mortalities were attributed to raptors (all bald eagles [*Haliaeetus leucocephalus*]). Of greatest significance, 7 of the 8 female fishers killed by predators were adults, and 5 of the 7 were killed while they still had dependent young in natal dens, indirectly resulting in the death of their 14 kits. The deaths of these 5 kit-rearing females represent 36% of the adult female fishers monitored during the kit-rearing season since the study began. We hypothesize that the timing and magnitude of female mortality is a result of increased movement and increased vulnerability at this time of year. However, it remains unclear whether the pattern we have observed to date is consistent with past dynamics, and if not, whether the underlying explanation is related to short-term (e.g., periodic fluctuations in prey) or long-term (e.g., deteriorating habitat quality) changes affecting fisher energetics/activity, or a result of changes in the predator community. What is clear from initial results is that for both species, predation has been the dominant source of mortality.

INTRODUCTION

American marten and fisher are native to Minnesota, but reliable documentation of their historic distribution is limited. Undoubtedly, northeastern Minnesota was a stronghold for the marten population, though notable numbers likely occurred in the northern border areas as far west as Roseau County. Limited information suggests they occurred as far south as Crow Wing County and as far southwest as Polk County. As a result of unregulated harvest, marten were considered rare in Minnesota by 1900, and extensive logging and burning around the turn of the century further contributed to the near extirpation of marten from Minnesota by the 1930s (Swanson et al. 1945). Fishers in Minnesota appear to have historically occupied a larger geographic area than martens, extending further south and west into the hardwood dominated transition zone, including southeast Minnesota (Swanson et al. 1945, Balser and Longley 1966). The impacts of unregulated harvest and habitat alteration were equally as detrimental to fisher, with populations substantially reduced by the 1930s.

Legally, fisher and marten were unprotected in Minnesota prior to 1917, after which harvest season length restrictions were implemented. These protections were removed in the mid-1920s, and remained so until all harvest was prohibited in 1929. Seasons remained closed until 1977 for fisher and 1985 for marten, when limited harvests were reinstated. While marten harvest is legal in approximately the northern 50% of the state, most harvest occurs in counties bordering Canada, particularly in northeast and north-central Minnesota. Fisher harvest occurs in most of the northern 50% of the state, though harvest is comparatively low in extreme northeast Minnesota (Lake and Cook counties), and lower, though perhaps increasing, in the Red River Valley (western Minnesota) and the highly fragmented transitional forests in central Minnesota. Peak harvest levels have been near 4,000 and 3,500 for marten and fisher, respectively. However, due to apparent multi-year population declines for both species, harvest seasons the past 4 years were reduced from 16 days to 9 days, and in 2010, the fisher harvest limit was reduced from 5 to 2. During this 4-year period, harvests have averaged ~2,000 and 1,400 for marten and fisher, respectively.

While both species appear to have naturally re-colonized a significant portion of their historic range, Minnesota-specific information on survival and causes of mortality is limited. Except for harvest data, we are aware of only 1 published field study in Minnesota. Specifically, Mech and Rogers (1977) opportunistically radio-collared 4 marten and reported survival and home range information for those animals. This information is specific to marten, now nearly 30 years old, and based on a very limited sample size. Gathering cause-specific mortality information can be useful for informing population models, detecting unknown mortality agents, and guiding management remedies to any population declines of concern.

Krohn et al. (1994) estimated 11% annual non-harvest mortality for adult fisher in Maine, while York (1996) estimated 19% and 7% annual non-harvest mortality (including 4% poaching mortality on males) for adult male and female fisher, respectively, in Massachusetts. Excluding the first 4-5 months of life, juvenile non-harvest mortality rates have been estimated to be 28% in Maine (Krohn et al. 1994), and 0% (females) and 23% (males) in Massachusetts (York 1996). While mortality may be higher in the first months of life than the rest of the year, if we assume a similar non-harvest mortality rate during the first 4-5 months of life, we calculate that annual non-harvest mortality for juveniles would be ~56% in Maine. Combining minimum summer survival estimates for kits with telemetry estimates of survival the rest of the year, York (1996) estimated ~22% (females) and 67% (males) annual non-harvest mortality for juveniles in Massachusetts. Kelly (1977, in Paragi et al. 1994) reportedly estimated 18% annual mortality of juveniles and 44% annual mortality for adult fisher in New Hampshire. More recently, Koen et al. (2007) estimated annual mortality rate (including harvest mortality) of fishers in Ontario to be 55–67% for males, and 29–37% for females. While non-harvest mortality of adult fishers is often presumed to be ‘low,’ it has not always proven to be the case. Furthermore, there is limited data on which to assess the amount of geographic or temporal variation in non-harvest mortality of fisher.

Natural mortality, particularly via predation, appears more common with martens. Marten survival data is available from Wisconsin (McCann et al. 2010), Maine (Hodgman et al. 1994, 1997), Ontario (Thompson 1994), Oregon (Bull and Heater 2001), British Columbia (Poole et al. 2004), Alaska (Flynn and Schumacher 1997, 2009), Quebec (Potvin and Breton 1997), and Newfoundland (Fredrickson 1990). While we do not summarize details of these studies here, a couple of conclusions are worthwhile. First, when comparing across studies, annual adult non-harvest mortality rates varied from ~0.07 to 0.48. Juvenile data were rarely separated, but a few studies pooled ages, and mortality rates also fell within the above interval. While this variability may be attributable to both sampling and biological variability, the wide range suggests that it is risky to assume results from any area are applicable elsewhere. Secondly, at least 1 study (Maine; Hodgman et al. 1997) has documented significantly higher natural mortality for females compared to males, and others researchers have postulated this to be common given the typical male–biased harvest, 50:50 sex ratio at birth, and often balanced adult sex ratio (Strickland et al. 1982, Strickland and Douglas 1987). Due to male-biased harvest and our *assumed* sex-related equality in non-harvest mortality, our marten population model previously projected a very female-biased population, contradicting our preliminary capture results and suggesting that our model inputs were overestimating female survival, underestimating male survival, or incorrectly assuming a 50:50 birth sex ratio.

As part of a larger project on *Martes* ecology in Minnesota (Erb et al. 2009), we began monitoring survival and causes of mortality for fisher and marten. After initial evaluation of field methods during the pilot year of the study, winter 2008-09 marked the beginning of full-scale research activities. While details are not further discussed here, we are also collecting data on various potential correlates to survival (e.g., prey dynamics, winter severity, diet, habitat use, activity patterns, and body condition). Herein we present basic information on field methods, and descriptive information regarding number of captures and number and causes of deaths. We defer a more comprehensive and statistically-oriented analysis until a later time.

STUDY AREA

Marten research is focused on 1 study area located in northeastern Minnesota (Figure 1; Area 1), though an occasional marten is captured and radiocollared in Area 2. Area 1 (~700 km^2) includes approximately 69% mixed coniferous-deciduous forest, 15% lowland conifer or bog, 5% upland coniferous forest, 4% gravel pits and open mines, 3% regenerating forest (deciduous and coniferous), 2% shrubby grassland, 1% marsh and fen, 1% open water, and < 1% deciduous forest. Area 1 is 90% public ownership, including portions of the Superior National Forest and state and county lands. Fishers are also present in this area at low to moderate density.

Fisher research will take place in 3 areas (Figure 1; Areas 1, 2, and 3). The work in Area 3 is a collaborative effort between Camp Ripley Military Reservation, Central Lakes Community College, and the Minnesota Department of Natural Resources. While we do include animals captured in that area in our basic summaries, we do not discuss other aspects of that project in this report. Area 2 (1,075 km^2), our primary fisher study area, includes 74% deciduous forest, 11% open water, 5% lowland conifer or bog, 5% marsh and fen, 2% regenerating forest (deciduous and coniferous), 1% coniferous forest, 1% grassland, and 1% mixed forest. Area 2 is 67% public ownership, including portions of the Chippewa National Forest and state and county lands. Few martens occupy Area 2.

METHODS

We used cage traps to capture both fishers (Tomahawk Model 108) and martens (Tomahawk Model 106 or 108) during winter. Traps were typically baited with either deer (*Odocoileus virginianus*) or beaver (*Castor canadensis*) meat, and commercial lure was placed in or above the traps. We enclosed traps inside white plastic 'feed sacks' or burlap bags and further covered traps with natural vegetation. All traps were checked daily.

We physically immobilized captured animals using metal 'combs"' to restrict them to a small portion of the trap, or we restrained the animal against the side of the trap by pulling its tail through the cage mesh. We injected animals with a hand-syringe using a 10:1 mixture of ketamine and xylazine (fisher: 30 mg/kg ketamine and 3 mg/kg xylazine; marten: 20 mg/kg ketamine, 2 mg/kg xylazine) (Kreeger et al. 2002). After processing, we reversed the xylazine with yohimbine at a dosage of 0.1 mg/kg (marten) or 0.15 mg/kg (fisher). Fisher were either ear-tagged with a monel #3 tag in one ear (National Band and Tag Co., Newport, Kentucky) and a 2-piece plastic mini-tag (Dalton I.D. Systems, UK) in the other ear, or with a monel #3 tag in both ears. Marten were ear-tagged with a monel #1 tag (National Band and Tag Co., Newport, Kentucky) in each ear.

During processing, we placed animals on either chemical hand-warmers or heating pads connected to a power inverter and 12-volt battery. Portable shelters and propane heaters were also used to keep animals warm during processing. We monitored respiration, pulse, and rectal temperature during anesthesia. We weighed and sexed animals and typically removed a first pre-molar for aging. Morphological measurements taken included body, tail, and hind foot lengths, and chest, neck, and head circumferences. We removed guard hair samples for possible genotyping, and for evaluating the use of stable isotope analysis for deciphering food habits (Ben-David et al. 1997). To determine which females were pregnant in mid-winter, and eventually the percent of those that actually produce a litter in spring, we attempted to draw blood samples to measure serum progesterone concentrations (Frost et al. 1997). Antibiotics were administered subcutaneously to all animals prior to release.

During the pilot year, we deployed several radiocollar designs on fisher, including an ATS M1585 zip-tie collar (~43 g), an ATS M1930 collar (~38 g), and a Lotec SMRC-3 collar (~61 g; deployed on adult males only). Since the pilot year, we have primarily deployed ATS M1940 (~43 g) or Sirtrack TVC-162 collars (~45 g) on fisher. The majority of martens in both years have been fitted with Holohil MI-2 collars (~31 g). While not discussed in detail here, we retrofitted each collar with a temperature data-logger, in part, to allow for determination of exact time of death.

All radio-locations, except for some taken during the den-monitoring period, are obtained from fixed-wing aircraft at approximately weekly intervals. When a radiocollar emits a mortality signal, we usually investigate and recover the animal or collar within 1–2 days. To determine cause of mortality, we use a combination of field investigation and animal necropsy. Starting in the second year of the project, we also began collecting forensic samples (hair by wound, wound swabs) from all animals exhibiting signs of being predated, particularly if a mammalian predator is suspected. Forensic samples are submitted to the University of California-Davis Veterinary Genetics Laboratory. If non-predation natural causes are suspected after initial analysis (i.e., no visible trauma), the animal is submitted to the University of Minnesota's Veterinary Pathology Lab for a full pathological exam.

RESULTS AND DISCUSSION

Including the pilot year of the study, a total of 128 martens (58 females, 70 males) and 63 fishers (36 females, 27 males) have been radiocollared. An additional 6 animals (3 martens, 3 fishers) were ear-tagged only. Tooth-aging has not yet been completed for all animals, and herein, we do not report any formal survival estimates. Instead, we provide a simple overview of the fate of collared animals.

Of the 128 martens radiocollared, 51 are actively being monitored (18 females, 33 males), radio-contact has been lost on 23 (8 slipped collars, 15 missing), and 54 deaths have occurred. Of the 54 known marten deaths (26 females, 28 males), most have been from regulated fur-trapping (*n* = 16; 13 males, 3 females) and predation (*n* = 29; 19 females, 10 males). Four animals died of other natural causes, including being crushed by a tree, perforation and blockage of the intestine from a piece of bone, starvation related to an intestinal polyp, and 1 unknown natural cause. Four martens also died from capture/collar related complications. The status of one additional animal is unknown at this time, pending retrieval of the collar/animal from an underground location.

Of the 29 predation events, 21 marten were killed by mammalian predators, 7 by raptors, and 1 by an unknown predator. After censoring 5 individuals whose deaths occurred with 14 days post-capture, 75% of the predation mortalities occurred in late-winter through spring (i.e., February–May; Figure 2). Forensic (DNA) analysis of samples collected from predated marten (mammalian predation only) is incomplete. To date, DNA analysis has confirmed bobcat (*Lynx rufus*) predation in all 4 cases for which analysis is complete. Felids (bobcat or lynx [*Lynx canadensis*]) are the likely predator in 2 additional cases for which partial information (inconclusive DNA or obvious field sign) is available. Remaining forensic analysis is pending, and field evidence suggests fox (*Vulpes vulpes*) or fisher may be responsible for at least a couple predation deaths on marten.

While total marten mortality has *not* been noticeably sex-biased (52% male), and is similar to the sex ratio of the radiocollared sample (55% males), predation mortality has been female-biased (~2:1), with males comprising only 30% of the predation deaths. Conversely, harvest mortality of martens has been significantly male-biased (~4:1), comprising 81% of the harvest mortalities. Considering the timing of mortality in relation to the apparent start of the biological year for marten (~3rd week of April), current data suggests that harvest mortality of males may be *comparatively* additive (i.e., little natural mortality occurs on males post-harvest), whereas harvest mortality on females may be *comparatively* compensatory (i.e., a large percentage of natural mortality occurs on females post-harvest). If the overall population sex ratio for martens is reasonably balanced (which our initial data suggest), the subsequent sex-biases we have observed in number of predated (female-biased) and harvested (male-biased) marten suggest differential vulnerability, not differential abundance. Regardless, the combination of male-biased harvest mortality and female-biased non-harvest mortality may produce offsetting effects on the population sex ratio.

Of the 65 fishers radiocollared (36 females, 29 males), 24 are still being monitored (14 femalesF, 10 males), radio-contact has been lost with 17 animals (12 shed their collars due to belting design failures, 4 are missing, and 1 collar was removed due to neck abrasion), and 24 deaths have been confirmed (12 females, 12 males). In addition, 3 juvenile males were ear-tagged only. Of the 24 known deaths (12 females, 12 males), 12 (8 females, 4 males) were killed by other predators (scavenging by an eagle cannot be ruled out in 1 case), 4 (1 females, 3 males) died from unknown but apparently natural causes, 4 were legally trapped (1 females, 3 males), 2 (1 male, 1 female) were struck by vehicles (both while apparently dispersing in the fall), 1 male was accidentally trapped out of season, and 1 female was illegally trapped.

Although sample size is small, 83% (10 of 12) of the predation deaths of fishers took place in late winter and spring (i.e., February–May; Figure 3), similar to the pattern observed with marten. Three of the four male fisher predation deaths were attributed to bald eagles, although we cannot rule out scavenging in 1 case (only the radiocollar was retrieved directly underneath an active eagle nest). The fourth male fisher predated appears to have been killed by a mammalian predator, but full necropsy has not yet occurred. Conversely, only 1 of the 8 female predation deaths was attributed to a raptor (great-horned owl [Bubo virginianus]) suspected). We are awaiting forensic DNA analysis on many fishers killed by mammalian predators. However, bobcat was confirmed (DNA and via trail camera) in one case, and field evidence (fisher was cached) strongly indicates bobcat or lynx in another.

Of greatest significance, 7 of the 8 female fishers killed by other predators were adults, and 5 of those 7 were killed while they still had dependent kits in natal or maternal dens, indirectly resulting in the death of 14 kits. The deaths of these 5 kit-rearing females represent 36% of the adult females monitored during the kit-rearing season since the study began.

We hypothesize that 2 broad factors may explain the high mortality of kit-rearing females during late-winter and spring: increased movement and increased vulnerability (independent of movement). First, female fishers likely have high energetic demands after ‘emerging’ from winter, compounded by the added energy demands of gestation and lactation. In addition, the need to locate suitable (and multiple) natal/maternal dens likely increases movement. Preliminary data from temperature data-loggers attached to radiocollars suggest that fishers do spend increasing amounts of time (compared to winter) outside of den and rest-sites during late-winter and spring. Regardless of the motivation for increasing activity, increased movement likely increases the risk of predation. Secondly, independent of their activity level, fishers may be more vulnerable in spring, because concealment cover is diminished (i.e., before ‘green-up’) and others predators may also increase activity in spring.

Regardless of the explanation, and acknowledging the limited sample size, it seems unlikely that the high level of predation we have observed to date on nursing female fishers is sustainable, which may partially explain the recent decline in fisher abundance. However, many of the correlates to the timing of predation mortality that we have mentioned are not new challenges for adult female fisher, and the population appears to have been in decline only for the last ~7 years, suggesting that other factors may be ‘altering the system’. While it seems unlikely that the fisher mortality pattern we have observed to date is consistent with past dynamics, it remains uncertain whether the changing dynamics are related to comparatively short- (e.g., periodic fluctuations in prey) or long-term (e.g., suspected decline in natal and maternal den availability) changes affecting fisher energetics/activity, the result of relatively rapid changes in the predator community (i.e., the rapid increase in bobcats, which are known or suspected to be responsible for many of the female fisher predation deaths), or some other unknown factor.

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LITERATURE CITED

Balser, D. S., and W. H. Longley. 1966. Increase of the fisher in Minnesota. Journal of Mammalogy 47:547-550.

Ben-David, M., R.W. Flynn, and D.M. Schell. 1997. Annual and seasonal changes in diet of martens: evidence from stable isotope analysis. Oecologia 111:280-291.

Bull, E. L., and T. W. Heater. 2001. Survival, causes of mortality, and reproduction in the American marten in northeastern Oregon. Northwestern Naturalist 82:1-6.

Erb, J., B. Sampson, and P. Coy. 2009. Fisher and marten demography and habitat use in Minnesota. Summaries of Wildlife Research Findings, 2008. Minnesota Department of Natural Resources, St. Paul.

Flynn, R. W., and T. V. Schumacher. 1997. Ecology of martens in southeast Alaska. Alaska Department of Fish and Game Federal Aid Progress Report.

Flynn, R. W., and T. V. Schumacher. 2009. Temporal changes in population dynamics of American marten. Journal of Wildlife Management 73:1269-1281.

Fredrickson, R. J. 1990. The effects of disease, prey fluctuation, and clear-cutting on American marten in Newfoundland, Canada. M.S. thesis, Utah State University, Logan.

Frost, H. C., W. B. Krohn, and C. R. Wallace. 1997. Age-specific reproductive characteristics of fisher. Journal of Mammalogy 78:598-612.

Hodgman, T. P., D. J. Harrison, D. D. Katnik, and K. D. Elowe. 1994. Survival in an intensively trapped marten population in Maine. Journal of Wildlife Management 58:593-600.

Hodgman, T. P., D. J. Harrison, D. M. Phillips, and K. D. Elowe. 1997. Survival of American marten in an untrapped forest preserve in Maine. Pages 86-99 *in* G. Proulx, H. N. Bryant, P. M. Woodard, eds. *Martes*: Taxonomy, ecology, techniques, and management. Provincial Museum of Alberta, Edmonton, Canada.

Kelly, G. M. 1977. Fisher (*Martes pennanti*) biology in the White Mountain national Forest and adjacent areas. Ph.D. dissertation, University of Massachusetts, Amherst.

Kreeger, T.J., J.M. Arnemo, and J. P. Raath. 2002. Handbook of wildlife chemical immobilization: International edition. Wildlife Pharmaceuticals Inc., Fort Collins, Colorado.

Koen, E. L., J. Bowman, and C. S. Findlay. 2007. Fisher survival in eastern Ontario. Journal of Wildlife Management 71:1214-1219.

Krohn, W. B., S. M. Arthur, and T. F. Paragi. 1994. Mortality and vulnerability of a heavily trapped fisher population. Pages 137-146 *in* S. W. Buskirk, A. S. Harestad, M. G. Raphael, and R. A. Powell, eds. Martens, sables, and fishers: Biology and conservation. Cornell University Press, Ithaca, N.Y.

McCann, N. P., P. A. Zollner, and J. H. Gilbert. 2010. Survival of adult martens in northern Wisconsin. Journal of Wildlife Management 74:1502-1507.

Mech, L. D., and L. L. Rogers. 1977. Status, distribution, and movements of martens in northeastern Minnesota. USDA Forest Service Research Paper NC-143. North Central Forest Experiment Station, St. Paul.

Paragi, T. E., W. B. Krohn, and S. M. Arthur. 1994. Using estimates of fisher recruitment and survival to evaluate population trend. Northeast Wildlife 51:1-11.

Poole, K. G., A. D. Porter, A. De Vries, C. Maundrell, S. D. Grindal, and C. C. St. Clair. 2004. Suitability of a young deciduous-dominated forest for American marten and the effects of forest removal. Canadian Journal of Zoology 82:423-435.

Potvin, F., and L. Breton. 1997. Short-term effects of clearcutting on martens and their prey in the boreal forest of western Quebec. Pages 452-474 *in* G. Proulx, H. N. Bryant, P. M. Woodard, eds. *Martes*: Taxonomy, ecology, techniques, and management. Provincial Museum of Alberta, Edmonton, Canada.

Strickland, M. A., C. W. Douglas, M. Novak, and N. P. Hunziger. 1982. Pages 599-612 *in* G. A. Feldhamer and J. A. Chapman, eds. Wild mammals of North America: Biology, management, and economics. The Johns Hopkins University Press, Baltimore, Maryland.

Strickland, M. A., and C. W. Douglas. 1987. Marten. Pages 531-546 *in* M. Novak, J. A. Baker, M. E. Obbard, and B. Malloch, eds. Wild furbearer management and conservation in North America. Ontario Ministry of Natural Resources, Toronto.

Strickland, M. A., and C. W. Douglas. 1981. The status of the fisher in North America and its' management in southern Ontario. Pages 1443-1458 *in* J. A. Chapman and D. Pursley, eds. Proceedings of the worldwide furbearer conference, Frostburg, Maryland.

Swanson, G., T. Surber, and T. S. Roberts. 1945. The mammals of Minnesota. Minnesota Department of Conservation Technical Bulletin No. 2.

Thompson, I. D. 1994. Marten populations in uncut and logged boreal forests in Ontario. Journal of Wildlife Management 58:272-280.

York, E. C. 1996. Fisher population dynamics in north-central Massachusetts. M. S. thesis, University of Massachusetts, Amherst.

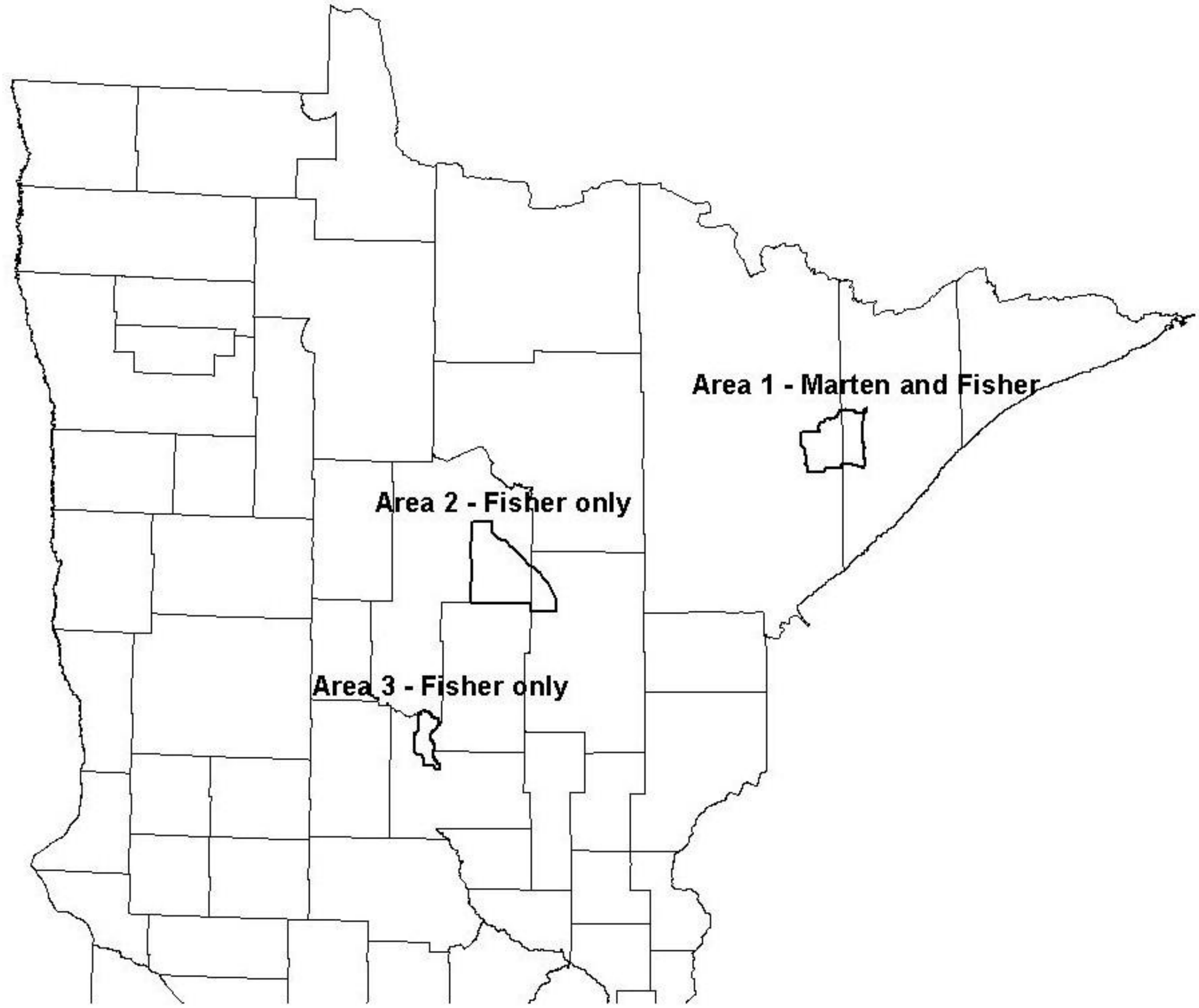

Figure 1. Fisher and marten study areas in Minnesota 2007–2010.

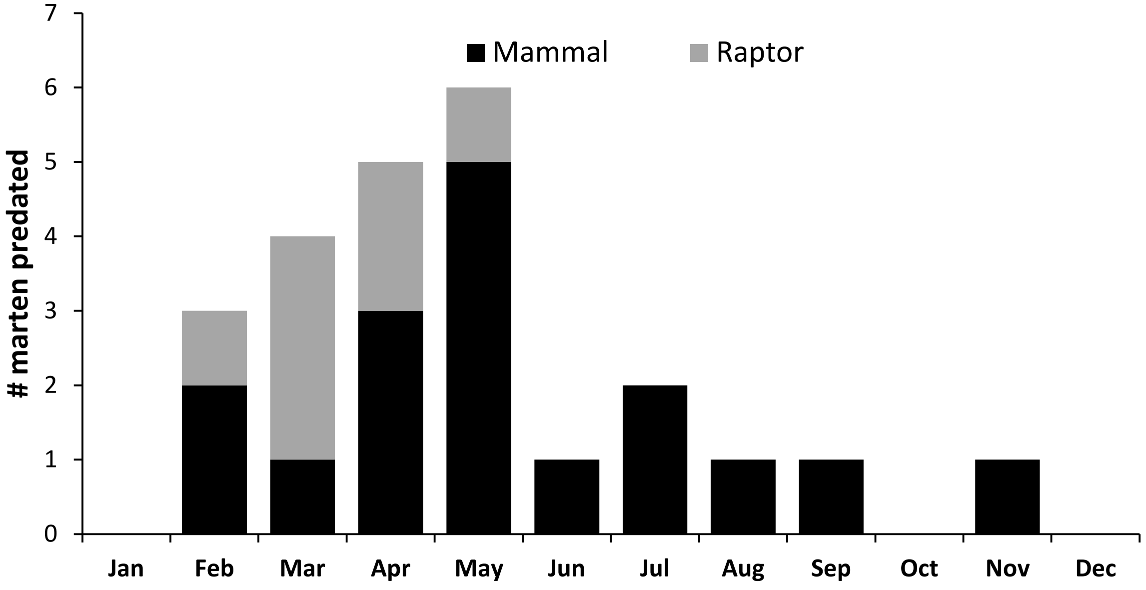

Figure 2. Seasonal timing of marten deaths attributable to predation in northeastern Minnesota, 2007–2010.

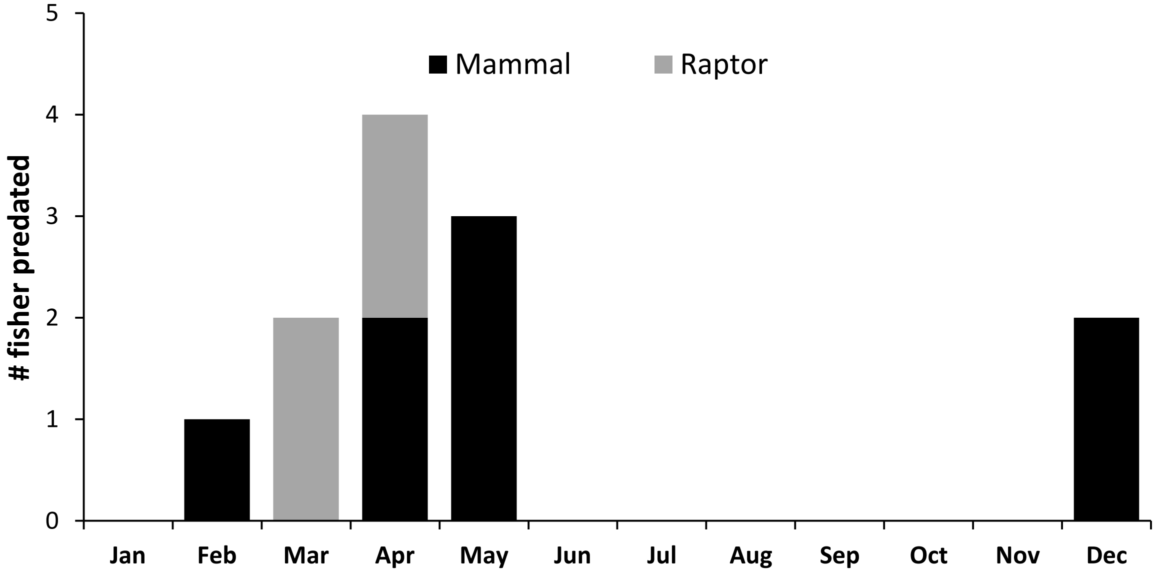

Figure 3. Seasonal timing of fisher deaths attributable to predation in north-central Minnesota, 2007–2010.

REPRODUCTIVE ECOLOGY OF FISHER AND MARTEN IN MINNESOTA

John Erb, Pam Coy, and Barry Sampson

SUMMARY OF FINDINGS

As part of a larger project on *Martes* ecology in Minnesota, we began monitoring reproductive success of radio-collared fishers (*Martes pennanti*) and martens (*Martes americana*) during spring 2009. Including the pilot year of the study, we have captured 128 martens (58 females, 70 males) and 65 fishers (36 females, 29 males). To date, age and reproductive status have been confirmed on 12 adult (≥2 years old) female martens, 83% of which produced litters. Of the 10 that produced litters, we have obtained litter counts for 8 (average minimum litter size = 3.4). Of the 15 natal or maternal dens identified, 53% have been in underground burrows, commonly in rock-laden soils, while 47% have been in elevated tree cavities (primarily cedar trees). We have also confirmed litters for 21 adult (≥2 years old) female fishers, all but 1 for which we have confirmed litter size (average litter = 2.7). Excluding 2 potential juveniles, 83% of adult (≥2 years old) female fishers produced litters, though initial data suggests that pregnancy rate and average litter size is smaller for 2 year old fishers compared to older adults. All of the fisher natal or maternal dens we have located prior to June 1 (*n* = 23) have been in elevated cavities of large diameter (average diameter at breast height [dbh] = 20.6) live trees or snags, predominantly in aspen (75%) and oak (17%). The only fisher maternal den located after June 1 was in a hollow log on the ground. Fisher kits appear to be born during the last 2 weeks of March, while marten parturition appears to be centered on the last 2 weeks of April. Both species appear to move their kits from the natal den to 1 or more different maternal dens in the first 6 weeks following birth.

INTRODUCTION

American marten and fisher are native to Minnesota, but reliable documentation of their historic distribution is limited. Undoubtedly, northeastern Minnesota was a stronghold for the marten population, though notable numbers likely occurred in the northern border areas as far west as Roseau County. Limited information suggests they occurred as far south as Crow Wing County and as far southwest as Polk County. As a result of unregulated harvest, marten were considered rare in Minnesota by 1900, and extensive logging and burning around the turn of the century further contributed to the near extirpation of marten from Minnesota by the 1930s (Swanson et al. 1945). Fishers in Minnesota appear to have historically occupied a larger geographic area than martens, extending further south and west into the hardwood dominated transition zone, including southeastern Minnesota (Swanson et al. 1945, Balser and Longley 1966). The impacts of unregulated harvest and habitat alteration were equally as detrimental to fisher, with populations substantially reduced by the 1930s.

Legally, fisher and marten were unprotected in Minnesota prior to 1917, after which harvest season length restrictions were implemented. These protections were removed in the mid-1920s, and remained so until all harvest was prohibited in 1929. Seasons remained closed until 1977 for fisher and 1985 for marten, when limited harvests were reinstated. While harvest is legal in approximately the northern half of the state, most marten harvest occurs in counties bordering Canada, particularly in northeast and north-central Minnesota. Fisher harvest occurs in most of the northern half of the state, though harvest is comparatively low in extreme northeast Minnesota (Lake and Cook counties), and lower, though perhaps increasing, in the Red River Valley (western Minnesota) and the highly fragmented transitional forests in central Minnesota. Peak harvest levels have been near 4,000 and 3,500 for marten and fisher, respectively. However, due to apparent multi-year population declines for both species, harvest seasons the past 4 years were reduced from 16 days to 9 days, and in 2010, the fisher harvest limit was reduced from 5 to 2. During this 4-year period, harvests have averaged ~2,000 and 1,400 for marten and fisher, respectively.

While both species appear to have naturally re-colonized a significant portion of their historic range, Minnesota-specific information on reproductive ecology is limited to carcass (corpora lutea, placental scar) data collected from harvested animals primarily from 1985-90 (Kuehn 1989; Minnesota Department of Natural Resources [MNDNR], unpublished data). Reproductive data are also available from other geographic areas, but questions remain on the accuracy of various methods to assess reproduction, and the amount of spatial and temporal variation in reproductive parameters. Minnesota-specific data on structures and sites used by fisher for natal and maternal dens are also lacking.

Martes pregnancy rate and litter size data are generally quantified from 1 of 4 methods: counts of corpora lutea (CL) in ovaries; counts of blastocysts (BC) in uteri; placental scar (PS) counts; or direct observation of litter size (Gilbert 1987, Mead 1994). Assuming both species are induced ovulators (but see Cherepak and Connor 1992, Frost et al. 1997), CL counts should accurately reflect copulation and ovulation rates, but all CL persist even if only 1 ovum is fertilized. Blastocyst counts reflect the number of fertilized ova, but not all BC may implant in the uterus and develop, and BC are often destroyed in poorly preserved carcasses. Hence, these 2 measures may not only overestimate litter size for parous females, but may also overestimate parturition rate (i.e., females may ovulate, 1 or more ova become fertilized, yet they fail to ultimately den and give birth). Placental scars, formed last in the reproductive process, would seem the most reliable carcass-based estimate of parturition rate and litter size. However, several authors (Payne 1982, Gilbert 1987, Strickland and Douglas 1987) have suggested that PS may not always persist long enough in mustelids to be detected during the harvest season when carcasses are easily collected, and PS can persist in some species even if fetuses are resorbed (Conaway 1955), and may be affected by observer variability (Johnson et al. 1995). Nevertheless, PS have been reliably used in the past (e.g., Coulter 1966, Crowley et al. 1990), though others have noted that reliable results may only be obtainable when doing microscopic analysis of fresh and properly preserved/prepared uteri (Mead 1994, Frost et al. 1999).

In spite of these concerns, average litter size estimates from reproductive organs do not appear to be substantially biased. Strickland and Douglas (1987), summarizing data from 136 captive marten litters, computed average litter size of 2.9 for marten. This is within the range of average litter sizes reported from ovary or uterine analysis (~2.5–3.5; Strickland et al. 1982; Strickland and Douglas 1987; Flynn and Schumacher 1997, 2009; Aune and Schladweiler 1997; MNDNR, unpublished data). For fisher, the same appears to be true, with an average litter size of 2.8 from 60 captive fisher litters (reviewed in Strickland and Douglas 1987) and 19 wild litters (York 1996), which compares favorably to estimates based on reproductive organs (2.7–3.9 (CL), 2.7–3.2 (BC), and 2.5–2.9 (PS); review in Powell 1993).

Of greater concern is the possibility that ovary, and to lesser degree uterine, analyses might consistently overestimate parturition rate, thereby underestimating annual variability in parturition rates. Various indications of pregnancy may be detected, though not all of those females may den and produce kits in spring. This might occur, for example, if ova are not fertilized following copulation or females experience nutritional stress during the period of embryonic diapause (Arthur and Krohn 1991). Overall, CL counts have generally yielded ovulation rates for fisher of ≥95% (Shea et al. 1985; Douglas and Strickland 1987; Crowley et al. 1990; Paragi 1990; MNDNR, unpublished data), while more 'direct' estimates of average parturition rate from radio-marked animals have been lower (46–75%; Crowley et al. 1990, Paragi 1990, Arthur and Krohn 1991, Paragi et al. 1994, York 1996, Truex et al. 1998, Higley and Mathews 2009), and are often highly variable. Conversely, Kuehn (1989) did not detect changes in pregnancy rate (from CL analysis) during a 64% decline in snowshoe hare indices in Minnesota.

For marten, several largely ovarian-based estimates of annual pregnancy rate have often been in the range of 80–90% (Archibald and Jessup 1984; Strickland and Douglas 1987;

Aune and Schladweiler 1997; Fortin and Cantin 2004; MN DNR, unpublished data). However, like for fisher, several marten studies have documented (also based largely on CL counts) lower or more variable pregnancy rates (Strickland and Douglas 1987, Thompson and Colgan 1987, Aune and Schladweiler 1997, Flynn and Schumacher 2009), perhaps a result of fluctuations in prey abundance (Hawley and Newby 1957, Weckwerth and Hawley 1962, Strickland 1981, Strickland and Douglas 1987, Thompson and Colgan 1987, Fryxell et al. 1999, Flynn and Schumacher 2009). We are aware of direct field-based estimates of parturition rate from radio-marked marten in only one state (Maine). Pooling samples across 4 years, 75, 81, and 92% of adult females were lactating for their 3 different study areas (Phillips 1994, Payer 1999), similar to much of the CL-based pregnancy studies.

Understanding reproductive ecology of these species also necessitates gathering information on natal and maternal den structures and selection of den-sites. Natal dens are the structures where kits are born, whereas maternal dens are sites used subsequently by the female with her dependent young. Although data are absent for Minnesota, nearly all reported fisher natal dens have been in cavities of large-diameter trees or snags (Leonard 1986, Paragi et al. 1996, Powell et al. 1997, Truex et al. 1998). In northern studies, the majority of fisher natal dens have been in large diameter aspens (*Populus* spp), and females may use up to 3 or more different maternal dens (Powell et al. 2003, Higley and Mathews 2009). Marten natal and maternal dens are also frequently in tree cavities (Gilbert et al. 1997), but may occur in more varied features (e.g., under-ground burrows, exposed root masses of trees, rock piles, large downed logs; Ruggiero et al. 1998). Though not further discussed here, the literature is also voluminous with documentation of the importance of tree cavities, large downed logs, and other forest ‘structure’ for fisher and marten resting sites (see Powell et al. 2003 for a review). Given the continuing pressure to maximize fiber production from forests (i.e., short forest rotation, biomass harvesting), the forest structural attributes critical to fisher and marten could become limiting in the future, if not already. Hence, acquiring Minnesota-specific information is critical to better inform forest management activities.

As part of a larger project on *Martes* (Erb et al. 2009), we began efforts to better describe the reproductive ecology of fisher and marten in Minnesota, specifically: 1) denning chronology; 2) structures used for natal and maternal dens; 3) vegetative characteristics in the area surrounding natal and maternal dens; 4) field-based estimates of pregnancy rate, litter size, and where possible, kit survival; and 5) the influence of age, food habits, prey fluctuations, home range habitat quality, and winter severity on reproductive success. After initial evaluation of field methods during the pilot year of the study, spring 2009 marked the beginning of full-scale research activities. Herein we present basic information on field methods, though we only report preliminary findings related to items 1, 2 and 4. We defer a more complete evaluation of results until additional data is collected or additional analysis is completed.

STUDY AREA

Marten research is focused on 1 study area located in northeastern Minnesota (Figure 1; Area 1), though an occasional marten is captured and radio-collared in Area 2 (Figure 1). Area 1 (~700 km^2) is composed of approximately 69% mixed coniferous-deciduous forest, 15% lowland conifer or bog, 5% upland coniferous forest, 4% gravel pits and open mines, 3% regenerating forest (deciduous and coniferous), 2% shrubby grassland, 1% marsh and fen, 1% open water, and < 1% deciduous forest. Area 1 is 90% public ownership, including portions of the Superior National Forest and state and county lands. Fishers are also present in this area at low to moderate density.

Fisher research will take place in 3 areas (Figure 1; Areas 1, 2, and 3). The work in Area 3 is a collaborative effort between Camp Ripley Military Reservation, Central Lakes Community College, and the MNDNR. While we do include animals captured in that area in our basic summaries, we do not discuss other aspects of that project in this report. Area 2 (1,075 km^2), our primary fisher study area, is composed of 74% deciduous forest, 11% open water, 5% lowland conifer or bog, 5% marsh and fen, 2% regenerating forest (deciduous and coniferous), 1% coniferous forest, 1% grassland, and 1% mixed forest. Area 2 is 67% public ownership, including portions of the Chippewa National Forest and state and county lands. Extremely few martens occupy Area 2.

METHODS

We used cage traps to capture both fishers (Tomahawk Model 108) and martens (Tomahawk Model 106 or 108) during winter. Traps were typically baited with deer (*Odocoileus virginianus*) or beaver (*Castor canadensis*) meat, and we placed commercial lure in or above the traps. We enclosed traps inside white plastic 'feed sacks' or burlap bags and further covered traps with natural vegetation. All traps were checked daily.

To immobilize animals, we used metal 'combs' to restrict the animal to a small portion of the trap, or restrained the animal against the side of the trap by pulling its tail through the cage mesh. Animals were injected with a hand-syringe using a 10:1 mixture of ketamine and xylazine (fisher: 30 mg/kg ketamine and 3 mg/kg xylazine; marten: 20 mg/kg ketamine, 2 mg/kg xylazine) (Kreeger et al. 2002). After processing, the xylazine was reversed with yohimbine at a dosage of 0.1 mg/kg (marten) or 0.15 mg/kg (fisher). Fisher were either ear-tagged with a monel # 3 tag in one ear (National Band and Tag Co., Newport, Kentucky) and a 2-piece plastic mini-tag (Dalton I.D. Systems, United Kingdom) in the other ear, or with a monel #3 tag in both ears. Marten were ear-tagged with a monel #1 tag (National Band and Tag Co., Newport, Kentucky) in each ear.

During processing, we placed animals on chemical hand-warmers or heating pads connected to a power inverter and 12-volt battery. Portable shelters and propane heaters were also used to keep animals warm during processing. We monitored respiration, pulse, and rectal temperature during anesthesia. We weighed and sexed animals and typically removed a first pre-molar for aging. Morphological measurements taken included body, tail, and hind foot lengths, and chest, neck, and head circumferences. We removed guard hair samples for possible genotyping and for evaluating the use of stable isotope analysis for deciphering food habits (Ben-David et al. 1997). To assist with determining which females would likely produce kits, blood samples were drawn when possible to measure serum progesterone concentrations (Frost et al. 1997). All blood samples were sent to the University of Minnesota Veterinary Diagnostics Lab for progesterone analysis. Antibiotics were administered subcutaneously to all animals prior to release.

During the pilot year, we deployed several radiocollar designs on fisher, including an ATS M1585 zip-tie collar (~43 g), an ATS M1930 collar (~38 g), and a Lotec SMRC-3 collar (~61 g; deployed on adult males only). Since the pilot year, we have primarily deployed ATS M1940 (~43 g) or Sirtrack TVC-162 collars (~45 g) on fisher. The majority of martens have been fitted with Holohil MI-2 collars (~31 g). We retrofitted each collar with a temperature data logger to provide ancillary information on winter activity and spring den attendance patterns, as well as to provide information on time of death for other study objectives.

We primarily used ground-tracking to locate den-sites, but also deployed remotely-activated cameras (Reconyx PC-85 or RC-55, Reconyx, Inc, Holmen, Wisconsin) at suspected den-sites to monitor female activity. However, we considered a female to have given birth only if kits were confirmed via sound or video/camera, or if other reliable evidence (e.g., obvious lactation, placental scars, or kit bite marks on collar) was obtained when an animal was subsequently handled as a mortality or recapture. Litter size was ascertained via visual confirmation in most cases, although we also utilized placental scar counts on any females that died during summer or fall, and for which other methods failed to produce a count. To confirm or count kits at dens located in tree cavities, we used an MVC2120-WP color video camera (Micro Video Products, Bobcaygeon, Ontario), attached to a telescoping pole, if necessary, and connected to a laptop computer. Underground dens were examined when possible using the same video probe attached to a flexible rod. Dens were only examined when the radio-marked female was not present. If video inspection equipment did not work at a particular den structure, we deployed remote cameras in an effort to obtain pictures of kits when they emerged or were moved by the female (Jones et al. 1997).

When a natal or maternal den was confirmed, we recorded den location (above/on/below-ground), as well as various location-specific details (e.g., tree species, log/tree diameter, burrow entrance attributes). We note that since birth is never observed, and kits may be moved to new dens within days following birth, distinguishing natal dens from maternal dens can rarely be done with certainty. Hence, while we report our best assessment of den type, our focus is ultimately on determining whether initial dens (be they natal or maternal) used early in the kit-rearing period (e.g., prior to 1 June) are structurally different than dens used as kits get larger and more mobile. Hence, we organize our tabular reporting on the date at which the den was first documented to be in use.

We will also be collecting more detailed information on vegetative characteristics of the site surrounding each den structure, with a goal of not only developing a biologically meaningful den-site selection model, but also to do so using methods and metrics that will be 'transferrable' to long-term habitat monitoring over large areas using existing forest sampling data (e.g., see Zielinski et al. 2006). Following the United States Forest Service's Forest Inventory and Analysis (FIA) protocol, we will quantify vegetative characteristics in a 1-acre (120-ft radius) area surrounding the den structure by sampling in 4 circular subplots, each being 0.04-acre (24-ft radius) in size. One subplot will be centered on the den structure, with the other 3 subplots centered 120 feet from the den at 360°, 120°, and 240°. Within each subplot, 3 24-ft coarse woody debris sampling transects are established, originating from the subplot center, and oriented at 30º, 150º, and 270º. Deviating from FIA protocol, we also establish 3 (not 1, as with FIA) 0.003-acre (6.8-ft radius) circular micro-plots for estimating sapling density, each micro-plot situated at the end of the 3 coarse woody debris sampling transects. Details of vegetation sampling methods within each subplot will be outlined in subsequent years as results become available. Herein, we simply note that we will collect quantitative data on: 1) mean DBH and basal area of live trees, overall and by species; 2) percent overhead (angular) canopy; 3) sapling density; 4) understory cover density; 5) density and volume of snags and stumps; 6) volume of coarse woody debris; 7) distance to improved road; and 8) distance to water. Canopy structure will also be categorized based on number and distribution of canopy layers.

To better understand any observed fluctuations in reproductive parameters, we are also collecting data on factors that may influence reproductive success, including winter severity and prey fluctuations. In each study area, a temperature monitor was placed in each of 6 cover types. Each sensor records temperature every 30 minutes, and was placed on the north-facing side of a tree situated along a transect that we used for recording cover-type specific snow information. In addition to monitoring temperature at each of 3 locations along a transect and repeated once within each 10-day interval (1 December–1 April), we recorded snow depth and 2 measures of snow compaction. Two snow compaction tools were constructed using PVC pipe, one each with an end-cap similar in diameter to a typical marten and fisher track in the snow. Each pipe length was then adjusted to ensure the pipe-specific load (g/cm^2) was similar to marten and fisher foot-load measures (females) reported by Krohn et al. (2004). Depth of snow compaction was recorded by dropping each load tool from 1 in. above snow level and measuring compaction depth.

Prey-sampling transects have also been established in both study areas. Prey sampling is being conducted primarily to document between-area differences in prey abundance, annual within-area fluctuations in prey, and ultimately to assess whether fisher or marten habitat use, diet, survival, or reproductive success is correlated with prey dynamics. Prey-sampling transects (n ≈ 125 in each study area) consist of 10 sampling locations (2 parallel lines of 5 stations) spaced 20 m apart, with transects distributed in 6 cover types throughout each study area. Transects are generally oriented perpendicular to roads or trails, with the first plot 30 m off the trail. In spring, we count snowshoe hare (*Lepus americanus*) pellets in a 1-m^2 plot at each sampling station (McCann et al. 2008). During fall, small mammal snap-trapping will occur for 2 consecutive days at the same sampling stations, similar to protocol used on an existing small mammal survey in Minnesota (Aarhus-Ward 2009). During both spring (hare pellet sampling) and fall (small mammal trapping), we also will count the number of red squirrels (*Tamiasciurus hudsonicus*) observed or heard along each transect. Rather than using 10-min point counts (e.g., Mattson and Reinhart 1996, Bayne and Hobson 2000) with our small mammal/hare pellet stations as the sampling points, we will simply record the number of unique squirrels observed/heard along each transect while checking pellet plots and small mammal traps. Information on white-tailed deer (*Odocoileus virginianus*) and ruffed grouse (*Bonasa umbellus*) populations may be available from existing surveys or population models.

RESULTS AND DISCUSSION

Including the pilot year of the study, a total of 128 martens (58 females, 70 males) and 65 fishers (36 females, 29 males) have been captured. Herein we provide a basic summary of data collected to date on denning chronology, den structures, pregnancy status, and litter size. Because tooth aging has not yet been completed for all animals, and some yet-to-be-aged females may be only 1 year of age (i.e., not capable of producing kits), we present results only for animals known to be ≥2 years of age during spring den visits, or those of unknown age, but for which we have confirmed parturition at the time of this writing (i.e., until age is known, we do not include animals that we have confirmed to be nulliparous). No reproductive data are yet available for spring 2011.

Treating females that were alive during multiple parturition periods as independent units, and excluding females known to be 1 year of age during the parturition period, a total of 17 female martens have been available for monitoring during the kit-rearing season. However, we have confirmed age and reproductive status for only 12 female martens (Table 1). Three additional females for which we do not yet have age data were confirmed to be nulliparous, and we were unable to confirm birth status for 2 adult females in 2009. Because it has been more difficult to inspect marten natal dens with video equipment, we have had to rely more on remote cameras to obtain litter information when kits are moved by the female, or when they are older and more mobile. Hence, many estimates of marten litter size are reported as minimums. Acknowledging this, average size of 8 litters confirmed to date is 3.4 (Table 1). Based on initial data, it appears marten kits are typically born in mid- to late-April. Given the timing of our marten capture (blood-drawing) operations (i.e., mid-December through early February), preliminary results indicate that marten progesterone concentrations have not sufficiently elevated in pregnant animals at that time to allow us to confirm mid-winter pregnancy status.

A total of 15 marten natal or maternal dens have been located to date (Table 2). Based on 11 dens confirmed prior to June 1 of each year, 64% have been in tree cavities, while 36% have been in underground tunnels (Table 2). We have confirmed only 4 maternal dens used after June 1, and all 4 were in underground burrows situated in rock-laden soils (Table 2). Most female marten appear to move their kits from their natal den to 1 or more maternal dens in the first 6 weeks following birth.

Similar to marten, we treat female fishers that were alive during multiple parturition periods as independent units. Excluding individuals known to be 1 year of age during the parturition period, a total of 31 female fishers have been available for monitoring during the kit-rearing season. At the time of this writing, we have confirmed both age and reproductive status for 20 female fishers and reproductive status (but not yet age) from 8 additional females (Table 3). In addition, we have confirmed reproductive status for 3 females for which there is currently some uncertainty or discrepancy in age data collected. Pooling all female fishers that produced a litter (*n* = 20), average litter size is 2.7 (range = 1–4). Age-specific sample sizes are small (Table 3), but there is some indication that average litter size for 2 year olds is lower than older females (~2.4 versus 2.9). There also is some indication that birth rates are lower for 2-year-olds compared to older females. Parturition rate for 2 year olds is between 50 and 71% depending on the age assigned to 3 females with uncertain age assignment. Parturition rate for female fishers ≥3 years of age is 85–86%. However, the only 2 apparently 'failed' reproductive events detected for adults ≥3 years of age were from the same 7+ year old female in 2009 and 2010.

Based on data collected to date, it appears fisher kits are typically born in mid- to late-March, or ~1 month earlier than marten kits. Perhaps owing to earlier parturition, as well as apparently longer active gestation (Powell et al. 2003), it appears that the fisher progesterone concentrations are sufficiently elevated in pregnant females at the time of our winter capture operations (i.e., mid-December through mid-March) to allow accurate assessment of mid-winter pregnancy status using seasonal hormone profiles developed in Maine (Frost et al. 1999). Furthermore, although blood has not been collected on all females, all those confirmed pregnant in mid-winter (progesterone) produced litters in spring, suggesting that overwinter disruption of pregnancy is not common.

A total of 29 fisher natal or maternal dens have been confirmed. With 1 exception (hollow base of a live oak tree), all natal/maternal dens located prior to 1 June of each year (n = 28) have been in elevated tree cavities. Only 1 maternal den has been located after June 1, in a hollow log on the ground (15.7”-diameter sugar maple). Of the 28 natal/maternal dens located prior to June 1, detailed measurements have been completed on only 24. Cavities have been located in both live trees and snags (overall average dbh = 20.6 inches), of which 44% were in live aspen (average dbh = 21.5 inches), 31% in aspen snags (average dbh = 19.9 inches), 17% in live oaks (average dbh = 19.5 inches), and 1 each in a pine snag (dbh = 21.9 inches) and live cedar (dbh = 20.3 inches) (Table 4). While monitoring has not been standardized across animals, many female fishers appear to move kits from their natal den to at least 2 different maternal dens prior to 1 June.

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LITERATURE CITED

Aarhus-Ward, A. J. 2009. 2007-2008 Small mammal survey report. 1854 Treaty Authority Technical Report 09-05. Duluth, Minnesota.

Archibald, W. R., and R. H. Jessup. 1984. Population dynamics of pine marten in the Yukon Territory. Pages 81-97 *in* R. Olson, R. Hastings, F. Geddes, eds. Northern ecology and resource management. University of Alberta Press, Edmonton, Canada.

Arthur, S. M., and W. B. Krohn. 1991. Activity patterns, movements, and reproductive ecology of fishers in southcentral Maine. Journal of Mammalogy 72:379-385.

Aune, K. E., and P. Schladweiler. 1997. Age, sex structure, and fecundity of the American marten in Montana. Pages 61-77 *in* G. Proulx, H. N. Bryant, P. M. Woodard, eds. *Martes*: Taxonomy, ecology, techniques, and management. Provincial Museum of Alberta, Edmonton, Canada.

Balser, D. S., and W. H. Longley. 1966. Increase of the fisher in Minnesota. Journal of Mammalogy 47:547-550.

Bayne, E., and K. Hobson. 2000. Relative use of continuous and fragmented boreal forest by red squirrels (*Tamiasciurus hudsonicus*). Canadian Journal of Zoology 78:359-365.

Ben-David, M., R. W. Flynn, and D. M. Schell. 1997. Annual and seasonal changes in diet of martens: evidence from stable isotope analysis. Oecologia 111:280-291.

Cherepak, R. B., and M. L. Connor. 1992. Constantly pregnant - well almost. Reproductive hormone levels of the fisher (*Martes pennanti*), a delayed implanter. Norwegian Journal of Agricultural Sciences (Supplement) 9:150-154.

Conaway, C. H. 1955. Embryo resorption and placental scar formation in the rat. Journal of Mammalogy 36: 516-532.

Coulter, M. W. 1966. Ecology and management of fishers in Maine. Ph.D. dissertation, Syracuse University, New York, New York.

Crowley, S. K., W. B. Krohn, and T. F. Paragi. 1990. A comparison of fisher reproductive estimates. Transactions of the Northeast Section of the Wildlife Society 47:36-42.

Douglas, C. W., and M. A. Strickland. 1987. Fisher. Pages 511-530 *in* M. Novak, J. A. Baker, M. E. Obbard, and B. Malloch, eds. Wild furbearer management and conservation in North America. Ontario Ministry of Natural Resources, Toronto, Canada.

Erb, J., B. Sampson, and P. Coy. 2009. Fisher and marten demography and habitat use in Minnesota. Pages 51-60 *in* M. W. DonCarlos, R. O. Kimmel, J. S. Lawrence, and M. S. Lenarz, eds. Summaries of wildlife research findings, 2008. Minnesota Department of Natural Resources, Wildlife Populations and Research Unit, St. Paul.

Flynn, R. W., and T. V. Schumacher. 1997. Ecology of martens in southeast Alaska. Alaska Department of Fish and Game Federal Aid Progress Report. Juneau.

Flynn, R. W., and T. V. Schumacher. 2009. Temporal changes in population dynamics of American marten. Journal of Wildlife Management 73:1269-1281.

Fortin, C., and M. Cantin. 2004. Harvest status, reproduction, and mortality in a population of American martens in Quebec, Canada. Pages 221-234 *in* D. J. Harrison, A. K. Fuller, and G. Proulx, eds. Martens and fishers in human-altered environments: an international perspective. Springer Science, New York, New York.

Frost, H. C., W. B. Krohn, and C. R. Wallace. 1997. Age-specific reproductive characteristics in fishers. Journal of Mammalogy 78:598-612.

Frost, H. C., E. C. York, W. B. Krohn, K. D. Elowe, T. A. Decker, S. M. Powell, and T. K. Fuller. 1999. An evaluation of parturition indices in fishers. Wildlife Society Bulletin 27:221-230.

Fryxell, J. M., J. B. Falls, E. A. Falls, R. J. Brooks, L. Dix, and M. J. Strickland. 1999. Density dependence, prey dependence, and population dynamics of martens in Ontario. Ecology 80:1311-1321.

Gilbert, F. F. 1987. Methods for assessing reproductive characteristics of furbearers. Pages 180-189 *in* M. Novak, J. A. Baker, M. E. Obbard, and B. Malloch, eds. Wild furbearer management and conservation in North America. Ontario Ministry of Natural Resources, Toronto, Canada.

Gilbert, J. H., J. L. Wright, D. J. Lauten, and J. R. Probst. 1997. Den and rest-site characteristics of American marten and fisher in northern Wisconsin. Pages 135-145 *in* G. Proulx, H. N. Bryant, P. M. Woodard, eds. *Martes*: Taxonomy, ecology, techniques, and management. Provincial Museum of Alberta, Edmonton, Canada.

Hawley, V. D., and F. E. Newby. 1957. Marten home ranges and population fluctuations. Journal of Mammalogy 38:174-184.

Higley, J. M., and S. Matthews. 2009. Fisher habitat use and population monitoring on the Hoopa Valley Reservation, California. Final Report USFWS TWG U-12-NA-1. Hoopa Valley Tribe, Hoopa, California.

Johnson, W. N., T. F. Paragi, and D. D. Katnik. 1995. The relationship of wildland fire to lynx and marten populations and habitat in interior Alaska. Final Report. U. S. Fish and Wildlife Service, Galena, Alaska.

Jones, L. L. C., M. G. Raphael, J. T. Forbes, and L. A. Clark. 1997. Using remotely activated cameras to monitor maternal dens of martens. Pages 329-349 *in* G. Proulx, H. N. Bryant, P. M. Woodard, eds. *Martes*: taxonomy, ecology, techniques, and management. Provincial Museum of Alberta, Edmonton, Canada.

Kreeger, T. J., J. M. Arnemo, and J. P. Raath. 2002. Handbook of wildlife chemical immobilization: International edition. Wildlife Pharmaceuticals, Inc. Fort Collins, Colorado.

Krohn, W. B., C. Hoving, D. Harrison, D. Phillips, and H. Frost. 2004. *Martes* foot-loading and snowfall patterns in eastern North America: implications to broad-scale distributions and interactions of mesocarnivores. Pages 115-131 *in* D. J. Harrison, A. K. Fuller, and G. Proulx, eds. Martens and fishers in human-altered environments: an international perspective. Springer Science, New York, New York.

Kuehn, D. W. 1989. Winter foods of fishers during a snowshoe hare decline. Journal of Wildlife Management 53:688-692.

Leonard, R. D. 1986. Aspects of reproduction of the fisher in Manitoba. The Canadian Field-Naturalist 100:32-43.

Mattson, D. J., and D. P. Reinhart. 1996. Indicators of red squirrel (*Tamiasciurus hudsonicus*) abundance in the whitebark pine zone. The Great Basin Naturalist 56:272-275.

McCann, N. P., R. A. Moen, and G. J. Niemi. 2008. Using pellet counts to estimate snowshoe hare numbers in Minnesota. Journal of Wildlife Management 72:955-958.

Mead, R. A. 1994. Reproduction in the *Martes*. Pages 404-422 *in* S. W. Buskirk, A. S. Harestad, M. G. Raphael, and R. A. Powell, eds. Martens, sables, and fishers: Biology and conservation. Cornell University Press, Ithaca, New York.

Paragi, T. F. 1990. Reproductive biology of female fisher in southcentral Maine. M. S. thesis, University of Maine, Orono.

Paragi, T. E., W. B. Krohn, and S. M. Arthur. 1994. Using estimates of fisher recruitment and survival to evaluate population trend. Northeast Wildlife 51:1-11.

Paragi, T. E., S. M. Arthur, and W. B. Krohn. 1996. Importance of tree cavities as natal dens for fishers. Northeastern Journal of Applied Forestry 13:79-83.

Payer, D. C. 1999. Influences of timber harvesting and trapping on habitat selection and demographic characteristics of American marten. Ph.D. dissertation, University of Maine, Orono.

Payne, N. F. 1982. Assessing productivity of furbearers. Pages 39-50 *in* G. C. Sanderson, ed. Midwest furbearer management. Proceedings of the 43rd Midwest Fish and Wildlife Conference, Wichita, Kansas.

Phillips, D. M. 1994. Social and spatial characteristics, and dispersal of marten in a forest preserve and industrial forest. M. S. thesis, University of Maine, Orono.

Powell, R. A. 1993. The fisher: life history, ecology, and behavior. University of Minnesota Press, Minneapolis.

Powell, R. A., S. W. Buskirk, and W. J. Zielinski. 2003. Fisher and marten. Pages 635-649 *in* G. A. Feldhamer, B. C. Thompson, and J. A. Chapman, eds. Wild mammals of North America: biology, management, and conservation, The Johns Hopkins University Press, Baltimore, Maryland.

Powell, S. M., E. C. York, J. J. Scanlon, and T. K. Fuller. 1997. Fisher maternal den sites in central New England. Pages 265-278 *in* G. Proulx, H. N. Bryant, P. M. Woodard, eds. *Martes*: taxonomy, ecology, techniques, and management. Provincial Museum of Alberta, Edmonton, Canada.

Ruggiero, L. F., D. E. Pearson, and S. E. Henry. 1998. Characteristics of American marten den sites in Wyoming. Journal of Wildlife Management 62:663-673.

Shea, M. E., N. L. Rollins, R. T. Bowyer, and A. G. Clark. 1985. Corpora lutea number as related to fisher age and distribution in Maine. Journal of Wildlife Management 49:37-40.

Strickland, M. A. 1981. Fisher and marten study 1979-80 and 1980-81. Ontario Ministry of Natural Resources Algonquin Region Progress Report 7. Owen Sound, Ontario, Canada.

Strickland, M. A., and C. W. Douglas. 1987. Marten. Pages 531-546 *in* M. Novak, J. A. Baker, M. E. Obbard, and B. Malloch, eds. Wild furbearer management and conservation in North America. Ontario Ministry of Natural Resources, Toronto, Canada.

Strickland, M. A., C. W. Douglas, M. Novak, and N. P. Hunziger. 1982. Pages 599-612 *in* G. A. Feldhamer, and J. A. Chapman, eds. Wild mammals of North America: biology, management, and economics. The Johns Hopkins University Press, Baltimore, Maryland.

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

Thompson, I. D., and P. W. Colgan. 1987. Numerical responses of martens to a food shortage in northcentral Ontario. Journal of Wildlife Management 51:824-835.

Truex, R. L., W. J. Zielinski, R. T. Gollightly, R. H. Barrett, and S. M. Wisely. 1998. A meta-analysis of regional variation in fisher morphology, demography, and habitat ecology in California (draft report). California Department of Fish and Game, Wildlife Management Division, Nongame Bird and Mammal Section. Berkeley, California.

Weckwerth, R. P., and V. D. Hawley. 1962. Marten food habits and population fluctuations in Montana. Journal of Wildlife Management 26:55-74.

York, E. C. 1996. Fisher population dynamics in north-central Massachusetts. M. S. thesis, University of Massachusetts, Amherst.

Zielinski, W. J., R. L. Truex, J. R. Dunk, and T. Gaman. 2006. Using forest inventory data to assess fisher resting habitat suitability in California. Ecological Applications 16:1010-1025.

Table 1. Parturition status and litter size for radiocollared female marten in Minnesota[1].

ID	Year	Age	Litter	Litter size
M09-280	2010	2	Yes	≥3
M09-262	2009	2	Yes	
M09-254	2010	2	Yes	≥3
M09-264	2009	3	No	
M10-290	2010	3	No?	
M09-262	2010	3 or 4	Yes	4
M09-247	2009	5	Yes	4
M08-140	2008	9	Yes	
M09-286	2009	9	Yes	≥3
M08-140	2009	10	Yes	≥2
M09-286	2010	10	Yes	≥4
M09-237	2010	11	Yes	4

[1] Excludes unknown-aged nulliparous females and all 1-year-olds.

Table 2. Natal and maternal den structures used by radiocollared female marten in Minnesota.

ID	Year	Date confirmed	Den type	Den structure	Den details
M09-254	2010	4/19	natal	tree cavity	15.9" dbh live red maple
M09-237	2010	4/19	natal	tree cavity	16.8" dbh live tamarack
M08-140	2009	4/21	natal	underground burrow	rock-laden soil
M09-280	2010	4/28	natal	underground burrow	rock-laden soil
M08-140	2008	4/30	natal	underground burrow	rock-laden soil
M09-286	2010	5/7	natal	tree cavity	21.5" dbh live cedar
M09-262	2010	5/10	natal	tree cavity	18.8" dbh live cedar
M09-286	2009	5/19	natal	tree cavity	16.1" dbh live cedar
M09-286	2010	5/19	maternal	tree cavity	18.6” dbh live cedar
M09-286	2009	5/22	maternal	tree cavity	20.9" dbh live cedar
M09-254	2010	5/26	maternal	underground burrow	rock-laden soil
M09-286	2010	6/12	maternal	underground burrow	rock-laden soil
M08-140	2009	7/6	maternal	underground burrow	base of snag, rocky soil
M09-286	2009	7/9	maternal	underground burrow	along roots; base of cedar
M09-254	2010	7/12	maternal	underground burrow	rock-laden soil

Table 3. Parturition status and litter size for radiocollared female fishers in Minnesota[1].

ID	Year	Age	Litter	Litter size
F10-501	2010	1 or 2	No	
F09-362	2009	1 or 2	No	
F08-375	2008	2	Yes	≥2
F09-360	2009	2	Yes	2
F08-304	2009	2	Yes	2
F08-077	2009	2	Yes	4
F09-362	2009	2	No	
F09-364	2009	2	No	
F10-501	2011	2 or 3	Yes	2
F09-394	2009	3	Yes	3
F08-375	2009	3	Yes	3
F08-353	2009	3	Yes	3
F10-503	2011	3	Yes	2
F09-380	2009	4	Yes	3
F09-394	2010	4	Yes	2
F10-507	2011	4	Yes	3
F08-353	2010	4	Yes	3
F09-394	2011	5	Yes	3
F09-354	2009	7	No?	
F09-354	2010	8	No?	
F09-370	2009	11	Yes	3
F10-328	2010		Yes	2
F09-461	2010		Yes	3
F10-507	2010		Yes	3
F11-316	2011		Yes	1
F11-340	2011		Yes	3

[1] Excludes unknown-aged nulliparous females, and all 1-year-olds.

Table 4. Natal and maternal den structures used by radiocollared female fishers in Minnesota.

ID	Year	Date confirmed	Den type	Den structure	Den details
F10-507	2011	3/9	Natal	Tree cavity	16.9” dbh aspen snag
F08-353	2010	3/24	Natal	Tree cavity	15.1" dbh live aspen
F10-507	2010	3/26	Natal	Tree cavity	25.6” dbh live oak
F09-394	2010	3/26	Natal	Tree cavity	24.9” dbh live aspen
F09-394	2011	3/28	Natal	Tree cavity	15.8” dbh live aspen
F10-501	2011	3/30	Natal	Tree cavity	20.3” dbh live cedar
F08-375	2009	4/7	Natal	Tree cavity	21.9” dbh w. pine snag
F09-360	2009	4/8	Natal	Tree cavity	15.3” dbh aspen snag
F08-353	2009	4/8	Natal	Tree cavity	23.2" dbh live aspen
F09-394	2009	4/9	Natal	Tree cavity	13.8” dbh aspen snag
F09-394	2010	4/9	Maternal	Tree cavity	22.1” dbh live aspen
F09-394	2011	4/9?	Maternal	Tree cavity	24” dbh live aspen
F09-461	2010	4/11	Natal	Tree cavity	18.3" dbh live oak
F10-507	2011	4/12	Maternal	Tree cavity	15.1” dbh live oak
F10-507	2010	4/13	Maternal	Tree cavity	22.1” dbh aspen snag
F09-380	2009	4/14	Natal	Tree cavity	23.6” dbh aspen snag
F09-370	2009	4/15	Natal	Tree cavity	23.5” dbh aspen snag
F09-394	2009	4/18	Natal	Tree cavity	21.5” dbh live aspen
F09-394	2010	4/20	Maternal	Tree cavity	26.1” dbh live aspen
F08-353	2010	4/22	Maternal	Tree cavity	24.3" dbh aspen snag
F09-394	2011	5/4	Maternal	Tree cavity	19.8” dbh live aspen
F09-461	2010	5/18	Maternal	Tree cavity	22.3” dbh live aspen
F09-360	2009	5/29	Maternal	Hollow tree base	19.1” dbh live oak
F08-375	2008	6/25	Maternal	Hollow log	15.7" diam. sugar maple

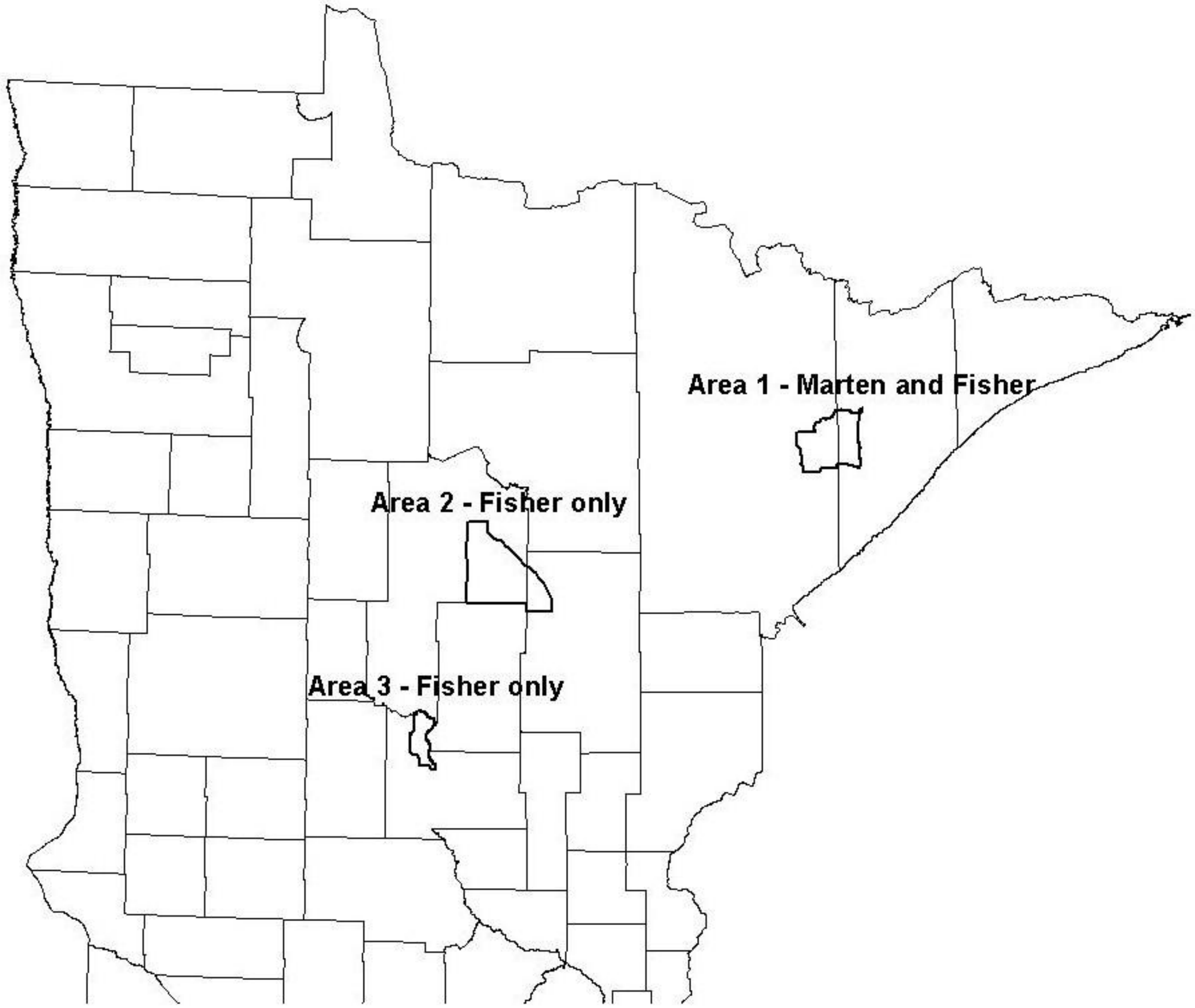

Figure 1. Fisher and marten study areas in Minnesota, 2008–2010.

HABITAT SELECTION BY MALE RUFFED GROUSE AT MULTIPLE SPATIAL SCALES

Meadow J. Kouffeld[1], Michael A. Larson, and R. J. Gutiérrez[1]

SUMMARY OF FINDINGS

We collected data about ruffed grouse drumming structures during 2 spring field seasons. During 2009, we located 454 used drumming structures within 200 m of survey transects, and we sampled vegetation characteristics at 434 of them and at 434 nearby unused structures. During 2010, we located 449 individual drumming structures and sampled vegetation characteristics at all of them and at 449 nearby unused structures. Data collection is complete, and we have begun analyzing the data. Research results will be available beginning in fall 2011.

INTRODUCTION

The Minnesota Department of Natural Resources (MNDNR) set a goal of increasing the hunting harvest of ruffed grouse (*Bonasa umbellus*) from a mean of 561,000 birds/year (1976–2005; MNDNR, unpublished data) to a mean of 650,000 birds/year (MNDNR 2007). Achieving that goal likely will require increasing the quality or quantity of ruffed grouse habitat in Minnesota.

Although ruffed grouse occur in forest stands not dominated by aspen (*Populus* spp.) and in regions where aspen is sparse or does not exist (Devers et al. 2007), they reach their highest densities in aspen forests (Rusch et al. 2000). Young aspen stands provide dense vertical stems used as cover by grouse, particularly drumming males and females with broods. The flower buds of older male aspen trees are a favored winter food for grouse. Classic grouse habitat, therefore, consists of close juxtaposition of multiple age classes of aspen in relatively small patches, so within an area the size of a typical grouse home range a grouse can access the various resources the different age classes provide (Gullion and Alm 1983, Gullion 1984).

All of the MNDNR's Subsection Forest Resource Management Plans (SFRMPs) that have reached the stage of defining "Desired Future Forest Conditions" have prescribed a conversion of many acres of managed forest land from an aspen cover type to another cover type (-5 to -33%, MNDNR 2001, 2003, 2004 [revised 2006]). Recent plans for the 2 national forests in Minnesota call for similar conversions (U. S. Forest Service [USFS] 2004*a*,*b*). Restoration of an historical forest composition (i.e., range of natural variation or pre-settlement benchmark) was used to justify reducing the area of the aspen cover type in the future. Furthermore, global climate change is likely to influence conversions of forest cover types and other aspects of ruffed grouse habitat.

Although Gullion clearly showed an association between ruffed grouse and aspen (Gullion and Alm 1983), he did not explicitly investigate landscape patterns in ruffed grouse habitat. Furthermore, he left some uncertainty about the effect of pine (*Pinus* spp.) stands in particular on ruffed grouse habitat by reporting high densities of drumming males associated with aspen clones in pine plantations under some unspecified conditions (Gullion 1990). Zimmerman (2006) conducted the only recent analysis of ruffed grouse habitat at a landscape scale. He found that the densities of drumming male grouse along ~5-km strip transects were most highly correlated ($r \approx 0.53$) with an index of evenness in the distribution of land area among 6 types of land cover, including 4 types of forest overstory. Evenness was correlated with the proportions of aspen and conifer cover types (positively and negatively, respectively). The data, therefore, were inconclusive about the effects of specific forest cover types on the density of drumming grouse at a landscape scale. Thus, it remains uncertain what the effect of landscape-scale changes in forest overstory composition will be on ruffed grouse populations.

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[1] Department of Fisheries, Wildlife, and Conservation Biology, University of Minnesota, 1980 Folwell Avenue, St. Paul, Minnesota 55108

At the scale of a few forest stands, the preference of grouse for aspen in several age classes is well known (Gullion 1984, Rusch et al. 2000). Zimmerman (2006) found that variation in the number of drumming male grouse in individual forest stands was best explained by a model that included patch shape and 9 forest overstory types. More grouse were located in young aspen stands and stands with low edge density, and fewer were in mixed hardwood-conifer stands and mature spruce-fir (*Picea* spp., *Abies balsamea*) stands. Less is known, however, about the influence on grouse of the following patch and adjacency characteristics of forest stands: the presence of conifers in aspen stands, the presence of aspen clones in conifer stands, the relative importance of different age classes of aspen, and variation in the density of woody stems regenerating after harvesting aspen.

We designed this study to address remaining uncertainties about the relationships between grouse habitat and forest characteristics at multiple spatial scales. Our results will help wildlife managers make forest management recommendations consistent with achieving the ruffed grouse harvest goal stated in the MNDNR's Strategic Conservation Agenda.

OBJECTIVES

1. To determine forest characteristics which are correlated with the presence of male ruffed grouse in stands and at specific drumming structures.
2. To determine forest characteristics correlated with the abundance of male ruffed grouse within landscapes comprised of many forest stands.

STUDY AREA

The study area is in portions of Red Lake Wildlife Management Area and adjacent Beltrami Island State Forest in Roseau, Beltrami, and Lake of the Woods counties in north-central Minnesota. The study area encompassed approximately 251,038 ha and was located in the Laurentian Mixed Forest Province (MNDNR 2003). The study area was divided into 2 sampling units with the northern part (116,454 ha) of the study area dominated by conifer and the southern part (134,584 ha) dominated by aspen. We did not include Red Lake Band Tribal Lands in our study.

METHODS

Data for this study will come from 2 sources. We collected new data by surveying grouse and measuring vegetation characteristics at a study area that is as representative as possible of forests in northern Minnesota. These data will be used to analyze habitat selection by grouse at all 3 spatial scales (i.e., drumming structure, forest stand, and landscape). We will also use existing data from the MNDNR's annual ruffed grouse drumming count survey routes to conduct an independent analysis of habitat selection at the landscape scale.

Data Collection

New field data—We identified 60 3- to 5-km transects in the study area. Each transect was delineated by starting at a point along a road or trail that was nearest to one of 30 randomly located points in the aspen study site and 30 randomly located points in the conifer study site. We determined randomly the directions of each transect originating from that point along the road or trail and also when it intersected another road or trail. Drumming grouse can be detected from approximately 200 m away (Zimmerman 2006), so we created a 200-m buffer around each transect to define sample landscapes. The transects were ≥400 m apart at all points. We divided the sample landscapes into 3 groups of 20 based on the proportions of aspen and conifer cover—those with the most aspen, those with the most conifer, and those with the most equal proportions. The aspen and conifer cover types comprised ≥50% of each sample transect. We randomly selected 10 transects from each of the 3 groups to survey for our study.

Each of the 30 selected transects were surveyed on foot beginning 0.5 hours before sunrise during 8 different mornings during an 8-week period ending on the Friday nearest 31 May. When drumming grouse were detected during a survey, the exact location of each one was determined by approaching it and identifying the log or other structure on which it was standing to drum, often indicated by the presence of fresh droppings. Universal Transverse Mercator (UTM) coordinates were taken using a hand held global positioning system (GPS) unit at drumming structures, and the drumming structure's location was confirmed by approaching during subsequent surveys.

During Zimmerman's (2006) study, only 6% of detections were >200 m from the transect, and the probability of detecting a drumming grouse within 175 m of survey transects was not correlated with the distance from the transect. Assuming the mean probability of detection will be similar during our study (0.31), the probability that a drumming grouse that is present within 175 m of our transects will be detected at least once during 8 surveys will be approximately 0.95.

We measured characteristics of ruffed grouse habitat at 3 spatial scales. The smallest scale was the area immediately surrounding drumming locations identified during surveys. Characteristics at this scale were measured in the field. The same variables were measured at an unused but potential drumming structure (e.g., log or stump with no signs of use by grouse) nearest a randomly selected point within 85 m of each used drumming structure. A circle with a radius of 85 m represents the "core area" (2.3 ha) of a male's home range during the 2-month "drumming season" (6.7 ha, Archibald 1975). An 85-m radius ensured that selected unused locations were within the home range, whereas the 146-m radius of the home range would not have. This information was collected for all used drumming structures that fell within 200 m of the transect line.

The next scale will be the forest stand, which may be characterized by forest inventory data, but will also be sampled in the field. The buffered transects will be the sampling unit for the landscape-level questions. Larger spatial scales for analysis (e.g., study area, Ecological Classification System land type association) may be possible by aggregating survey transects. Habitat characteristics at landscape scales will be quantified using the same forest inventory and land use/land cover data we use to identify study areas.

Existing MNDNR annual survey data—We will use existing ruffed grouse survey data, which are counts of drums heard at 10 points along roadside transects that have been surveyed once each year for many years. We will define sample landscapes as the area within 175 m of each transect (i.e., to be more conservative about detection distance, given that each transect is surveyed only once each year) and seek existing Geographic Information System (GIS) data that represent land use and land cover information that may be related to ruffed grouse habitat quality. We may randomly select a subsample of roadside landscapes to ground-truth remotely sensed data or digitize important features from aerial photos. We will quantify variables associated with ruffed grouse habitat in each roadside landscape using a GIS. We will select for analysis only drum count data collected within 2 years of when the landscape imagery was captured (i.e., 5 years total).

Data Analysis

New field data—We will conduct a separate analysis at each spatial scale of interest. At the scale of specific drumming locations the analysis will follow a case-control logistic regression design in which the response variable is whether the point was used or not used (Keating and Cherry 2004). This may reveal selection for characteristics of drumming locations, given the constraint of occupying a limited home range. At all larger spatial scales we will use regression analyses in which the response variable is the density of drumming males per hectare. For all analyses we will define *a priori* models consisting of explanatory variables that represent hypothesized habitat relationships. We will use information-theoretic model selection procedures and consider multimodel inference (e.g., Burnham and Anderson 2002).

Existing MNDNR annual survey data—Annual drum counts are associated with specific points along each roadside transect. However, in most cases, much uncertainty exists about the locations of the points, because the locations may not be documented and observers may not stop at exactly the same points each year. Therefore, we will use the entire transect, rather than survey points as the sampling unit. We will sum the counts from all survey points on each transect for each annual survey. There may be much interannual variation in counts along a transect that is not associated with either habitat quality or the long-term grouse population cycle, so we will use the mean of 5 consecutive annual sums, rather than counts from a single survey, as an indication of the relative quality of grouse habitat along each transect. We will use the 5-year mean of annual counts as the response variable in regression models. Landscape metrics will be used in various combinations that represent our *a priori* hypotheses about ruffed grouse habitat relationships. We will use information-theoretic model selection procedures and consider multimodel inference (e.g., Burnham and Anderson 2002).

RESULTS

We collected data about ruffed grouse drumming structures during 2 spring field seasons. During 2009, we located 454 used drumming structures within 200 m of survey transects, and we sampled vegetation characteristics at 434 of them and at 434 nearby unused structures. During 2010, we located 449 individual drumming structures and sampled vegetation characteristics at all of them and at 449 nearby unused structures. Data collection is complete, and we have begun analyzing the data. Research results will be available beginning in fall 2011.

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LITERATURE CITED

Archibald, H. L. 1975. Temporal patterns of spring space use by ruffed grouse. Journal of Wildlife Management 39:472–481.

Burnham, K. P., and D. R. Anderson. 2002. Model selection and multimodel inference: a practical information-theoretic approach. Second edition. Springer-Verlag, New York, New York.

Devers, P. K., D. F. Stauffer, G. W. Norman, D. E. Steffen, D. M. Whitaker, J. D. Sole, T. J. Allen, S. L. Bittner, D. A. Buehler, J. W. Edwards, D. E. Figert, S. T. Friedhoff, W. W. Giuliano, C. A. Harper, W. K. Igo, R. L. Kirkpatrick, M. H. Seamster, J. Spiker, A. Harry, D. A. Swanson, and B. C. Tefft. 2007. Ruffed grouse population ecology in the Appalachian Region. Wildlife Monographs 168.

Gullion, G. W. 1984. Managing northern forests for wildlife. Ruffed Grouse Society, Coraopolis, Pennsylvania.

Gullion, G. W. 1990. Ruffed grouse use of conifer plantations. Wildlife Society Bulletin 18:183–187.

Gullion, G. W, and A. A. Alm. 1983. Forest management and ruffed grouse populations in a Minnesota coniferous forest. Journal of Forestry 81:529–536.

Keating, K. A., and S. Cherry. 2004. Use and interpretation of logistic regression in habitat-selection studies. Journal of Wildlife Management 68:774–789.

MNDNR. 2001. Issues, strategies, desired future forest composition, and stand-selection criteria. Border Lakes Subsection Forest Resource Management Plan. St. Paul, Minnesota.

MNDNR. 2003. Issues, strategies, desired future forest composition, and stand-selection criteria. Mille Lacs Uplands Subsection Forest Resource Management Plan. St. Paul, Minnesota.

MNDNR. 2004 (revised 2006). Strategic direction document. North Shore area Subsection Forest Resource Management Plan. St. Paul, Minnesota.

MNDNR. 2007. A strategic conservation agenda 2003-2007: measuring progress toward mission. Minnesota Department of Natural Resources, St. Paul.

Rusch, D. H., S. DeStefano, M. C. Reynolds, and D. Lauten. 2000. Ruffed grouse. Number 515 *in* A. F. Poole and F. B. Gill, eds. The birds of North America. Academy of Natural Sciences and American Ornithologists' Union, Philadelphia, Pennsylvania.

USFS. 2004*a*. Land and resource management plan: Chippewa National Forest. USDA Forest Service, Milwaukee, Wisconsin.

USFS. 2004*b*. Land and resource management plan: Superior National Forest. USDA Forest Service, Milwaukee, Wisconsin.

Zimmerman, G. S. 2006. Habitat and population regulation of ruffed grouse in Minnesota. Ph.D. dissertation, University of Minnesota, St. Paul.

HABITAT SELECTION OF SPRUCE GROUSE AT MULTIPLE SPATIAL SCALES IN NORTHWESTERN MINNESOTA

Michael A. Larson and J. Wesley Bailey

SUMMARY OF FINDINGS

During July and August of 2010, we attempted to locate spruce grouse (*Falcipennis canadensis canadensis*) hens and broods by surveying likely habitat and broadcasting a chick call. We detected a few spruce grouse, but determined that the survey method would not be successful for studying habitat selection by broods.

The current study focuses on habitat selection by adult spruce grouse during spring. During spring of 2011, we surveyed at least 38 plots centered on stands of jack pine (*Pinus banksiana*) and 18 plots centered on stands of lowland black spruce (*Picea mariana*). We detected 44 spruce grouse in those plots; approximately half were males and half were females. The 2011 field season was not complete when this summary was written, and we plan to continue collecting data during the spring of 2012.

INTRODUCTION

The spruce grouse is listed as a Species of Greatest Conservation Need (SGCN) by the Minnesota Department of Natural Resources (MNDNR), which cited its dependence on a potentially vulnerable habitat type (i.e., boreal forest) and a lack of population trend data (MNDNR 2006). Due to the unknown or tenuous status of spruce grouse along the southern edge of its range and the existence of several threats to the viability of their populations, there is interest in learning more about their status and ecology. The Association of Fish and Wildlife Agencies (Williamson et al. 2008) recommended developing formal surveys for monitoring population change and conducting research on the impacts of habitat change and hunting on spruce grouse.

All 3 of the previous studies of spruce grouse in Minnesota were conducted in rather unique study areas (i.e., either entirely black spruce lowlands or primarily peatlands; Anderson 1973, Haas 1974, Pietz and Tester 1979), so it is difficult to apply their results broadly. They were similar to other habitat studies in focusing on the associations between the presence of spruce grouse and certain forest cover types and traditional metrics of forest structure (e.g., tree density and height). However, these studies did not address some of the important questions that are relevant to how we currently manage forests. For example, we do not know whether the density and species of residual trees (i.e., those left after logging) are important, what size and shape of forest stands are optimum, what proportions of different cover types on a landscape are best, or if spatial juxtaposition of cover types or other habitat characteristics are important. In short, wildlife managers want to know how to manage forests with suitable cover types for the most benefit to spruce grouse.

The habitat needs or preferences of spruce grouse may vary by seasonally (Stenlund and Magnus 1951, Pietz and Tester 1982). However, the only time of year when there is a reasonable chance of detecting them is during spring when males display with a flutter flight (Keppie 1992). Investigating habitat selection during other times of year will require radio-tracking. We attempted to survey hens and broods during summer 2010 using a chick call, but we were unsuccessful, so studying habitat selection by broods also will require radio-tacking. This study will focus on the spring display period, so we can learn about habitat selection with minimal costs using auditory surveys, while also learning basic information about the densities of spruce grouse and the feasibility of encountering a sufficient number of grouse for a potential radio-tracking study in the future.

OBJECTIVES

1. To estimate parameters in regression models relating habitat characteristics at the landscape scale to the density of spruce grouse during spring.
2. To test whether specific measures of the structure of forest stands that were associated with the presence of spruce grouse during previous studies are associated with the presence of spruce grouse during spring in a new data set.
3. To estimate parameters in capture–recapture models of time-of-detection data (Alldredge et al. 2007) for estimating the probability of detection and densities of spruce grouse.

STUDY AREA

The study will be conducted in Red Lake WMA and adjacent portions of Beltrami Island State Forest, which are in Lake of the Woods, Beltrami, and Roseau counties in northwestern Minnesota (Figure 1). Coniferous forests and mixed coniferous-deciduous forests cover much of the study area, and spruce grouse are often seen and hunted there.

METHODS

Spatial Sampling Design

We are interested in landscapes containing 1 of 2 focal cover types associated with spruce grouse—jack pine (JP) and black spruce (BS). Spruce grouse also may be found in stands of tamarack, balsam fir, and white cedar, but the associations with those cover types are not as strong. To further reduce variation from sources in which we are not interested, we will screen the population of landscapes from which we will draw a sample. The population of landscapes will include square plots of 64 ha (i.e., 800 m x 800 m) containing near the center a forest stand that meets the following criteria: (1) the cover type is JP or lowland BS, (2) it is not classified as stagnant (i.e., not commercially productive), (3) it is 4–16 ha in size, (4) it is within 1.6 km (1 mile) of an accessible road in the study area, and (4) it is on public land. Furthermore, if the cover type is JP, the stand will be 11–50 years old (Szuba and Bendell 1983: 203) and the site index (i.e., a relative measure of productivity for growing trees based on soil type, hydrology, and other factors) will be 45–65; and if the cover type is BS, the stand will be 19–120 years old (Szuba and Bendell 1983: 203, Lycke et al. 2011) and the site index will be 25–50.

We will draw a random sample of points from the area defined by all forest stands meeting the criteria listed above. Each stand containing a random point (i.e., a focal stand) will be the center of a landscape that will be sampled. From a grid of points regularly spaced 200 m apart across the study area we will select the ones nearest the centroid of the randomly selected focal stands. Each of these focal grid points will be the northeastern 1 of 4 grid points in the middle of a 4- × 4-point grid that will define a plot (Figure 2). Circles of 100-m radius from the 16 grid points will completely fill the 64-ha plot without any overlap.

Estimating Grouse Density

In Koochiching County during the early-1970s the breeding display period of spruce grouse lasted 26 days, from within 2 days of snowmelt being complete (e.g., late-April) until late-May, with less intense display continuing until mid-June (Anderson 1973: 27-28, 97, 105). However, more recently in Wisconsin (WI), most spruce grouse quit responding to the cantus by early May (Nick Anich, WIDNR, personal communication). The flutter flight displays of males are detectable from at least 50 m away and perhaps up to 100 m away (Keppie 1992, Worland et al. 2009). The cantus, or song, of females is not performed as regularly as the flight displays of males, but females may be recorded during surveys of spruce grouse (Worland et al. 2009).

Flight displays and the cantus may be elicited by broadcasting a recording of the female cantus (MacDonald 1968). Although unsolicited breeding displays may occur primarily or only during a few hours near sunrise (Keppie 1992), broadcasting a call may be "effective in locating breeding males throughout the day" (Bouta 1991: 6).

We will visit each randomly selected 64-ha landscape plot once, beginning 60 minutes before sunrise (Keppie 1992:309), and survey for spruce grouse for 8 minutes at each of the 16 regularly spaced grid points. Each survey will be divided into 4 intervals of 2 minutes each. Detections of spruce grouse, including the direction and estimated distance, will be recorded separately for each interval and each grouse. During the last interval we will broadcast a recording of the female cantus for 30-second bursts spaced 30 seconds apart (Jakob et al. 2010). The first 3 intervals will consist of silent observation, because we want to estimate probabilities of detection based on spontaneous grouse behaviors, rather than just behaviors that are elicited.

We will use time-of-detection methods (Alldredge et al. 2007) to estimate probabilities of detection and densities of spruce grouse. The methods involve treating each 2-minute survey interval as a separate occasion, so each grouse detected during an 8-minute survey has a 4-digit detection history consisting of 1s (i.e., detected during the interval) and 0s (i.e., not detected). The data are analyzed in a capture–recapture framework in which we will estimate the potential effects of detection distance, broadcasting the cantus, observer behavior after first detecting a grouse at a survey point (Riddle et al. 2010), wind speed, presence of precipitation, time of day, day of the year, sex of the bird, and forest cover type on probabilities of detection. We will also estimate grouse abundance with these data and calculate grouse density by applying a fixed-distance radius to define an area in which probabilities of detection may be assumed to be similar (e.g., 50 m). However, reasonable precision of estimates of density may require sample sizes greater than we will generate during this study. If so, we will use plot-specific probabilities of detection or simply the number of spruce grouse detections as the response variable in the landscape-scale habitat analysis (see **Data Analysis** below).

Habitat Characteristics and Vegetation Sampling

Characteristics of the landscape (i.e., plot) that are potentially related to spruce grouse habitat will be collected from relevant GIS databases. Such characteristics of landscapes include the proportions, juxtaposition, and diversity of different cover types; densities of roads and other edges between cover types; and presence or proportions of native plant community types. Many habitat characteristics at the scale of the forest stand, including age, species composition, size, shape, and management history, will be collected from forest inventory data. However, previous studies have indicated that certain structural characteristics of forest stands that are not available from inventory data also may affect use by spruce grouse. Therefore, we will measure those characteristics in the focal stands (i.e., the randomly selected stands around which the landscape plots were placed).

We will measure vegetation characteristics at the focal grid point (i.e., the point nearest the centroid of the focal stand) for use in addressing Objective 2. We will measure height to the top and bottom of the live forest overstory canopy 10 m away in the 4 cardinal compass directions. We will estimate densities of overstory stems (i.e., >3 m tall) and understory shrubs (i.e., 0.2–3 m tall) using the point-center-quarter method (Higgins et al. 2005: 531). We will estimate vertical cover above 1.5 m with a densiometer. We will confirm the 1–3 most dominant tree species from forest inventory data and record the predominant ground cover as 1 of 10 standard categories from MNDNR Forestry's CSA Tatum Guide. To account for spatial variation within the stand, we will repeat the point-center-quarter and vertical cover measurements at 2 additional vegetation sampling points 30 m apart along a transect toward the center of the stand. The compass bearing for the transect will be determined from maps before going afield. If the center of the stand is < 60 m from the focal grid point, we will use a randomly selected compass bearing for the transect and place the 2 additional vegetation sampling points 30 m on either side of the focal grid point.

Data Analysis

To address the objectives about habitat relationships, we will specify *a priori*, fit, and rank regression models representing our hypotheses about which combinations of habitat characteristics are most highly correlated with space use by spruce grouse during spring (Burnham and Anderson 2002). We will restrict the number of variables per model to 1/20th of the sample size for the analysis. To help limit the number of variables under consideration, we will focus on variables with the strongest purported influence in the literature, a wide range of observed values, and low correlation with other variables, and we will consider combining similar variables into indices.

For the landscape scale analysis the response variable will be density of spruce grouse, plot-specific probability of detection, or number of grouse detections, depending upon whether or not we attain sufficient precision when estimating density or probability of detection. For stand scale analyses the response variable will be presence or absence of spruce grouse. For Objective 2, the measures of forest structure identified as important during previous studies were tree density and tree height, which are data we will have for only the focal stands. In addition to regression analysis of the sample of focal stands with more detailed vegetation measurements, we will summarize grouse detection data by known characteristics of all stands in the sampled landscapes (e.g., proportions of points with a grouse detection in different overstory cover types).

RESULTS AND DISCUSSION

During spring of 2011, we surveyed at least 38 plots centered on stands of jack pine and 18 plots centered on stands of lowland black spruce. We detected 44 spruce grouse in those plots; approximately half were males and half were females. The 2011 field season was not complete when this summary was written, and we plan to continue collecting data during the spring of 2012.

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LITERATURE CITED

Alldredge, M. W., K. H. Pollock, T. R. Simons, J. A. Collazo, and S. A. Shriner. 2007. Time-of-detection method for estimating abundance from point-count surveys. Auk 124:653–664.

Anderson, L. J. 1973. Habitat use, behavior, territoriality, and movements of the male Canada spruce grouse of northern Minnesota. M.S. thesis, University of Minnesota, St. Paul.

Bouta, R. P. 1991. Population status, historical decline, and habitat relationships of spruce grouse in the Adirondacks of New York. M.S. thesis, State University of New York, Syracuse.

Burnham, K. P., and D. R. Anderson. 2002. Model selection and multimodel inference: a practical information-theoretic approach. Second edition. Springer-Verlag, New York, New York.

MNDNR. 2006. Tomorrow's habitat for the wild and rare: an action plan for Minnesota wildlife. Division of Ecological Services, St. Paul.

Haas, G. H. 1974. Habitat selection, reproduction, and movements in female spruce grouse. Ph.D. dissertation, University of Minnesota, St. Paul.

Higgins, K. F., K. J. Jenkins, G. K. Clambey, D. W. Uresk, D. E. Naugle, J. E. Norland, and W. T. Barker. 2005. Vegetation sampling and measurement. Pages 524–553 *in* C. E. Braun, ed. Techniques for wildlife investigations and management. The Wildlife Society, Bethesda, Maryland.

Jakob, C., F. Ponce-Boutin, A. Besnard, and C. Eraud. 2010. On the efficiency of using song playback during call count surveys of red-legged partridges (*Alectoris rufa*). European Journal of Wildlife Research 56:907–913.

Keppie, D. M. 1992. An audio index for male spruce grouse. Canadian Journal of Zoology 70:307–313.

Lycke, A., L. Imbeau, and P. Drapeau. 2011. Effects of commercial thinning on site occupancy and habitat use by spruce grouse in boreal Quebec. Canadian Journal of Forest Research 41:501–508.

MacDonald, S. D. 1968. The courtship and territorial behavior of Franklin's race of the spruce grouse. Living Bird 7:5–25.

Pietz, P. J., and J. R. Tester. 1979. Utilization of Minnesota peatland habitats by snowshoe hare, white-tailed deer, spruce grouse, and ruffed grouse. Minnesota Department of Natural Resources, St. Paul.

_____. 1982. Habitat selection by sympatric spruce and ruffed grouse in north central Minnesota. Journal of Wildlife Management 46:391–403.

Riddle, J. D., R. S. Mordecai, K. H. Pollock, and T. R. Simons. 2010. Effects of prior detections on estimates of detection probability, abundance, and occupancy. Auk 127:94–99.

Stenlund, M. H., and L. T. Magnus. 1951. The spruce hen comes back. Conservation Volunteer 14:20–24.

Szuba, K. J., and J. F. Bendell. 1983. Population densities and habitats of spruce grouse in Ontario. Pages 199–213 in R. W. Wein, R. R. Riewe, and I. R. Methven, eds. Resources and dynamics of the boreal zone: proceedings of a conference held at Thunder Bay, Ontario, August 1982. Association of Canadian Universities for Northern Studies, Ottawa, Ontario, Canada.

Williamson, S. J., D. M. Keppie, R. Davison, D. Budeau, S. Carrière, D. Rabe, and M. A. Schroeder. 2008. Spruce grouse continental conservation plan. Association of Fish and Wildlife Agencies, Washington, DC.

Worland, M., K. J. Martin, and L. Gregg. 2009. Spruce grouse distribution and habitat relationships in Wisconsin. Passenger Pigeon 71:5–18.

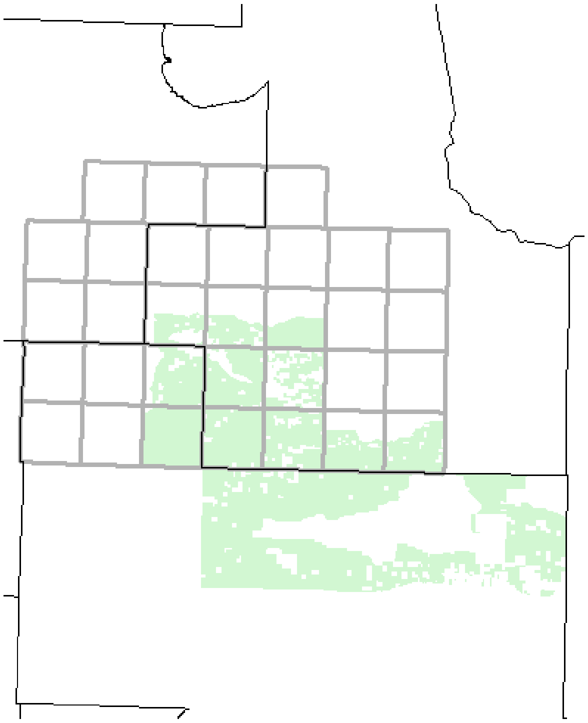

Figure 1. Study area (gray township lines) for spruce grouse research relative to Red Lake Wildlife Management Area (shaded area) and county boundaries (black lines) in northwestern Minnesota.

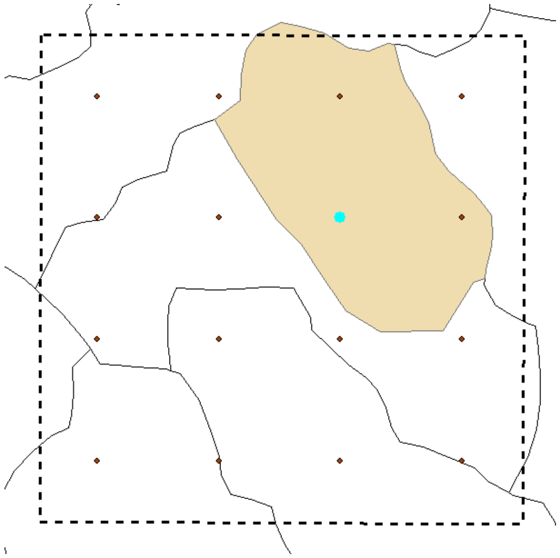

Figure 2. Diagram of a 64-ha spruce grouse survey plot (i.e., landscape; thick dashed line) showing the focal stand (shaded area), other forest stands (solid lines), the grid of 16 survey points spaced 200 m apart, and the survey point nearest the centroid of the focal stand (large blue point).

LANDSCAPE CHARACTERISTICS ASSOCIATED WITH DANCING GROUNDS OF SHARP-TAILED GROUSE

Michael A. Larson and J. Wesley Bailey

SUMMARY OF FINDINGS

We are developing a habitat model to describe the landscape characteristics associated with "dancing grounds" of sharp-tailed grouse (*Tympanuchus phasianellus campestris*) across their range in Minnesota. We provide preliminary results, including a table and figure, but our analyses are not complete and the results are subject to potential revision.

INTRODUCTION

Sharp-tailed grouse in Minnesota occur in open landscapes of "grass, brush, savanna, and boreal peatland," which "are sometimes associated with small grain and livestock farming" (Berg 1997:1, 4). Although sharp-tailed grouse habitat was widely distributed in Minnesota during the early and mid-1900s, the range of sharp-tailed grouse is now limited to areas in the northwest and east-central portions of the state (Figure 1). The succession and conversion of their habitat to unsuitable cover types coincided with a dramatic decline in estimates of annual harvest by hunters from 120,000 sharp-tailed grouse in 1952 to 4,000 in 1965 (Landwehr 1984). Since 1980 the average number of grouse per dancing ground during spring has fluctuated between 7 and 13 and has had a slightly positive trend (Larson 2009), whereas harvest has exhibited a noticeable negative trend, ending with harvests of 6,000–16,000 birds/year during the last decade (Dexter 2009).

To benefit sharp-tailed grouse and other wildlife, the Minnesota Department of Natural Resources' (MNDNR) Section of Wildlife has emphasized the management and restoration of targeted open lands within the forested part of the state. These efforts include designating priority open landscapes within the Subsection Forest Resources Management Plan process and spending more money on openland/brushland management than any other habitat improvement activity in the forested regions of the state. However, identifying landscapes to target with openland management is challenging.

Although Solberg (1999) attempted to identify priority areas for sharp-tailed grouse management using maps and landscape characteristics, Hanowski et al. (2000) were the first to quantify the habitat characteristics of dancing grounds at the landscape scale. Both studies focused on sharp-tailed grouse range in east-central Minnesota and provided valuable information. We were interested in quantifying variations in landscape characteristics associated with dancing grounds across their full geographic range in Minnesota. Our goal was to develop a spatially explicit habitat model for identifying priority areas for sharp-tailed grouse management, including habitat improvement, land acquisition, population monitoring, and potential reintroduction.

OBJECTIVES

1. To determine which landscape characteristics are most highly correlated with the presence of dancing grounds of sharp-tailed grouse in Minnesota.
2. To map variations in the quality of habitat for sharp-tailed grouse dancing grounds throughout their range in Minnesota.

STUDY AREA

We defined the study area as occurring within both of 2 different boundaries for describing the geographic extent of sharp-tailed grouse range in Minnesota (Figure 1). One boundary encompassed the subsections of Minnesota's Ecological Classification System (ECS, following Cleland et al. 1997), where dancing grounds were observed during 1991–1993. The sample of dancing ground locations that we used is described and justified in the Methods section below. The other boundary was the 85% kernel density estimate around observed dancing grounds. We selected the 85% kernel boundary, because it encompassed 21% less area than the 95% kernel boundary and excluded only 1% of the used sites. The 80% kernel boundary encompassed 32% less area than the 95% kernel boundary, but we thought it excluded too many used sites (5%).

METHODS

We investigated habitat selection of sharp-tailed grouse for dancing grounds in Minnesota by comparing the attributes of a sample of locations known to have been used as dancing grounds (i.e., used sites) and an independent sample of locations that were representative of areas available for use as dancing grounds (i.e., available sites).

Use-availability Data

Used sites were detected during annual surveys conducted by the MNDNR during spring of each year (see Larson 2008 for survey methods). Although the spatial sampling design of the survey was haphazard, the spatial extent of the survey covered the known range of the species in Minnesota, and we think the probability of detecting an existing dancing ground in a given year was >0.3 (M. A. Larson, unpublished data). The sample of used sites consisted of locations where a dancing ground was observed at least once during 1991–1993, because that was the time interval during which the land cover imagery was captured (see Landscape Data below). Each used site was included in the data set only once, and locations were precise to the quarter-section of the Public Land Survey.

We selected the sample of available sites from the spatial extent defined in the Study Area section above. The only other constraint we applied for the area from which available sites were randomly selected was that the forest and non-habitat cover types (defined below) were excluded. The definition of the study area, or spatial extent, is important for use-availability comparisons (Johnson 1980). Using a more restrictive study area (e.g., within a limited-distance buffer of known dancing grounds) would lead to inferences focusing on specific characteristics of patches of open cover types (e.g., area, edge density). Using a broader extent for the study area (e.g., all of northern Minnesota) likely would lead to inferences emphasizing the importance of open lands in general. We sought a balance between those extremes.

Landscape Data

We created for the study area a Geographic Information System (GIS) data layer consisting of cover types relevant to sharp-tailed grouse habitat. We started with Level 4 classes of land use/land cover from the Minnesota Gap Analysis Project (MN-GAP, MNDNR 2001) and reclassified them to the following 8 cover types: cropland, disturbed grass (grassland and prairie cover types on non-public lands), undisturbed grass (grassland and prairie cover types on public lands), sedge meadow, shrub (lowland deciduous shrub), bog (lowland evergreen shrub, stagnant black spruce [*Picea mariana*], and stagnant tamarack [*Larix laricina*]), forest (all other MN-GAP Level 4 forest classes, including upland shrub, which is primarily post-harvest regeneration), and non-habitat (all other MN-GAP Level 4 classes).

Then we superimposed (i.e., replaced the MN-GAP data with) data from better sources for 3 of the cover types. Using the National Wetlands Inventory (NWI, Cowardin et al. 1979, Minnesota Land Management Information Center 2007), we selected scrub-shrub (broad-leaved deciduous and deciduous) and persistent emergent types that occurred within flooded, saturated, and seasonally flooded NWI water regime modifiers. We added the NWI scrub-shrub areas to our shrub cover type and the persistent emergent areas to our sedge meadow cover type, regardless of classification by the MN-GAP. Then we added areas with herbaceous vegetation cover practices from the 1997 Conservation Reserve Program (CRP, Minnesota Natural Resources Conservation Service 2010) to our undisturbed grass cover type, regardless of the MN-GAP or NWI classifications.

Our land cover layer is a raster (ESRI) grid in Universal Transverse Mercator (UTM) Zone 15 (NAD 83) with a cell size of 30 m x 30 m. We used ArcGIS 9.3.1 to calculate landscape metrics for areas within 4 different buffer distances of each used and available point (i.e., 400, 800, 1,600, and 3,200 m). We considered a total of 19 variables for inclusion in our models (Table 1). To preclude potential computational problems caused by large values, we normalized the values of all covariates (i.e., [ ]/SD[ ]) before fitting the models.

Model Set

Correlations between values from different spatial scales for the same variable were very high for most variables, so we decided to use only the 800-m scale for our *a priori* models. That spatial scale was similar to those at which Hanowski et al. (2000) found that characteristics differed most between active and inactive leks (i.e., 500 and 1,000 m). We also considered Simpson's Evenness Index, but its values were highly correlated with values of Simpson's Diversity Index, so we retained only the latter, because it accounted for the number of cover types, as well as the evenness among the areas of the different cover types (McGarigal et al. 2002).

We used different combinations of the variables to define 73 *a priori* models (Table 2). Thirty, 10, 9, 10, 2, 4, 2, 3, and 2 of the models had 3, 4, 5, 6, 7, 8, 9, 10, and 11 parameters, respectively, including an intercept term. The 73rd model was the global model, which had 19 parameters, because it did not include the OPEN covariate, which was the sum of the area in 3 other cover types that were included in the global model. Several of the models were formulated to be similar to the best models of Hanowski et al. (2000) and Niemuth and Boyce (2004). This is a relatively large set of *a priori* models, because there are relatively few previous studies and there is still much uncertainty about the importance of different landscape characteristics.

With a second phase of analysis we are investigating models that contain variables measured at different spatial scales, not just the 800-m scale. During future phases of analysis we plan to investigate separate models for the eastern and northwestern portions of the study area, which differ in their proportions of different land-use/land-cover categories (Figure 1).

Model Fitting

The most appropriate way to analyze and interpret data from a use-availability study design is still debated in the literature (Keating and Cherry 2004, Johnson et al. 2006). We found the approach advocated by Lele and Keim (2006), which is a form of logistic regression, to be the most appealing, because it addressed potential concerns about logistic regression that were raised by Keating and Cherry (2004), and the concept of weighted distributions upon which it is based is more intuitive than alternative approaches to the analysis. We fit our models using scripts for programs R and WinBUGS provided by S. Lele (personal communication), which were based on partial likelihood and data cloning methods described by Lele (2009). We used AIC values to rank the *a priori* models based on how well they fit the data. We assessed the fit of the global model to the data using *k*-fold cross validation (Johnson et al. 2006).

RESULTS AND DISCUSSION

We used 1,245 randomly selected available sites and 249 used sites in our analyses. Our sample of used sites excluded 3 of the 252 dancing grounds observed at least once during 1991–1993, because they were outside the 85% kernel boundary (Figure 1). Both methods used to generate initial values resulted in the same AIC rankings for the best 5 models, which had 9–19 parameters. The global model was the best-fitting model in the *a priori* set, because it was the only model that contained all 7 of the parameters that were most highly correlated with the presence of a dancing ground (i.e., had estimates whose confidence intervals did not include 0; Table 2, Figure 2). We did not include distance to road (DIRD) in Figure 2, because that parameter confirmed that dancing grounds closer to roads were more likely to be documented during surveys rather than providing insight about habitat selection. The second best model ($K = 12$, $\Delta AICc = 1{,}311$) included DILK, SHRB, RDDN, FRST, and OPEN, but not SIMP or DIRD. Looking at the best model with a given number of parameters for models with 3–8 parameters ($n = 6$ models), the distance to nearest lek (DILK) variable occurred in all of them and the area of the shrub cover type (SHRB) occurred in 4 of them.

Results are preliminary and are subject to revision based on continuing work on this project. When our analyses are complete we will compare results to those of Hanowski et al. (2000) and Niemuth and Boyce (2004), who have developed similar models of landscape characteristics associated with the dancing grounds of sharp-tailed grouse.

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LITERATURE CITED

Berg, W. E. 1997. The sharp-tailed grouse in Minnesota. Minnesota Department of Natural Resources, Wildlife Populations and Research Unit, Wildlife Report Number 10, St. Paul.

Cleland, D. T., P. E. Avers, W. H. McNab, M. E. Jensen, R. G. Bailey, T. King, and W. E. Russell. 1997. National hierarchical framework of ecological units. Pages 181–200 *in* M. S. Boyce and A. Haney, eds. Ecosystem management applications for sustainable forest wildlife resources. Yale University Press, New Haven, Connecticut.

Cowardin, L. M., V. Carter, F. Golet, and E. LaRoe. 1979. Classification of wetlands and deepwater habitats of the United States. U. S. Fish and Wildlife Service, Washington, D.C.

Dexter, M. H. 2009. 2008 small game hunter mail survey. Pages 157–171 *in* M. H. Dexter, ed. Status of wildlife populations, fall 2009. Division of Fish and Wildlife, Minnesota Department of Natural Resources, St. Paul.

Hanowski, J. M., D. P. Christian, and G. J. Niemi. 2000. Landscape requirements of prairie sharp-tailed grouse *Tympanuchus phasianellus campestris* in Minnesota. Wildlife Biology 6:301–307.

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

Johnson, C. J., S. E. Nielsen, E. H. Merrill, T. L. McDonald, and M. S. Boyce. 2006. Resource selection functions based on use-availability data: theoretical motivation and evaluation methods. Journal of Wildlife Management 70:347–357.

Keating, K. A., and S. Cherry. 2004. Use and interpretation of logistic regression in habitat-selection studies. Journal of Wildlife Management 68:774–789.

Landwehr, T. J. 1984. A summary of Minnesota game regulations, harvest, and population statistics, 1858–1983. Minnesota Department of Natural Resources, St. Paul.

Larson, M. A. 2008. Grouse surveys in Minnesota during spring 2008. Pages 55–63 *in* M. H. Dexter, ed. Status of wildlife populations, fall 2008. Division of Fish and Wildlife, Minnesota Department of Natural Resources, St. Paul.

Larson, M. A. 2009. Grouse surveys in Minnesota during spring 2009. Pages 73–81 *in* M. H. Dexter, ed. Status of wildlife populations, fall 2009. Division of Fish and Wildlife, Minnesota Department of Natural Resources, St. Paul.

Lele, S. R. 2009. A new method for estimation of resource selection probability function. Journal of Wildlife Management 73:122–127.

Lele, S. R., and J. L. Keim. 2006. Weighted distributions and estimation of resource selection probability functions. Ecology 87:3021–3028.

McGarigal, K., S. A. Cushman, M. C. Neel, and E. Ene. 2002. FRAGSTATS: spatial pattern analysis program for categorical maps. Computer software, University of Massachusetts, Amherst. URL: www.umass.edu/landeco/research/fragstats/fragstats.html

Minnesota Land Management Information Center. 2007. National wetlands inventory, Minnesota: metadata summary web page. URL: http://www.mngeo.state.mn.us/chouse/metadata/nwi.html

Minnesota Natural Resources Conservation Service. 2010. CRP job sheets web page. URL: http://www.mn.nrcs.usda.gov/technical/ecs/agron/crp/planform.html

MNDNR. 2001. Full GIS metadata record for GAP land cover - raster web page. URL: http://jmaps.dnr.state.mn.us/mdreporter/dp_full_record.jsp?mpid=25000015&ptid=6&fcid=6&dsid=31

Niemuth, N. D., and M. S. Boyce. 2004. Influence of landscape composition on sharp-tailed grouse lek location and attendance in Wisconsin pine barrens. Ecoscience 11:209–217.

Solberg, K. 1999. Savannas suitable for restoration and management for sharp-tailed grouse. Final Technical Report for LCMR Project M32, Minnesota Department of Natural Resources, St. Paul.

Table 1. Variables considered in models for distinguishing sites used and available for dancing grounds of sharp-tailed grouse in Minnesota during 1991–1993.

Number	Name	Description
1	GRSU	Area in the undisturbed grass cover type
2	GRSD	Area in the disturbed grass cover type
3	SEDG	Area in the sedge meadow cover type
4	OPEN	Area in the undisturbed grass, disturbed grass, and sedge meadow cover types
5	CROP	Area in the crop cover type
6	SHRB	Area in the shrub cover type
7	BOG	Area in the bog cover type
8	FRST	Area in the forest cover type
9	SIMP	Simpson's Diversity Index[a]
10	DILK	Distance to nearest known lek, or dancing ground
11	DIGR	Distance to nearest patch of disturbed grass patch
12	DIFO	Distance to nearest patch of forest
13	DIRD	Distance to nearest road
14	RDDN	Road density
15	EDBS	Distance of edge between the bog and shrub cover types
16	EDBO	Distance of edge between the bog and open cover types
17	EDOF	Distance of edge between the open and forest cover types
18	PAFO	Number of patches in the forest cover type
19	PASH	Number of patches in the shrub cover type

[a] McGarigal et al. (2002).

Table 2. Estimates of parameters for the global logistic regression model of the presence of sharp-tailed dancing grounds in Minnesota.

Variable	Beta	SE	95% LCL[a]	95% UCL[a]
Intercept	-4.13	0.49	-5.09	-3.18
GRSU	0.02	0.27	-0.50	0.54
GRSD	0.52	0.26	0.01	1.03
SEDG	0.14	0.20	-0.25	0.52
CROP	-0.13	0.48	-1.06	0.80
SHRB	-0.84	0.36	-1.55	-0.13
BOG	-0.58	0.39	-1.35	0.20
FRST	-0.89	0.40	-1.67	-0.11
SIMP	0.33	0.15	0.03	0.62
DILK	-1.67	0.25	-2.15	-1.18
DIGR	0.13	0.33	-0.52	0.77
DIFO	0.17	0.17	-0.16	0.50
DIRD	-1.05	0.46	-1.96	-0.14
RDDN	-1.00	0.23	-1.45	-0.55
EDBS	0.27	0.18	-0.07	0.62
EDBO	0.11	0.14	-0.15	0.38
EDOF	-0.22	0.18	-0.58	0.14
PAFO	0.19	0.15	-0.10	0.47
PASH	0.13	0.12	-0.12	0.37

[a]LCL= lower confidence limit and UCL = upper confidence limit.

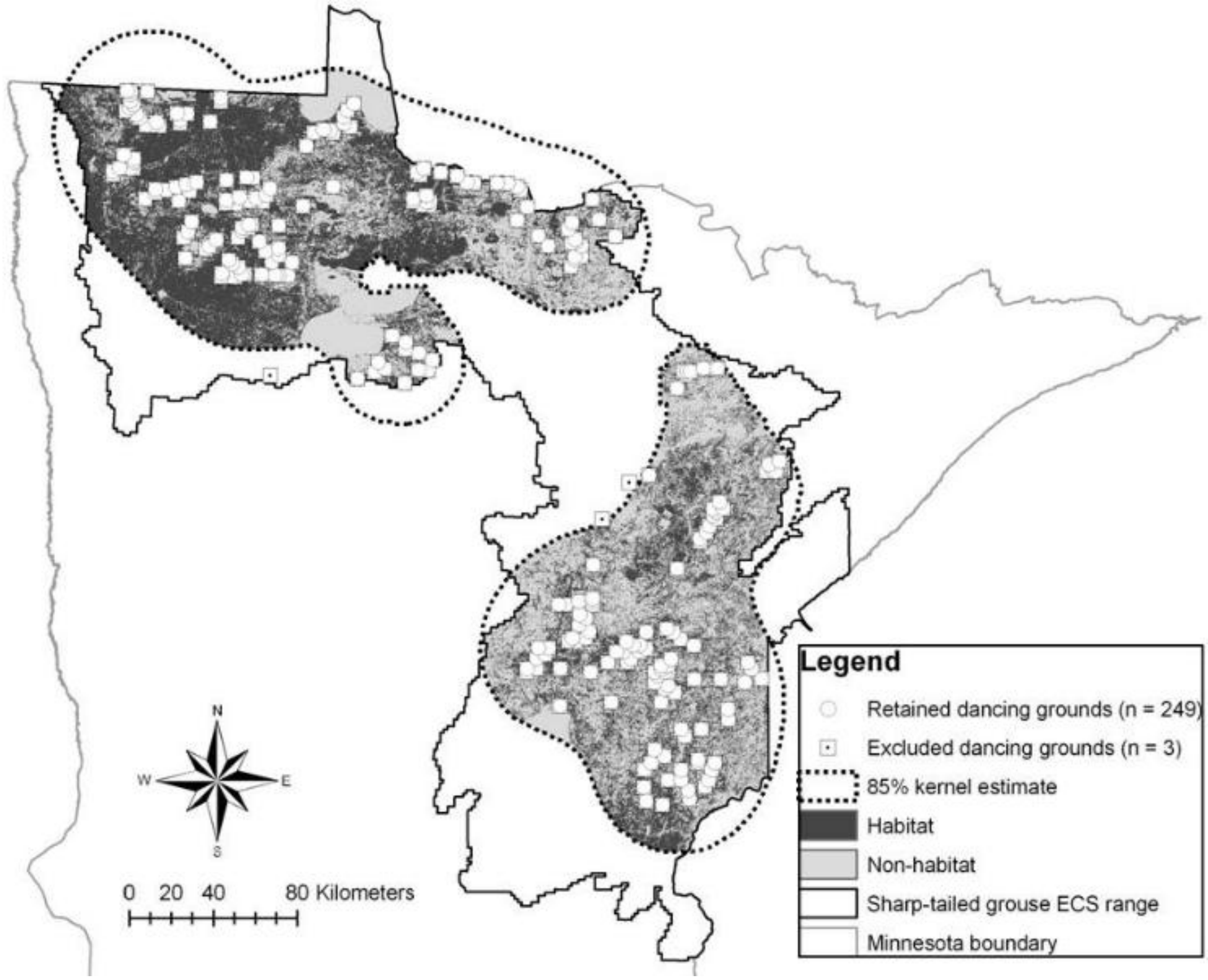

Figure 1. Map of the spatial extent of the habitat selection model for sharp-tailed grouse in northern Minnesota (shaded areas). The extent was defined as occurring within (1) occupied Ecological Classification System subsections and (2) the 85% kernel estimate of space use, based upon the locations of dancing grounds that were documented during 1991–1993.

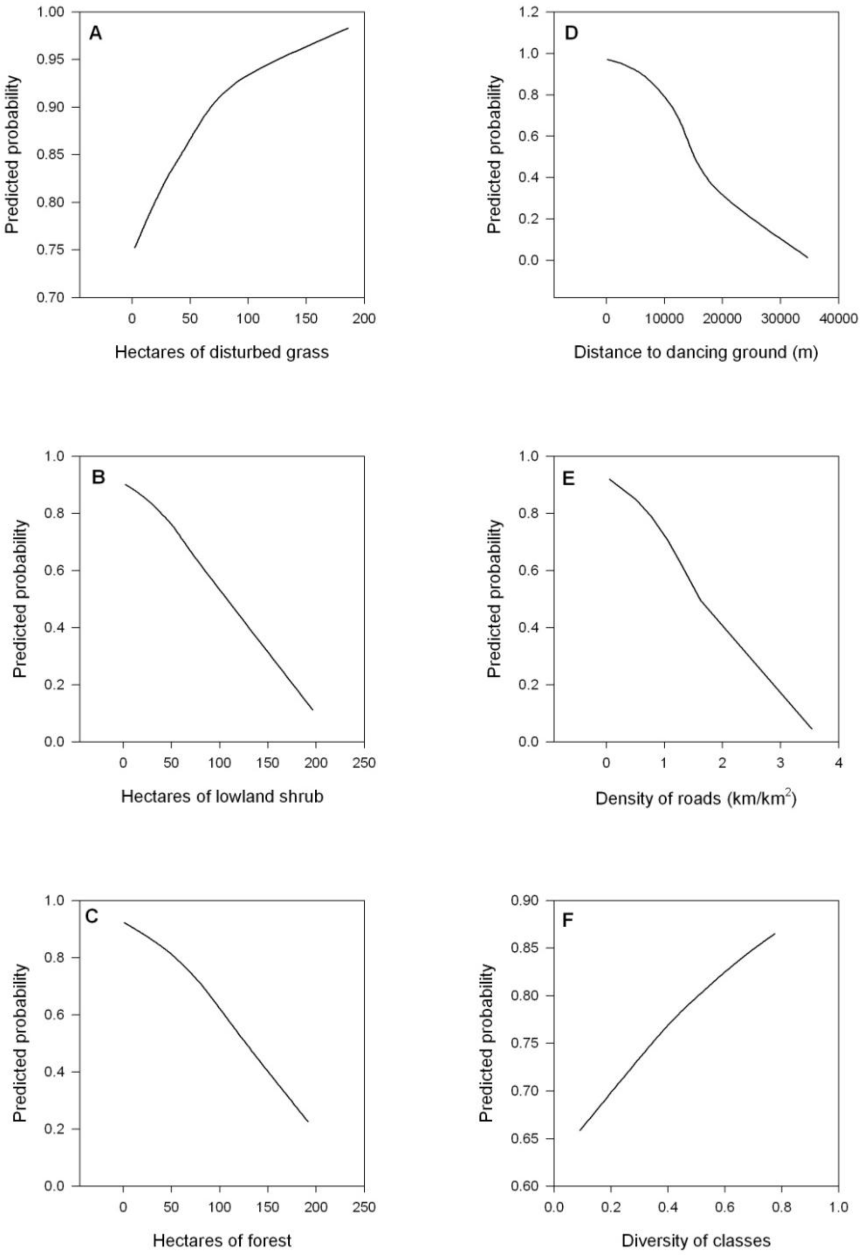

Figure 2. Predicted probabilities of use of dancing grounds of sharp-tailed grouse (points buffered by 800 m, 201-ha circles) relative to landscape characteristics with statistically significant parameter estimates in a use-availability model based on dancing grounds documented during 1991–1993 in Minnesota. The abscissa in panel F is Simpson's Diversity Index based on land cover classes.

Farmland Wildlife Populations and Research Group

Madelia Research Center
35365 – 800th Avenue
Madelia, Minnesota 56062-9744
(507) 642-8478

PILOT SEASON UPDATE: COMPARING COLLECTION METHODS FOR MONITORING PRAIRIE INVERTEBRATE ABUNDANCE AND DIVERSITY

Molly A.Tranel and Daren Carlson

SUMMARY OF FINDINGS

Invertebrates play critical functional roles in the prairie community from pollination to serving as essential food sources for grassland birds and other animals. Numerous trapping techniques exist for monitoring insect communities (Toermaelae 1982, Standen 2000, Schauff 2001, Epsky et al. 2008), but determining which method is most suitable depends on a variety of factors, such as landscape, habitat, and desired insect communities. One goal of this project was to evaluate insect collection methods to estimate diversity and abundance of invertebrates in grassland habitats, and to use the developed protocol to monitor invertebrate communities in both native prairies and planted grasslands. The pilot season of this project was conducted on 2 native prairie sites paired with 2 planted grassland sites located in south-central Minnesota. Sweep sampling was the least time-consuming and easiest sampling method, whereas vacuum sampling was the most physically demanding for this habitat type. Approximately 240 samples have been sorted with an average sorting time of 22 minutes per sample. Hymenoptera was the most common Order found at the 2 sites that have been analyzed. The results from this project will provide information to facilitate more effectively monitoring components of native prairie and surrogate grasslands, and will be used in a larger study to identify grassland management techniques that improve prairie and surrogate grassland habitat for Species of Greatest Conservation Need (SGCN) and other wildlife.

INTRODUCTION

Because many invertebrates are associated with native prairie and play critical functional roles, they have been identified as a key animal group to be monitored (Kremen et al. 1993). Fifteen insect species and 8 spider species, including the red-tailed prairie leafhopper (*Aflexia rubranura*), Dakota skipper (*Hesperia dacotae*), and *Marpissa grata* – a species of jumping spider, are prairie-associated of SGCN. Furthermore, invertebrates are an essential food for grassland birds and their broods (Buchanan et al. 2006). Yet, information on prairie invertebrates and how habitat management techniques may impact their populations is sparse.

Recent acceleration of efforts to maintain or restore prairies have accentuated the need for long term-data collection, storage, and analysis using a consistent set of monitoring protocols to: (1) detect changes and long-term trends (status and trend monitoring) and (2) evaluate the success of prairie management and restoration activities (effectiveness monitoring). Estimates of invertebrate diversity and abundance are the best measures of habitat quality for prairie invertebrates. In addition, some invertebrate species with a close functional relationship to prairie plant species may serve as indicators of prairie condition and quality.

Numerous sampling techniques exist for monitoring invertebrate communities (Toermaelae 1982, Standen 2000, Schauff 2001, Epsky et al. 2008). The purpose of this project is to identify efficient methods for monitoring the status and trends of invertebrate communities across a range of grassland habitats from high quality prairies to planted grasslands, and for monitoring the effectiveness of management treatments intended to maintain or improve quality of grassland habitats. Our objectives were to: (1) evaluate the effectiveness of 3 invertebrate sampling methods (i.e., pitfall traps versus sweep-nets versus vacuum-sampler) for estimating invertebrate diversity and abundance; and (2) identify invertebrate taxa that may serve as indicator species for trend and effectiveness monitoring of grassland habitats. This proposal expands on 2 studies currently in progress. The first is a study on vegetation and bird diversity on high-quality prairie sites in western Minnesota. The second is a study evaluating methods for establishing and maintaining forbs in existing species-poor grasslands (Tranel 2009).

METHODS

Four study sites located in southern Minnesota were chosen for insect sampling during the pilot study (Figure 1). Two sites were on Wildlife Management Areas (WMA) representing low vegetation diversity stands of restored native grass, and 2 sites were high diversity prairie remnants located on Scientific and Natural Areas (SNA). Samples were collected using 3 methods: pitfall traps, sweep-nets, and vacuum-sampling. We recorded weather parameters, including ambient temperature, wind speed, percent humidity, and cloud cover during each sampling event. Ten 50-m transects were randomly established with a sampling point flagged every 10 m for a total of 5 points along each transect. Locations of all sampling points were recorded using a handheld Global Positioning System (GPS) unit.

Pitfall traps were dug at each sampling point using a garden bulb digger to avoid disturbing surrounding vegetation. Two 532-ml plastic cups were placed in each hole and filled with water containing a small amount of dish soap to break the surface tension. A yellow plastic funnel was placed over each cup opening to attract pollinators and help prevent incidental catch of small rodents and amphibians. Traps were set out for 5-day sampling periods in June, July, and August. Samples were labeled, placed in heavy duty Ziploc bags, and immediately frozen after collection.

Vacuum samples were collected on transects parallel and 1.5 m to the side of transects containing pitfall traps. Vacuum samples were collected using a Stihl BG86 handheld leaf blower/vacuum. The end of the vacuum was modified to fit a fine mesh-bottomed collection chamber to prevent suctioned insects and debris from entering the bag of the machine. A 75-L plastic garbage can was cut in half and covered in fine mesh with an elasticized hole in the top to place the vacuum tube to create an insect enclosure in which to vacuum. This insect enclosure was placed at each vacuum sampling point and the vacuum was operated on full power for 15 seconds within the enclosure.

Two sweep-net samples were collected using standard muslin insect sweep-nets on transects parallel and 3 m to the side of transects containing pitfall traps. The first sample starting point began 5 m from the beginning of the transect. The second sample starting point began 5 m before the beginning point of the transect. Each sample included 15 sweeps while walking the transect. A back-and-forth motion counted as 1 sweep and a typical 15-sweep transect was approximately 20 m long. We attempted the use of photo extractors (Molano-Flores 2002) to use light to attract live insects from sweep-net samples out of the vegetative debris and into a clean container. However, we had limited success using this device and found sorting dead insects in the laboratory to be more efficient.

All samples were stored in a freezer until we were ready to sort them. The pitfall samples were removed from the bags and thawed in containers 1 day prior to sorting. Once thawed, specimens were sorted from organic material and the catch solution and stored in 70% isopropyl alcohol in 20-ml vials. The number of samples thawed each day depended on the number that could be expected to be visually inspected and sorted the following day.

The sweep and vacuum samples were removed from the freezer, stored in 70% isopropyl alcohol, then spread on a sheet of white paper in order to facilitate sorting, and the alcohol was allowed to evaporate. The sides of the original sample bag were also rinsed with isopropyl alcohol and emptied into a small plastic weigh boat to facilitate removal of additional specimens. The evaporated portion of the sample was examined for any specimens hidden within the plant matter. The plant matter was also washed with isopropyl alcohol to remove additional specimens. The samples were sorted under a high power light source using soft forceps to prevent any damage to the specimens.

The specimens (including all arthropods) were stored in 20-ml glass vials with 70% isopropyl alcohol. After sorting, specimens were identified to family. Each sample was separated by family, with each family in its own labeled vial. Following identification and sorting, each vial was topped off with isopropyl and the lid tightly sealed with para-film to prevent evaporation during long-term storage. Voucher specimens were maintained for each family.

RESULTS

Three sampling periods, approximately 1 month apart, were completed for each collection method. Ten transects were sampled at 3 sites for the months of June and July. Due to the prevalence of poison ivy at the Butternut SNA, we were able to sample only 5 transects at that site. Due to staff time constraints, we determined that it was not possible to collect all the samples at each site; therefore, sampling effort was reduced for the last month so that data were collected at only 2 sites, 1 restored and 1 remnant site. A total of 752 samples were collected at the 4 sites throughout all sampling periods.

To date, approximately 240 samples have been sorted with an average sorting time of 22 minutes per sample. The time to process the samples was shortest for those from the pitfall traps and averaged 10 minutes to sort and 15 minutes to identify specimens. The sweep and vacuum samples varied widely in sorting time (ranged from 35–60 minutes per sample) depending on how much plant material the sample contained. Once sorted, an additional 30–45 minutes per sample was required for identification of invertebrates in the sweep and vacuum samples.

Initial findings suggest that the vacuum samples usually contained more plant material and less identifiable invertebrates than the other sampling methods. Most of the invertebrates found in the vacuum samples were pieces of invertebrates that were dismembered by the suction of the vacuum. Vacuum-sampling in the tall-grass habitat was also physically demanding and more time-consuming than for the sweep method. The sweep samples, on the other hand, contained a large quantity of insects of a variety of families. Additionally, sweep sampling was the least time-consuming and easiest sampling method for this habitat type.

Of the small number of pitfall samples that have been completely identified (restored site, *n* = 5; remnant site, *n* = 10), Hymenoptera was the most common order found at the 2 sites that have been analyzed (Table 1; restored site, *n* = 910; remnant site, *n* = 235). Most of the individuals were in the ant family, Formicidae (restored site, *n* = 607; remnant site *n* = 184). Pitfall traps at the restored site contained more individuals (1,620) and had greater diversity (44 insect families) than at the remnant site (individuals = 676, families = 35), but half of the remnant samples have yet to be completed.

DISCUSSION

Toermaelae (1982) recommended suction-sampling for most arthropods, but found that sweep–net-sampling efficiently collected flying insects, such as Diptera and winged Hymenoptera, and pitfall traps were more effective at collecting ground-dwelling beetles, hoppers, and spiders. Borges and Brown (2003) recommended that pitfall- and suction-sampling methods be used in conjunction to get reliable estimates for herbivore guilds in grazed pastures.

In our study, vacuum-sampling was problematic, because (1) invertebrates were damaged and rendered difficult to identify, (2) samples contained large amounts of plant matter, (3) greater physical effort was needed, and (4) 2 people were required to collect these data. For those reasons, this collection method should not be used during the full study. Sweep-net-sampling appeared to yield a large quantity of insects and was an easy collection method that required only 1 person. Pitfall traps required more initial effort to dig holes, and were more destructive to local vegetation than the other methods. Pitfall traps only required 1 person, but also required more visits to the site than the other methods. Future identification of the pilot season samples will help us determine if pitfall- and sweep-net-sampling methods are sufficient to sample the entire invertebrate community at our study sites, or if additional collection methods are necessary.

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LITERATURE CITED

Borges, P.A.V., and V.K. Brown. 2003. Estimating species richness of arthropods in Azorean pastures: the adequacy of suction sampling and pitfall trapping. Graellsia 59:7-24.

Buchanan, G. M., M. C. Grant, R. A. Sanderson, and J. W. Pearce-Higgins. 2006. The contribution of invertebrate taxa to moorland bird diets and the potential implications of land-use management. Ibis 148:615-628.

Epsky, N. D., W. L. Morrill, and R. W. Mankin. 2008. Traps for capturing insects. Pages 3887-3900 *in* J.L. Capinera, ed. Encyclopedia of entomology. Springer, A. A. Dordrecht, The Netherlands.

Kremen, C., R. K. Colwell, T. L. Erwing, D. D. Murphy, R. F. Noss, and M. A. Sanjayan. 1993. Terrestrial arthropod assemblages: their use in conservation planning. Conservation Biology 7:796-808.

Moloano-Flores, B. 2002. Critical trends assessment program monitoring protocols. Illinois Natural History Survey, Office of the Chief, Technical Report 2002-2, Champaign. 38pp.

Schauff, M. E. 2001. Collecting and preserving insects and mites: techniques and tools. Systematic Entomology Laboratory, USDA National Museum of Natural History, Washington, D. C. 68pp.

Standen, V. 2000. The adequacy of collecting techniques for estimating species richness of grassland invertebrates. Journal of Applied Ecology 37:884-893.

Toermaelae, T. 1982. Evaluation of five methods of sampling field layer arthropods, particularly the leafhopper community, in grassland. Annales Entomologici Fennici 48:1-16.

Tranel, M. A. 2009. Research proposal: establishment and maintenance of forbs in existing grass stands. Pages 267-270 *in* M. W. DonCarlos, R. O. Kimmel, J. S. Lawrence, and M. S. Lenarz, eds. Summaries of wildlife research findings. Minnesota Department of Natural Resources, Wildlife Populations and Research Unit, St. Paul.

Table 1. Total number of specimens collected in pitfall traps (to date) at 2 of the 4 study sites in southern Minnesota during summer 2010. Joseph Tauer Scenic and Natural Area was a prairie remnant site that had 5 samples identified, and Peterson Lake Wildlife Management Area was restored grassland with 10 samples identified.

		Specimen number	
Order	Family	Restored *n* = 10	Remnant *n* = 5
Class: Arachnida			
Acari (Mites and Ticks)	Unidentified	138	24
Araneae (Spiders)	Unidentified	32	88
Opiliones (Daddy Long-legs)	Unidentified	9	20
Class: Malacostraca			
Isopoda	Unidentified	52	135
Class: Diplopoda			
Unidentified (Millipedes)	Unidentified	4	2
Phylon: Mulluscea/Class: Gastropoda			
Stylommatophora (Land Snails)	Unidentified	0	1
	Total Other:	235	270
Class: Insecta			
Coleoptera (Beetles)	Anthicidae (Antlike Flower Beetles)	0	1
	Carabidae (Ground Beetles)	26	10
	Chrysomelidae (Leaf Beetles)	8	2
	Cicindelidae (Tiger Beetles)	1	0
	Cryptophagidae (Silken Fungus Beetles)	1	0
	Curculionidae (Snout Beetles)	4	1
	Endomychidae (Handsome Fungus Beetles)	3	0
	Erotylidae (Pleasing Fungus Beetles)	1	0
	Lathridiidae (Minute Brown Scavenger Beetles)	47	3
	Nitidulidae (Sap Beetles)	18	1
	Phalacridae (Shining Flower Beetles)	54	2
	Scarabaeidae (Scarab Beetles)	2	0
	Silphidae (Carrion Beetles)	1	0
	Staphylinidae (Rove Beetles)	26	10
Collembola (Springtails)	Unidentified	9	3
Diptera (Flies)	Unidentified	159	54
	Syrphidae (Syrphid Flies or Flower Flies)	5	4
Ephemeroptera (Mayflies)	Unidentified	6	0
Homoptera	Aphididae (Aphids or Plantlice)	19	3
	Cercopidae (Frog Hoppers and Spittle Bugs)	9	17
	Cicadellidae (Leafhoppers)	59	40
Hemiptera (True Bugs)	Alydidae (Broad Headed Bugs)	0	2
	Lygaeidae (Seed Bugs)	5	3
	Pentatomidae (Stink Bugs)	2	1

	Thyreocoridae (Negro Bugs)	0	1
Hymenoptera (Bees,wasps,ants, etc.)	Anthophoridae (Cuckoo, Digger and Carpenter Bees)	0	1
	Braconidae	3	2
	Ceraphronidae	144	13
	Chalcidoidea (Unidentified)	57	13
	Chalcidoidea Mymaridae (Fairyflies)	11	3
	Cynipidae (Gall Wasps)	13	1
	Diapriidae	35	1
	Embolemidae	1	0
	Formicidae (Ants)	607	184
	Halictidae	1	0
	Ichneumonidae (Ichneumonids)	13	2
	Platygastridae	4	8
	Pompilidae (Spider Wasps)	1	0
	Megaspilidae	2	0
	Scelionidae	17	7
Lepidoptera (Butterflies and Moths)	Papilionidae (Swallowtails and Parnasians)	1	0
	Unidentified immature	2	1
Orthoptera (Grasshoppers, Crickets, and Katydids)	Acrididae (Short-Horned Grasshoppers)	1	1
	Gryllidae (Crickets)	5	18
Thysanoptera (Thrips)		1	0
	Total Insects:	1,385	413

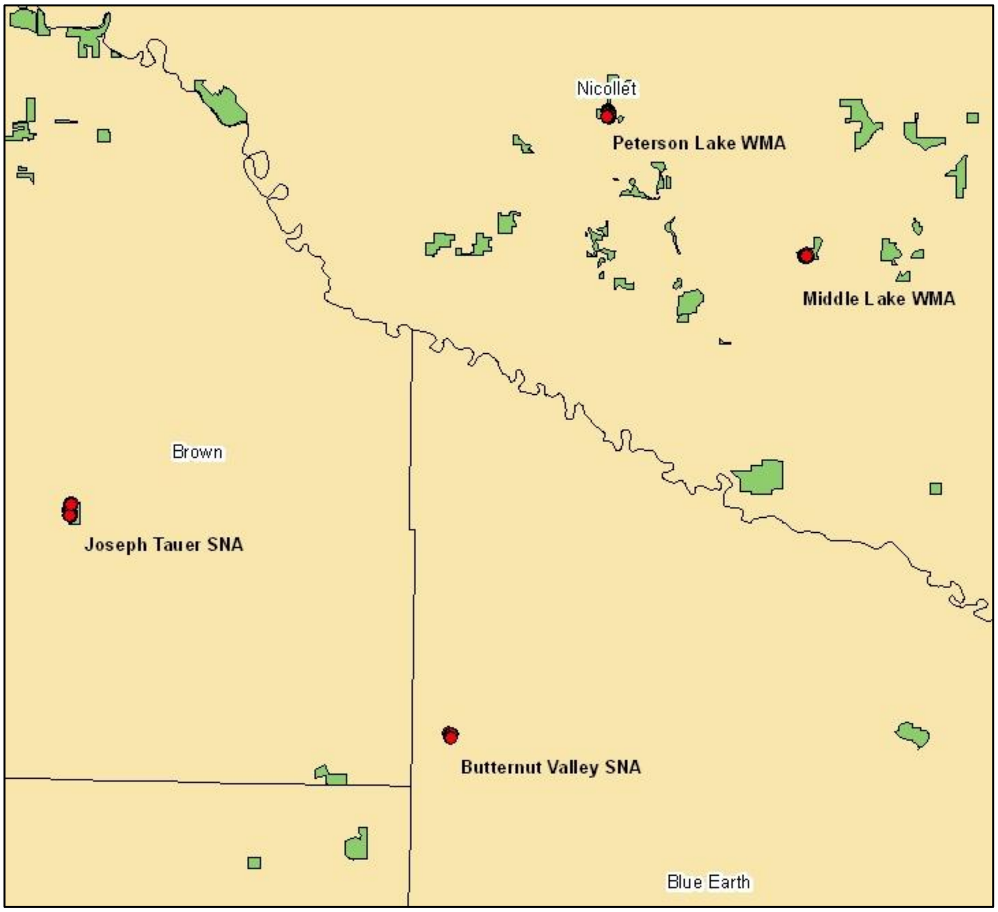

Figure 1. Map of sites where insects were sampled in summer 2010 in southern Minnesota. Wildlife Management Areas (WMA) represent low diversity grassland sites (restored) and Scenic and Natural Areas (SNA) represent high diversity prairie remnants. Green blocks represent other state owned natural areas.

ESTABLISHMENT AND MAINTENANCE OF FORBS IN EXISTING GRASS STANDS- PILOT SEASON FIRST YEAR FINDINGS

Molly A. Tranel

SUMMARY OF FINDINGS

Interseeding native forbs into reconstructed grasslands could restore plant species diversity and improve wildlife habitat. However, many managers report having limited experience with interseeding and poor success with a few early attempts. Survival of forbs interseeded directly into existing vegetation may be enhanced by management treatments that reduce competition from established grasses. In 2009, a study was initiated to investigate the effects of 2 mowing and 2 herbicide treatments on diversity and abundance of forbs interseeded into established grasslands in southern Minnesota. In January of 2009, a pilot site was interseeded and 2 mowing treatments (once or twice per season) and 2 grass-selective herbicide treatments (high and low rate) were applied during the 2009 growing season. Vegetation from the pilot site was monitored during the summer of 2010. One month after treatments were applied, mean visual obstruction readings were less in treated plots than in control plots, indicating the prescribed treatments were more effective at suppressing growth of grass. Black-eyed Susan (*Rudbeckia hirta*) was the most common seeded species in all treatments and was more frequent in the herbicide-low, mow, and herbicide-high treatments, respectively, than in the control. Results of the pilot study were used to guide a larger study on 16 sites in southern Minnesota.

INTRODUCTION

Minnesota Department of Natural Resources (MNDNR) wildlife managers indicated a need for more information on establishing and maintaining an abundance and diversity of forbs in grasslands (Tranel 2007). Forb diversity in grasslands provides the heterogeneous vegetation structure needed by some bird species for nesting and brood rearing (Volkert 1992, Sample and Mossman 1997). Forbs also provide habitat for invertebrates, an essential food for grassland birds and their broods (Buchanan et al. 2006).

The forb component on many restored grasslands has been lost or greatly reduced. Managers interested in increasing the diversity and quality of forb-deficient grasslands are faced with the costly option of completely eliminating the existing vegetation and planting into bare ground, or attempting to interseed forbs directly into existing vegetation. Management techniques that reduce competition from established grasses may provide an opportunity for forbs to become established in existing grasslands (Collins et al. 1998, McCain et al. 2010). Temporarily suppressing dominant grasses may increase light, moisture, and nutrient availability to seedling forbs, ultimately increasing forb abundance and diversity (Schmitt-McCain 2008, McCain et al. 2010). Williams et al. (2007) found that frequent mowing of grasslands in the first growing season after interseeding increased forb emergence and reduced forb mortality. Snyder et al. (unpublished data) reported that patch tilling and interseeding with forbs can increase species diversity in grass dominated stands. Additionally, Hitchmough and Paraskevopoulou (2008) found that forb density, biomass, and richness were greater in meadows where a grass herbicide was used.

In this study, we investigated the effects of 2 mowing and 2 herbicide treatments on diversity and abundance of forbs interseeded into established grasslands in southern Minnesota.

METHODS

We selected 1 pilot site in 2008 and 17 sites in 2009 for the full study. Study sites were distributed throughout the southern portion of Minnesota’s prairie/farmland region on wildlife areas owned by the state and federal government. Each site was ≥4 ha and characterized by relatively uniform soils, hydrology, and vegetative composition. All sites were dominated by relatively uniform stands of native grasses with few forbs, most of which were non-native, such as sweet clover (*Melitotus alba, M. officinalis*).

The pilot site was interseeded during January 2009 following a fall 2008 burn. For the full study, 8 sites were burned in October–November 2009 and frost interseeded during December 2009 and March 2010; 8 sites were burned and interseeded during April and May 2010, and 1 site in Faribault County that was not burned was dropped from the study. The same 30-species mix of seed was broadcast seeded at all sites at a rate of 239 pure live seeds/m^2. Seed used on spring-burned sites was cold-moist stratified for 3-5 weeks in wet sand to stimulate germination during spring 2010 and seed used on fall-burned sites was not.

Treatments

We divided sites into 10 plots of approximately equal size and randomly assigned each of 4 treatments and the control. Each site received all treatments to account for variability among sites, and each treatment was replicated twice at each site. The following treatments, designed to suppress grass competition, were applied during the first growing season after interseeding (2009 for the pilot study, 2010 for the full study) while the forbs were becoming established:

- Mowed once to a height of 10–15 cm when vegetation reached 25–35 cm in height.
- Mowed twice to a height of 10–15 cm when vegetation reached 25-35 cm in height.
- Applied grass herbicide Clethodim (Select Max®) at 108 mL/ha (9 oz/A) when vegetation reached 10–15 cm.
- Applied grass herbicide Clethodim (Select Max®) at 215 mL/ha (18 oz/A) when vegetation reached 10–15 cm.

Sampling Methods

Prior to burning and interseeding, all sites, except the pilot, were surveyed by a botanist in summer 2009 to determine species already present and general condition of each site. We sampled the pilot site in summer 2010 to determine initial success of the treatments. We randomly located 4 transects 50 m in length within each study plot and recorded map coordinates using a Global Positioning System unit. We estimated percent cover of live vegetation (Daubenmire 1959) within 76 x 31 cm^2 quadrats spaced every 5 m and litter depth was estimated every 10 m. We recorded visual obstruction readings (VOR, Robel et al. 1970) in the 4 cardinal directions at the beginning and the end of each transect. Species richness was estimated by counting the number of sampling quadrats for which each species was present (frequency). We conducted sampling on the pilot site in the summer of 2010 and will continue on the full study sites in following years.

RESULTS AND DISCUSSION

Pilot Site

Due to staff limitations, the second mowing treatment was not applied to the pilot site and herbicide treatments were *applied when the grass was taller (31 cm*) than prescribed (10-15 cm). Big bluestem (*Andropogon gerardi*) was observed most frequently in the control (90%), but appeared less frequently in the mow-once treatment (53.3%), herbicide-low treatment (72.5%), and the herbicide-high treatment (70.0%, Figure 1). Quackgrass (*Agropyron repens*) occurred most frequently in the mow-once treatment (56.7%), and was abundant in the control (50.0%), herbicide-low treatment (35.0%), and the herbicide-high treatment (45.0%, Figure 1).

Dandelion (*Taraxacum officinale*) was more frequent in the mow-once treatment (46.7%) and herbicide-high treatment (60.0%) than in the control (33.3%, Figure 2). Creeping woodsorrel (*Oxalis corniculata*) was more frequent in the herbicide-low treatment (57.5%), ,mow-once treatment (46.7%), and herbicide-high treatment (45.0%) than in the control (36.7%, Figure 2). Sweet clover was common regardless of treatment (Table 1 and Figure 2).

One year following treatments, native, seeded forbs were present in all treatments (Table 1). Black-eyed Susan was the most commonly seeded forb species in all treatments and was most common in the herbicide-low treatment (50%), but was rarely observed in the control (13.3%, Table 1). The mow-once treatment had the greatest diversity of native, seeded species and the herbicide-high treatment had the lowest diversity (Table 1).

One month after treatments were applied, mean VORs were less in treated plots than in control plots (Tranel 2009), and frequency of occurrence of big bluestem and foxtail in the first year of the pilot study (Figure 1) was reduced. These results, combined with the increased frequency of weedy, disturbance loving species in the herbicide treatments (Figure 2) suggests the prescribed treatments were effective in suppressing growth of dominant grasses. Hitchmough and Paraskevopoulou (2008) found that in treatments where grass was suppressed with a graminoid herbicide, sown forb density was higher in the second and third years after treatment and forb richness was greater 3 years after treatment. Additionally, Willliams et al. (2007) reported that in the fourth year sown forbs were twice as abundant in treatments where grass was suppressed by mowing than in untreated controls. A complete vegetation survey will be conducted on all sites in the study in summers 2011–2013 to determine the extent of forb survival, species diversity, and weed persistence.

MANAGEMENT IMPLICATIONS

The use of the pre-emergent grass-selective herbicide, Clethodim (Select Max), at 108 mL/ha (9 oz/A) and 215 mL/ha (18 oz/A) was effective at suppressing well established native and exotic grasses at the pilot site. Growth of grass was inhibited, but grass mortality was not observed even at the high application rate on any of the study sites. Because this herbicide is relatively inexpensive and requires only one application in a growing season, it could prove to be a cost effective alternative to repeated mowing in areas where grass suppression is desired. Using grass-selective herbicides followed by interseeding in order to achieve other management objectives warrants further investigation.

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LITERATURE CITED

Buchanan, G. M., M. C. Grant, R. A. Sanderson, and J. W. Pearce-Higgins. 2006. The contribution of invertebrate taxa to moorland bird diets and the potential implications of land-use management. Ibis 148:615-628.

Collins, S. L., A. K. Knapp, J. M. Briggs, J. M. Blair, and E. M. Steinauer. 1998. Modulation of diversity by grazing and mowing in native tallgrass prairie. Science 280:745-747.

Daubenmire, R. F. 1959. Canopy coverage method of vegetation analysis. Northwest Science 33:43-64.

Hitchmough, J., and A. Paraskevopoulou. 2008. Influence of grass suppression and sowing rate on the establishment and persistence of forb dominated urban meadows. Urban Ecosystems 11:33-44.

Mccain, K. N. S., S. G. Baer, J. M. Blair, and G. W. T. Wilson. 2010. Dominant grasses suppress local diversity in restored tallgrass prairie. Restoration Ecology 18:40-49.

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.

Sample, D. W., and M. J. Mossman. 1997. Managing habitat for grassland birds, a guide for Wisconsin. Wisconsin Department of Natural Resources, Bureau of Integrated Science Services, Madison.

Schmitt-Mccain, K. N. 2008. Limitations to plant diversity and productivity in restored tallgrass prairie. Ph.D. dissertation. Kansas State University, Manhattan.

Snyder, L. J., P. E. Rothrock, and R. T. Reber. Unpublished data. Interseeding forbs in a tallgrass prairie restoration: Effects of three disturbance regimes-first growing season. Randall Environmental Learning Center, Taylor University, Upland, Indiana. http://www.taylor.edu/dotAsset/104110.pdf Last accessed 20 September 2010.

Tranel, M. A. 2007. Management focused research needs of Minnesota's wildlife managers-grassland management activities. Pages 91-95 *in* M. W. DonCarlos, R. O. Kimmel, J. S. Lawrence, and M. S. Lenarz, eds. Summaries of wildlife research findings, 2006. Minnesota Department of Natural Resources, Wildlife Populations and Research Unit, St. Paul.

Tranel, M. A. 2009. Establishment and maintenance of forbs in existing grass stands-pilot season update. Pages 261-266 in M.W. DonCarlos, R.O. Kimmel, J. S. Lawrence, and M. S. Lenarz (eds.). Summaries of wildlife research findings, 2008. Minnesota Department of Natural Resources, Wildlife Populations and Research Unit, St. Paul.

Volkert, W. K. 1992. Response of grassland birds to a large-scale prairie planting project. The Passenger Pigeon 54:191-195.

Williams, D. W., L. L. Jackson, and D. D. Smith. 2007. Effects of frequent mowing on survival and persistence of forbs seeded into a species-poor grassland. Restoration Ecology 15:24-33.

Table 1. Frequency of selected forb species by treatment type at the Wood Lake, Minnesota pilot study site 1 year following treatments (2010). Sample size (*n*) is the number of quadrats placed in groups of 10 along transects within each treatment type.

Scientific name	Common name	Plant status	Presence	*n*	Frequency
Mow-once treatment					
Rudbeckia hirta	(Black-eyed Susan)	Native, seeded	8	30	26.7%
Solidago canadensis	(Canada goldenrod)	Native, volunteer	3	30	10.0%
Monarda fistulosa	(Wild bergamot)	Native, seeded	3	30	10.0%
Aster spp.	(Unidentified asters)	Native, seeded	1	30	3.3%
Zizia aurea	(Golden Alexanders)	Native, seeded	1	30	3.3%
Dalea candida	(White prairie clover)	Native, seeded	1	30	3.3%
Melitotus alba & M. officinalis	(White & yellow sweet clover)	Non-native	11	30	36.7%
Control - No treatment					
Rudbeckia hirta	(Black-eyed Susan)	Native, seeded	4	30	13.3%
Zizia aurea	(Golden Alexanders)	Native, seeded	1	30	3.3%
Monarda fistulosa	(Wild bergamot)	Native, seeded	1	30	3.3%
Amorpha canescens	(Leadplant)	Native, seeded	1	30	3.3%
Melitotus alba & M. officinalis	(White & yellow sweet clover)	Non-native	14	30	46.7%
Herbicide-low treatment					
Rudbeckia hirta	(Black-eyed Susan)	Native, seeded	20	40	50.0%
Ratibida pinnata	(Yellow coneflower)	Native, seeded	3	40	7.5%
Solidago canadensis	(Canada goldenrod)	Native, seeded	2	40	5.0%
Zizia aurea	(Golden Alexanders)	Native, seeded	1	40	2.5%
Monarda fistulosa	(Wild bergamot)	Native, seeded	1	40	2.5%
Melitotus alba & M. officinalis	(White & yellow sweet clover)	Non-native	17	40	42.5%
Herbicide-high treatment					
Rudbeckia hirta	(Black-eyed Susan)	Native, seeded	4	20	20.0%
Solidago canadensis	(Canada goldenrod)	Native, volunteer	3	20	15.0%
Asclepias syriaca	(Common milkweed)	Native, seeded	1	20	5.0%
Melitotus alba & M. officinalis	(White & yellow sweet clover)	Non-native	12	20	60.0%

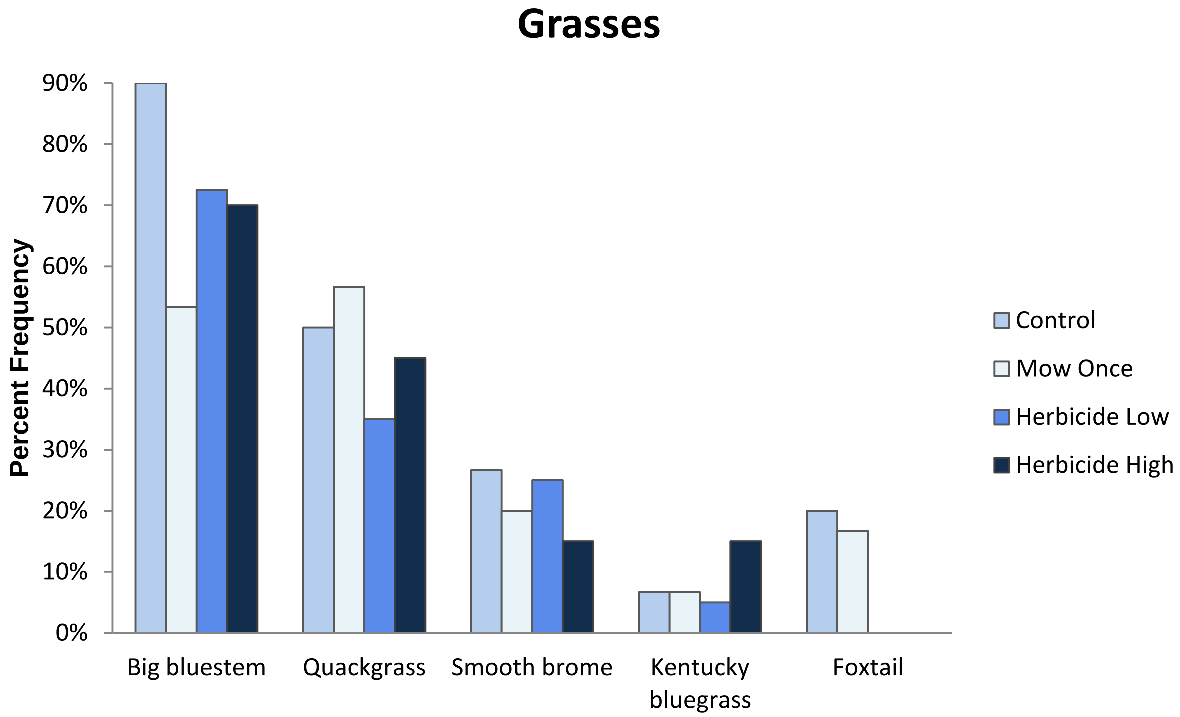

Figure 1. Percent frequency of selected grass species observed during treatment year at the pilot study site, Wood Lake, Minnesota, 2010 (*n* = 30 for mow-once and control, *n* = 40 for herbicide-low, and *n* = 20 for herbicide-high).

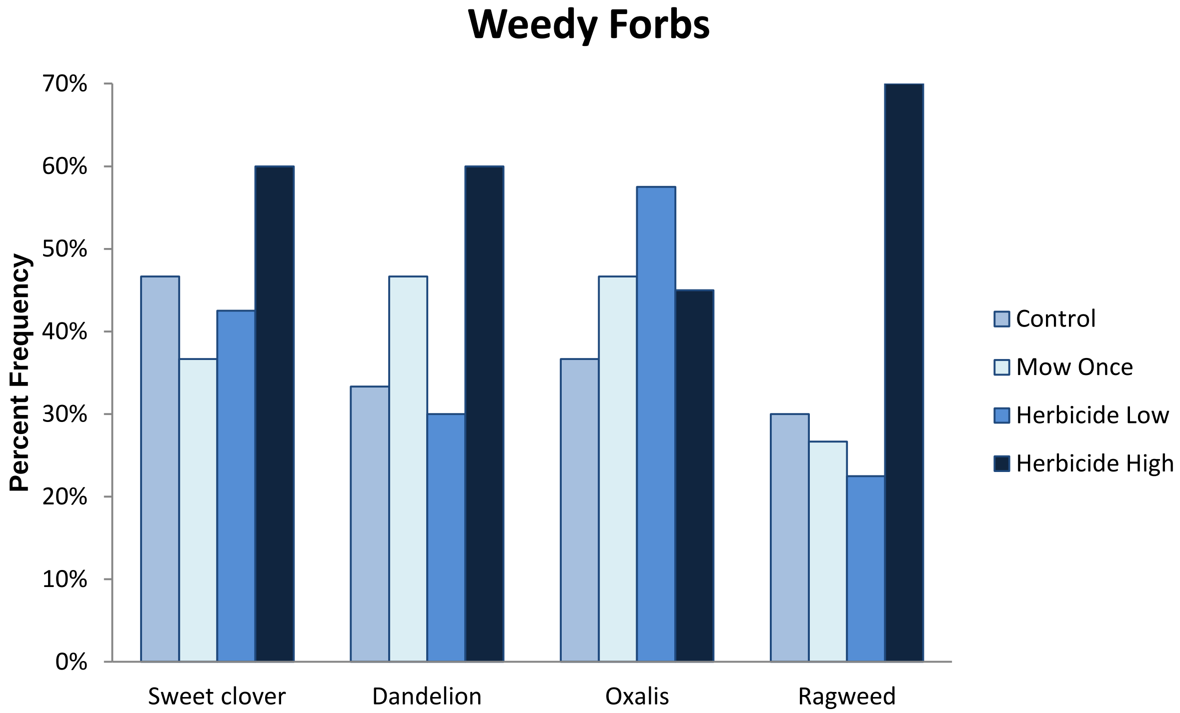

Figure 2. Percent frequency of selected weedy forb species observed during treatment year at the pilot study site, Wood Lake, Minnesota, 2010 (*n* = 30 for mow-once and control, *n* = 40 for herbicide-low, and *n* = 20 for herbicide-high).

HUNTER PERCEPTIONS AND ACCEPTANCE OF ALTERNATIVE DEER MANAGEMENT REGULATIONS[1]

Louis Cornicelli, Marrett D. Grund, and John Fieberg

ABSTRACT

Recreational hunting is the primary tool managers use to regulate deer populations, yet there is a belief that adequate hunter numbers may not exist in the future. Previous research has reported a link between regulatory acceptance, hunter satisfaction, and participation rates. Wildlife managers are often confronted with a policy paradox in that individuals desire an outcome, yet cannot agree on specific regulations. Thus, human dimensions research aimed at understanding hunter motivations and behavior is needed for effective management. In 2005, we surveyed Minnesota deer hunters (n = 6,000; 59% response) to evaluate attitudes regarding alternative deer harvest regulations. We also conducted a series of forced choice experiments in which respondents were asked to select an option from a list of representative regulations that might be adopted. Specifically, we modeled 5 deer management and population-level scenarios ranging from low populations with high buck harvest rates to populations 50% over goal density. Our results indicate that hunters preferred different regulations depending on the management scenario, but generally preferred antler point restrictions and disliked limiting buck licenses through a lottery. We also found consistency among scenarios in that a small percentage of respondents indicated they would not hunt if regulations were changed. The results from this study should help wildlife managers design deer harvest regulations that achieve management objectives and are acceptable to deer hunters.

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ASSESSING DEER HUNTER ATTITUDES TOWARDS REGULATORY CHANGE USING SELF-SELECTED RESPONDENTS[1]

Louis Cornicelli and Marrett D. Grund

ABSTRACT

State wildlife agencies are charged with managing wildlife populations of harvestable species. Opinions regarding how species should be managed differ dramatically and decisions are often made without comprehensive data. There is interest among deer hunters in Minnesota to change harvest regulations that benefit mature bucks. Beginning in 2005, our research focused on the biological and social implications of changing deer regulations. We used data collected via random surveys, public input meetings, and an internet survey to assist with the decision-making process. We observed demographic differences among respondents for the 3 data collection methods; however, the attitudinal differences were "minimal" (i.e., Cramer's $V \approx .1$). We believe that a structured public input process, even if self-selected, can help inform decision-makers. Agencies that use public input meetings should reevaluate their public input process to include an internet component if there are adequate baseline data available to make comparisons.

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[1] Paper published in Human Dimensions of Wildlife 16: 174–182

SURVEY OF SOUTHEAST AND CENTRAL DEER HUNTER PREFERENCES FOR PROPOSED REGULATION CHANGES IN MINNESOTA

Emily J. Dunbar, Louis Cornicelli, and Marrett D. Grund

SUMMARY OF FINDINGS

The Minnesota Department of Natural Resources (MNDNR) Section of Wildlife conducted a survey of firearm white-tailed deer (*Odocoileus virginianus*) hunters to assess support for proposed alternative hunting regulations. Alternative regulations are designed to shift white-tailed deer harvests from yearling bucks to mature bucks and antlerless deer and may be applied in areas that exceed population goals. However, hunter support is critical before the MNDNR can implement the regulations. The southeast and central regions of Minnesota have some of the highest deer densities in the state and were chosen to determine which alternative regulations would be supported by hunters. Surveys were sent to hunters who indicated they had hunted the 3A or 3B season in the southeast region, and hunters in the central region who had hunted in deer permit areas 240, 241, 243, or 244.

Results indicate that hunters from all 3 samples were in favor of regulations that protect a majority of yearling bucks and increase the proportion of mature bucks in the deer population, but hunters from the southeast generally differed from central region hunters in their level of support for specific regulations. Elimination of buck cross-tagging was supported by more southeast hunters than central hunters. An antler point restriction regulation was supported by more hunters than opposed in the southeast, but the opposite was true in the central region. Having a youth-only season was supported by more southeast hunters than central region hunters. Few differences were found in the level of support among years of hunting experience within and among samples. For an antler point restriction regulation, there were no differences in support among the 3 samples, but hunters in the central region with 11-20 years hunting experience had lower levels of opposition than other groups of hunters. A youth-only season was supported by more hunters in the central region with 1-20 years of experience than more experienced hunters. When hunters were divided into groups based on the type of land they hunted, there were differences concerning their level of support for mature buck regulations, elimination of buck cross-tagging, and a youth-only season. Southeast 3A hunters who hunt on leased private land had higher support for mature buck regulations than either southeast 3B hunters or central region hunters. These surveys demonstrate that while most hunters in the study areas support mature buck regulations, there is not majority support for any particular strategy. Southeast hunters tended to be more supportive of eliminating buck cross-tagging, instituting an antler point restriction, and holding a youth-only season than hunters in the central region. Central region hunters tended to be split between supporting and opposing the 3 regulations we proposed in the survey.

INTRODUCTION

In 2005, the MNDNR initiated a study to determine what impact alternative deer hunting regulations (early antlerless season [EA], earn-a-buck [EAB], and antler-point restriction [APR]) would have on deer populations. Another component of the research was to assess hunter support for these hunting regulations. In 2005, a survey was mailed to 6,000 firearm deer hunters and results indicated most hunters (66%) supported the concept of increasing the proportion of antlered bucks in the population; however, no regulation had >50% support (Fulton et al. 2006). Results also suggested that hunters with more hunting experience (>40 years) were less likely to support regulations to increase the proportion of antlered bucks in the herd than those with less hunting experience . Additionally, hunters in the southeast and central regions of the state were most receptive to implementation of alternative regulations. Hunter input was needed to determine which new regulations would be supported by hunters before the MNDNR could consider implementation. The purpose of this study was to assess hunter support for regulations meant to shift harvest from yearling bucks to antlerless deer in the southeast and central regions of the state.

METHODS

Minnesota residents aged 18 years or older who purchased a firearms license to hunt deer in each study area were included in our hunter survey. Names and addresses of hunters were provided by the electronic licensing system (ELS) maintained by the MNDNR. A total of 6,000 individuals were randomly chosen for the Zone 3A and 3B seasons from the 2008 ELS database (*n* = 1,500 each). The 2009 ELS database was used to select 3,000 individuals who indicated that they hunted in deer areas 240, 241, 243, and 244 (central study area). A self-administered, mail-back questionnaire with a personalized cover letter, and postage-paid return envelope was sent to individuals of the southeast sample in June 2009 and to individuals of the central sample in January 2010. The questionnaire was required 2 sides of a legal-sized piece of paper and was divided into the following categories: (1) hunter background, (2) hunting property, and (3) hunter support for past and proposed regulation changes (Appendix I). Two follow-up surveys were mailed out at 4-week intervals to non-respondents.

We studied the effect hunting experience had on supporting hunting regulations by placing hunters into 1 of the following experience categories: (1) 1-10, (2) 11-20, (3) 21-30, (4)31-40, and (5) ≥40 years. We investigated whether landownership patterns affected regulatory support by asking participants to estimate how much of their time spent hunting was located on: (1) private land they owned, (2) private land they leased for hunting, (3) private land neither owned nor leased, or (4) public land. Responses were on a 4-point scale that ranged from “none” to “all”. Respondents indicating they either spent “most” or “all” of their time hunting on any 1 type of land were included in the landownership analysis.

Data from the 3 study areas were not pooled due to the differences in seasonal framework. Questionnaires returned later than 4 weeks after the last mailing were not included in the analyses. We calculated the frequency score for each proposed regulation change (oppose = 1, support = 2, and neither = 3) in each study area based on hunter experience and type of land hunted.

RESULTS

For the southeast 3A sample, 41 surveys were undeliverable or addressed to out of state hunters (3%), leaving 1,459 potentially returnable and usable questionnaires. Of the 1,459 returnable questionnaires, 891 were returned, for a 61% response rate. For the southeast 3B sample, 44 surveys were undeliverable or addressed to out of state hunters (3%), leaving 1,456 potentially returnable and usable questionnaires. Of the 1,456 returnable questionnaires, 860 were returned, for a 59% response rate. For the central sample, 71 surveys were undeliverable or addressed to out of state hunters (2%), leaving 2,929 potentially returnable questionnaires. Of the 2,929 returnable questionnaires, 2,075 were returned, for a 71% response rate. Seventy-nine surveys were returned incomplete, for a remaining sample size of 1,996 surveys in the central region sample.

Since the response rate for southeast hunters was not >60%, a non-response survey was sent to both 3A and 3B hunters. Four hundred abbreviated surveys were sent out to hunters for each season. Non-response surveys differed from the original survey by not including questions concerning the type of hunter on the land in the opposite season, voluntary harvest restrictions, opinions on past regulation changes, and opinions on delaying the season, consolidating the 3A and 3B seasons, and a youth-only season. Fifty surveys were returned from 3A hunters and 45 surveys were returned from 3B hunters, for a response rate of 13% and 11%, respectively. The number of non-response surveys returned for each season made up only 5% of the total number of surveys returned; therefore, the results were not analyzed to compare differences.

We determined that the hunters who returned surveys did not represent the hunting population (Table 1). Younger hunters were underrepresented and older hunters were over represented, which could have skewed age-based analysis of responses. The only possible effect of the bias for this report was the analysis of how hunting experience impacts levels of support for proposed regulations. Responses were not weighted to take into account the possible bias.

A respondent profile based on demographic data was constructed for each sample (Table 2). Hunter profiles in the 3 survey samples were similar. Most respondents held only a regular firearms license, with southeast 3B hunters having fewer hunters with only a regular firearms license (68%) than southeast 3A hunters (70%) or central hunters (71%). More hunters in the southeast region had both a regular firearms license and archery license (17% and 16%) than hunters in the central sample (12%). The central sample had the highest percentage of hunters with both a regular firearms and muzzleloader license (10%). All 3 license types were held by more southeast 3B hunters (10%) than either southeast 3A hunters (6%) or central hunters (7%). Hunting experience within the particular region was similar across the samples, and ranged from 17 to 18 years. Most hunters were not a member of a deer hunting organization, but more southeast hunters were members of the Minnesota Deer Hunters Association (MDHA) than any other group, whereas more central region hunters were members of the Quality Deer Management Association (QDMA). The percentage of respondents that hunted on each type of land was similar between southeast 3A and 3B hunters, but differed between southeast and central region hunters. More central region hunters did most or all of their hunting on land that they owned (40%) than southeast region hunters (20% and 22%). A majority of southeast region hunters hunted on private land that they neither owned nor leased (58% and 62%), while a smaller percentage of central region hunters (49%) did the same. Central region hunters hunted on public land to a lesser extent (8%) than southeast region hunters (12%). Fewer hunters that bought licenses in the central region hunted (91%) than southeast hunters (98%). Respondents that hunted during a 9-day season (southeast 3B and central region) averaged the same amount of time hunting (5 days), while southeast 3A hunters who had a 7-day season averaged 4 days in the field.

Hunters were asked a series of questions about who else hunted on the private land that they hunted, any type of harvest restrictions used on the land they hunted, and their perceptions about the number of mature bucks and the size of the deer herd (central only) where they hunted (Appendix I). Similar percentages of southeast 3A and 3B hunters who hunted on private land they did not own indicated that a hunter who was not a relative of the landowner (37% and 40%) and/or a hunter who was the landowner or an immediate family member (34% and 37%) hunted on the same land as they did in the opposite deer season. For central hunters, the landowner or immediate family member (75%) and/or a hunter who was a relative (35%) also hunted on the same land during the season as the respondent. Hunters in southeast 3A and 3B who hunted on land they owned responded that family that did not own the property (31%) and/or friends who did not own the property (38% and 31%) hunted on their property in the opposite deer season. Central hunters who hunted on property they owned indicated that family who did not own the property (61%) and/or friends who did not own the property (39%) hunted on their land during the same deer season. Voluntary restrictions on deer harvest were rarely used by hunters in the study areas. In southeast 3A and 3B, 61% and 63% of hunters who hunt on private land indicated that there were no harvest restrictions and 52% of central hunters responded likewise. The most common harvest restriction for all 3 samples was a buck harvest that was limited to large antlered bucks, but any antlerless deer could be taken (20-22%). Hunters from all 3 samples agreed (40-42%) that there were about the same number of mature bucks now that there were 5 years ago. Central hunters were also asked about their perception of the deer herd in the areas where they hunt now as compared to 5 years ago and what they thought of the deer population currently. Most hunters perceived that there were fewer deer now than 5 years ago (49%) or about the same now as compared to 5 years ago (36%). More hunters felt that the deer population was “about right” (54%) than “too low” (32%).

Southeast region respondents were asked about their level of support for regulation changes that have occurred concerning their seasonal framework (Table 3). In general, more hunters support than oppose the changes that have occurred since 2003. Most hunters in both seasons supported having youth participate in both seasons, with 67% of 3A hunters slightly or strongly supporting the change, and 65% of 3B hunters either slightly or strongly supporting the change. Shortening the 3A season by 2 days was not a popular change for most hunters (43% and 45% either strongly or slightly opposed), but lengthening the 3B season was either strongly or slightly supported by more hunters (56% and 54%). Allowing antlerless harvest during the traditional 3A "buck season" was supported by a majority of southeast hunters (58% either strongly or slightly supported the change). More hunters were in favor of the current season (51% or 48% either strongly or slightly support) than opposed (34% either strongly or slightly opposed).

The majority of questions pertaining to proposed regulations were common between the 2 zones, but southeast surveys included questions concerning consolidation of the A and B seasons and restoration of the past seasonal framework, whereas central surveys asked hunters' opinions on limiting the number of buck licenses and enacting mature buck regulations in a specific area versus statewide. Responses to proposed regulations that were asked on only 1 or 2 surveys were summarized in Appendix I. Level of support for proposed regulations that were on all 3 surveys is summarized in Table 4. Overall, hunters in both regions were in favor of regulations that protect a majority of yearling bucks and increase the proportion of mature bucks in the deer populations (52-53%), and oppose delaying the start of the deer firearms season by 1 week (55-56%). But southeast 3A and 3B hunters generally differed from central hunters in level of support for other proposed regulations. Elimination of buck cross-tagging was supported by more southeast 3A and 3B hunters ($\geq$50%) than central hunters (45%). Having a youth-only season in October was supported by more southeast hunters (54%) than central hunters (41%). Delaying the start of the firearms season until late November was opposed by more hunters than delaying the season by 1 week, but southeast hunters were less opposed (72%) to the proposed regulation than central hunters (82%). An antler point restriction regulation was supported by more southeast hunters than opposed, but the opposite was true in the central region. In southeast 3A, 45% of hunters supported the regulation, whereas 41% opposed it, and in 3B, 48% of hunters supported the regulation and 37% of hunters opposed. In central, the opinion was split between support (43%) and opposition (46%).

One of the last questions on the southeast survey asked hunters about losing access to property if the 3A and 3B seasons were consolidated (Appendix I). Respondents were split, with 45% of 3A hunters responding that they either strongly or slightly disagreed with the statement about losing access and 41% either strongly or slightly agreed with it. For 3B hunters, the split was narrower, with 43% of hunters either strongly or slightly disagreeing with the statement and 42% either strongly or slightly agreeing with it. One of the final questions on the central survey asked hunters about their satisfaction with their 2009 firearms deer hunt (Appendix I). More hunters in central region were satisfied with their hunt than dissatisfied. Forty-nine percent of hunters were either very or slightly satisfied and 31% of hunters were either very or slightly dissatisfied with their hunt last year.

The hunting experience in a specific season (for southeast hunters) or specific deer permit area (for central hunters) and the type of land hunted was used to analyze hunter's level of support for the proposed regulations (mature buck regulations, elimination of buck cross-tagging, delaying the season, antler point restriction, and a youth-only season). Hunters' responses for each survey are summarized in Tables 5-10. Few differences were found among years of hunting experience within or among samples (Tables 5-7). There were differences in the level of support among hunters when asked about delaying the start of the season until late November. Central hunters were much more opposed to delaying the start of the season than southeast hunters. At every level of experience, $\geq$80% of central hunters opposed the regulation, whereas only hunters with 40+ years of hunting experience in southeast 3B season had $\geq$80% opposition. For an antler point restriction regulation, there were no differences in support among the 3 samples, but hunters in the central region with 11-20 years hunting experience had lower levels of opposition (40%) than other levels of experience. A youth-only season was supported by more hunters (40-42%) in the central region with 1-20 years of experience than hunters with greater years of experience.

When hunters were divided into groups based on the type of land they hunted most or all of the time, there were differences concerning their level of support for mature buck regulations, elimination of buck cross-tagging, and a youth-only season (Tables 8-10). Overall, hunters who hunt on private land they own or on private land they neither own nor lease had >50% support for each sample, but there were differences among the sample in regards to leased land and public land. Southeast 3A hunters who hunt on private land they leased had much higher support for mature buck regulations (67%) than either 3B hunters (33%) or central hunters (44%). Public land hunters in the central region had higher levels of support (60%), than southeast 3B hunters (53%) for mature buck regulations. Southeast 3A hunters who leased private land had much higher levels of support for mature buck regulations (67%), than hunters who neither owned nor leased land they hunted (51%). There were differences in level of support for the elimination of buck cross-tagging within southeast 3A and among the 3 samples. Elimination of buck cross-tagging had higher levels of support for southeast 3A hunters who hunted on private land they owned (54%) and public land hunters (57%), than hunters who lease land (43%) or who neither lease nor own the land they hunt (49%). Overall, hunters in the central and southeast 3B samples had higher levels of opposition to buck cross-tagging than southeast 3A hunters. Hunters in the southeast region had higher levels of support for a youth-only season than hunters in the central region. In the southeast region, there were lower levels of support for hunters who hunt on leased land (33% and 39%), but in the central region, hunters in this category had the highest level of support among central region hunters (47%). Hunters in the central region who hunted primarily on land they owned had lower levels of support (36%) than hunters on other types of private land (47% and 45%) and public land (43%).

DISCUSSION

While a majority of hunters in both regions support mature buck regulations, they generally do not agree which alternative regulations should be enacted. Delaying the start of the season 1 week or until late November was not supported by hunters in either study area, but support for the elimination of buck cross-tagging, antler point restriction, and a youth-only season differed between areas. Southeast hunters tended to be more supportive of eliminating buck cross-tagging, instituting an antler point restriction, and conducting a youth-only season than hunters in the central region. Central region hunters, on the other hand, tended to be split between supporting or opposing the 3 regulations. Even though central region hunters did not strongly support the proposed alternative regulations, a majority did support mature buck regulations. One possible explanation is that a regulation that would be supported by more hunters was not on the questionnaire. An early antlerless season was not included in our survey, but could be considered a mature buck regulation (Fulton et al. 2006). They reported that an early antlerless season had the highest support (49.9%) of the alternative regulations tested for all hunters statewide. In 2007 and 2008, an early antlerless season was in place in all or part of the central study area. Hunters in the central study area, if given the choice of the familiar early antlerless season, may have supported this regulation over the other regulations. Further work is needed to assess central region hunter support of mature buck regulations not addressed in these surveys.

LITERATURE CITED

Fulton, D. C, L. Cornicelli, and M. D. Grund. 2006. 2005 Survey of deer hunter satisfaction and preferences for regulation changes in Minnesota. Unpublished project report. Minnesota Cooperative Fish and Wildlife Research Unit, St. Paul. 81pp.

Table 1. Age-biased sampling of southeast and central region white-tailed deer hunters, Minnesota, 2008 and 2009.

	Southeast hunters		Central hunters	
Age (yr)	% of Licenses	% of Sample	% of Licenses	% of Sample
18-29	23.4	17.1	24.4	20.0
30-39	18.4	15.4	19.6	17.4
40-49	22.0	21.8	23.7	26.0
50-59	19.1	22.3	19.6	20.5
60+	17.1	23.5	12.7	16.1

Table 2. Profile of white-tailed deer hunter respondents, Minnesota, 2008 (southeast region) and 2009 (central region).

	Southeast 3A hunters			Southeast 3B hunters			Central hunters		
Demographic Characteristics	*n*	Percent	Mean	*n*	Percent	Mean	*n*	Percent	Mean
License Type									
Regular	606	70		561	68		1,406	71	
Regular + archery	145	17		130	16		237	12	
Regular + muzzleloader	57	7		58	7		202	10	
All 3 types	56	6		80	10		136	7	
Hunting experience (yr)	828		18	801		18	1871		17
Member of hunting group									
No	715	85		717	88		1,744	88	
Yes – MDHA	30	24		30	30		153	12	
Yes – BWA	14	11		9	9		0	0	
Yes – QDMA	9	7		2	2		65	27	
Yes – Other	22	18		20	20		34	14	

Table 2. continued.

Land hunted – most or all									
Private – I own	188	22		166	20		792	40	
Private – I lease	24	3		23	3		62	3	
Private – Neither	487	58		509	62		956	49	
Public	101	12		97	12		151	8	
Hunter characteristics									
Hunt last season?									
Yes	808	98		780	98		1,898	91	
Days hunted (days)									
1	22	3	4	9	1	5	66	3	5
2	132	17		90	12		323	17	
3	152	19		118	15		227	12	
4	145	19		141	18		306	16	
5	135	17		149	19		304	16	
6	48	6		78	10		211	11	
7	146	19		84	11		152	8	
8				39	5		72	4	
9				68	9		242	13	

Table 3. Level of support for past white-tailed deer harvest regulation changes in southeast region, Minnesota, 2008.

Regulations	*n*	Strongly oppose	Slightly oppose	Neither	Slightly support	Strongly support	Don't know
Allowing youth to hunt both 3A and 3B season							
3A Hunters	809	10%	7%	12%	15%	52%	4%
3B Hunters	783	9%	8%	16%	17%	48%	3%
Shortening the 3A season by 2 days							
3A Hunters	809	31%	12%	16%	12%	25%	4%
3B Hunters	776	32%	13%	16%	13%	22%	5%
Lengthening the 3B season by 2 days							
3A Hunters	813	18%	8%	16%	14%	42%	3%
3B Hunters	774	17%	8%	17%	14%	40%	4%
Allowing antlerless harvest during 3A							
3A Hunters	816	19%	9%	11%	17%	41%	3%
3B Hunters	778	17%	10%	13%	16%	42%	2%
The current season structure							
3A Hunters	818	19%	15%	13%	18%	33%	3%
3B Hunters	779	19%	15%	14%	16%	32%	3%

Table 4. Support for select proposed white-tailed deer harvest regulations for southeast 3A (2008), 3B (2008), and central region hunters (2009), Minnesota.

Regulations		*n*	Oppose	Support	Neither
DNR should enact regulations that protect a majority of yearling bucks and increase the proportion of mature bucks in the deer population					
	Southeast 3A hunters	803	30%	52%	18%
	Southeast 3B hunters	765	27%	53%	20%
	Central hunters	1,940	31%	52%	17%
Eliminate buck cross-tagging					
	Southeast 3A hunters	822	38%	51%	11%
	Southeast 3B hunters	788	41%	50%	9%
	Central hunters	1,977	45%	45%	10%
Delay the start of firearms deer season one week					
	Southeast 3A hunters	824	55%	32%	13%
	Southeast 3B hunters	784	56%	31%	14%
	Central hunters	1,981	56%	31%	12%
Delay the start of firearms deer season until late November					
	Southeast 3A hunters	824	72%	17%	11%
	Southeast 3B hunters	786	72%	18%	10%
	Central hunters	1,978	82%	10%	8%
Institute an antler point restriction					
	Southeast 3A hunters	819	41%	45%	14%
	Southeast 3B hunters	780	37%	48%	14%
	Central hunters	1,971	46%	43%	11%
Conduct a 4-day youth-only season in mid-October					
	Southeast 3A hunters	817	29%	54%	17%
	Southeast 3B hunters	776	30%	54%	16%
	Central 2 hunters	1,971	44%	41%	15%

Table 5. Hunter support of proposed white-tailed deer harvest regulations according to hunting experience in the southeast 3A region sample, Minnesota, 2008.

	Enact regulations to protect yearling bucks		Eliminate buck cross-tagging		Delay start of season 1 week		Delay start of season until late November		Institute antler point restriction		Conduct 4-day youth-only hunt	
Years	*n*	%	*n*	%	*n*	%	*n*	%	*n*	%	*n*	%
1-10												
Oppose	85	29%	106	35%	160	54%	199	67%	114	39%	83	28%
Support	155	53%	160	54%	101	34%	58	20%	140	47%	165	55%
Neither	50	17%	33	11%	36	12%	40	13%	42	14%	51	17%
11-20												
Oppose	58	30%	72	37%	107	54%	147	75%	78	40%	63	33%
Support	104	54%	98	50%	65	33%	29	15%	94	48%	94	49%
Neither	32	16%	26	13%	25	13%	21	11%	25	13%	34	18%
21-30												
Oppose	48	33%	58	38%	85	56%	114	75%	60	40%	40	27%
Support	73	50%	72	48%	44	29%	21	14%	68	45%	86	57%
Neither	24	17%	21	14%	23	15%	17	11%	23	15%	24	16%
31-40												
Oppose	31	33%	48	51%	60	62%	74	76%	46	48%	26	27%
Support	47	50%	40	42%	28	29%	18	19%	36	38%	56	58%
Neither	16	17%	7	7%	9	15%	5	5%	14	15%	15	15%
40+												
Oppose	11	22%	21	43%	26	53%	35	71%	22	46%	11	23%
Support	22	45%	24	49%	16	33%	9	18%	18	38%	26	54%
Neither	16	33%	4	8%	7	14%	5	10%	8	17%	11	23%

Table 6. Hunter support of proposed white-tailed deer harvest regulations according to hunting experience in the southeast 3B region sample, Minnesota, 2008

	Enact regulations to protect yearling bucks		Eliminate buck cross-tagging		Delay start of season 1 week		Delay start of season until late November		Institute antler point restriction		Conduct 4-day youth-only hunt	
Years	*n*	*%*	*n*	*%*	*n*	*%*	*n*	*%*	*n*	*%*	*n*	*%*
1-10												
Oppose	75	29%	104	39%	150	56%	187	70%	96	36%	77	29%
Support	134	51%	138	51%	80	30%	47	17%	124	47%	147	55%
Neither	53	20%	27	10%	37	14%	35	13%	46	17%	43	16%
11-20												
Oppose	59	28%	87	40%	119	55%	156	73%	84	39%	72	33%
Support	110	53%	111	51%	64	30%	39	18%	105	48%	115	53%
Neither	40	19%	19	9%	33	15%	20	9%	28	13%	28	13%
21-30												
Oppose	33	23%	61	42%	72	49%	100	68%	50	35%	42	30%
Support	81	56%	67	46%	57	39%	34	23%	77	53%	77	54%
Neither	31	21%	18	12%	17	12%	13	9%	17	12%	23	16%
31-40												
Oppose	25	31%	37	45%	45	54%	64	77%	32	39%	25	31%
Support	42	53%	41	49%	26	31%	14	17%	40	48%	41	51%
Neither	13	16%	5	6%	13	15%	5	6%	11	13%	15	19%
40+												
Oppose	13	28%	21	43%	32	67%	41	84%	20	42%	10	20%
Support	27	57%	24	49%	9	19%	3	6%	20	42%	31	63%
Neither	7	15%	4	8%	7	15%	5	10%	8	17%	8	16%

Table 7. Hunter support of proposed white-tailed deer harvest regulations according to hunting experience in the central region sample, Minnesota, 2009.

	Enact regulations to protect yearling bucks		Eliminate buck cross-tagging		Delay start of season 1 week		Delay start of season until late November		Institute antler point restriction		Conduct 4-day youth-only hunt	
Years	*n*	%	*n*	%	*n*	%	*n*	%	*n*	%	*n*	%
1-10												
Oppose	227	29%	343	44%	436	55%	626	80%	364	47%	313	40%
Support	418	54%	355	45%	242	31%	83	11%	324	41%	337	43%
Neither	126	16%	88	11%	109	14%	76	10%	93	12%	132	17%
11-20												
Oppose	134	29%	202	43%	273	58%	382	80%	189	40%	198	42%
Support	254	55%	222	47%	133	28%	49	10%	231	49%	197	42%
Neither	76	16%	48	10%	68	14%	42	9%	51	11%	75	16%
21-30												
Oppose	102	34%	156	51%	176	58%	261	86%	152	50%	149	49%
Support	147	49%	129	43%	89	30%	24	8%	121	40%	115	38%
Neither	48	16%	18	6%	36	12%	17	6%	29	10%	39	13%
31-40												
Oppose	58	31%	82	44%	105	56%	157	84%	86	46%	102	55%
Support	91	49%	85	45%	63	34%	20	11%	86	46%	58	31%
Neither	36	19%	21	11%	19	10%	10	5%	15	8%	27	14%
40+												
Oppose	42	40%	50	46%	58	53%	95	87%	58	54%	60	56%
Support	44	42%	52	48%	45	41%	7	6%	37	34%	40	37%
Neither	20	19%	6	6%	6	6%	7	6%	13	12%	8	7%

Table 8. Hunter support of proposed white-tailed deer harvest regulations according to land hunted in southeast 3A region sample, Minnesota, 2008.

	Enact regulations to protect yearling bucks		Eliminate buck cross-tagging		Delay start of season 1 week		Delay start of season until late November		Institute antler point restriction		Conduct 4-day youth-only hunt	
Type of Land	*n*	%	*n*	%	*n*	%	*n*	%	*n*	%	*n*	%
Private land I own												
Oppose												
Support	49	30%	57	34%	83	49%	119	72%	71	43%	42	25%
Neither	87	54%	93	56%	62	37%	29	18%	71	43%	88	53%
	25	16%	17	10%	23	14%	17	10%	22	13%	37	22%
Private land I lease												
Oppose	5	24%	7	33%	14	67%	17	81%	9	43%	8	38%
Support	14	67%	9	43%	5	24%	3	14%	8	38%	7	33%
Neither	2	10%	5	24%	2	10%	1	5%	4	19%	6	29%
Private land neither owned\leased												
Oppose	138	30%	198	42%	277	58%	340	71%	184	39%	138	29%
Support	240	51%	232	49%	141	30%	80	17%	225	47%	258	55%
Neither	89	19%	46	10%	59	12%	58	12%	67	14%	77	16%
Public land												
Oppose	91	94%	25	28%	42	47%	64	72%	31	35%	36	35%
Support	6	6%	48	54%	36	40%	16	18%	42	48%	48	48%
Neither	0	0%	16	18%	11	12%	9	10%	15	17%	17	17%

Table 9. Hunter support of proposed white-tailed deer harvest regulations according to land hunted in southeast 3B region sample, Minnesota, 2008.

	Enact regulations to protect yearling bucks		Eliminate buck cross-tagging		Delay start of season 1 week		Delay start of season until late November		Institute antler point restriction		Conduct 4-day youth-only hunt	
Type of Land	*n*	%	*n*	%	*n*	%	*n*	%	*n*	%	*n*	%
Private land I own												
Oppose												
Support	27	19%	46	32%	84	59%	101	72%	45	32%	40	29%
Neither	77	55%	82	58%	39	27%	22	16%	78	55%	72	52%
	35	25%	14	10%	19	13%	17	12%	19	13%	26	19%
Private land I lease												
Oppose	7	39%	10	53%	13	72%	12	67%	10	53%	6	33%
Support	6	33%	7	37%	3	17%	4	22%	6	32%	7	39%
Neither	5	28%	2	11%	2	11%	2	11%	3	16%	5	28%
Private land neither owned\leased												
Oppose	142	30%	204	42%	265	54%	358	73%	187	38%	149	31%
Support	250	53%	239	49%	155	32%	85	17%	226	47%	265	55%
Neither	84	18%	48	10%	69	14%	49	10%	73	15%	69	14%
Public land												
Oppose	22	30%	36	49%	43	57%	51	68%	29	40%	17	23%
Support	39	53%	32	43%	22	29%	16	21%	39	53%	44	60%
Neither	12	16%	6	8%	10	13%	8	11%	5	7%	13	18%

Table 10. Hunter support of proposed white-tailed deer harvest regulations according to land hunted in the central region sample, Minnesota, 2009.

	Enact regulations to protect yearling bucks		Eliminate buck cross-tagging		Delay start of season 1 week		Delay start of season until late November		Institute antler point restriction		Conduct 4-day youth-only hunt	
Type of Land	*n*	%	*n*	%	*n*	%	*n*	%	*n*	%	*n*	%
Private land I own												
Oppose												
Support	243	34%	324	44%	427	58%	610	83%	335	46%	363	49%
Neither	370	51%	350	47%	230	31%	76	10%	332	45%	265	36%
	111	15%	63	9%	81	11%	52	7%	68	9%	107	15%
Private land I lease												
Oppose	17	34%	25	50%	19	38%	37	76%	20	40%	17	35%
Support	22	44%	20	40%	22	44%	8	16%	23	46%	23	47%
Neither	11	22%	5	10%	9	18%	4	8%	7	14%	9	18%
Private land neither owned\leased												
Oppose	268	31%	410	46%	518	58%	738	83%	417	47%	349	39%
Support	446	51%	390	44%	255	29%	74	8%	361	41%	396	45%
Neither	159	18%	91	10%	121	14%	79	9%	110	12%	144	16%
Public land												
Oppose	34	26%	53	40%	70	53%	103	77%	60	45%	60	45%
Support	79	60%	64	48%	45	34%	18	14%	54	41%	57	43%
Neither	19	14%	16	12%	18	14%	12	9%	18	14%	15	11%

Profile of Respondents – Zone 3A Deer Hunters

Demographic Characteristics	*n*	Percent
Did you hunt during the 2008 firearm deer season?		
Yes	808	98%
No	14	2%
Total	822	
Did you hunt another season in 2008?		
No	606	70%
Yes – Archery	145	17%
Yes – Muzzleloader	57	7%
Yes – All	56	6%
Total	864	
Which one permit area did you hunt most often during the firearms deer season?		
338	38	5%
339	28	4%
341	82	11%
342	74	10%
343	93	13%
344	57	8%
345	56	8%
346	81	11%
347	72	10%
348	72	10%
349	80	11%
Total	733	
During 2008, how many days did you hunt during the firearms season?		
1	22	3%
2	132	17%
3	152	19%
4	145	19%
5	135	17%
6	48	6%
7	146	19%
Average	4	
Total	780	

Including 2008, how many years have you been hunting deer in Zone 3?

1-10	315	38%
11-20	205	25%
21-30	154	19%
31-40	100	12%
>40	50	6%
Average	18	
Total	828	

Are you a member of an organized deer hunting group? If yes, which group(s) do you belong to?

No	715	85%
Yes	125	15%
MDHA	30	24%
BWA	14	11%
QDMA	9	7%
Other	22	18%
Total	840	

How much of your deer hunting did you do on each of the following types of land during the 2008 firearm deer hunting season?

	n	None	Some	Most	All
Land I own	840	69%	8%	8%	15%
Land I lease	834	97%	1%	1%	1%
Neither own\lease	841	27%	14%	15%	15%
Public land	840	71%	16%	4%	4%

Hunting Behavior and Attitudes

	n	Percent
If you hunted on private land that you did not own during the 3A season, to the best of your knowledge who hunted that land during the 3B season:		
Landowner and\or immediate family	230	34%
Hunter who is not a relative of the landowner	246	37%
Hunter who is a relative of the landowner	146	22%
Nobody else hunts on this property during 3B	108	16%
Don’t know	115	17%
Total	671	

If you hunted on private land that you own during the 3A season, who hunted on your property during the 3B season?

Another party that also owns the property	28	9%
Friends that do not own the property	120	38%
Family that does not own the property	98	31%
Any hunter who asks permission	25	8%
Nobody else hunts my property during the 3B season	81	25%
Don’t know	48	15%
Total	318	

Please indicate if any voluntary harvest restrictions apply to the property you hunted.

Antlerless harvest restricted but can take any buck	28	4%
Buck harvest restricted to only large antlered bucks but can take any antlerless deer	162	22%
Buck harvest restricted to only large antlered bucks and antlerless harvest restricted	22	3%
No restrictions on the type of deer that can be harvested	445	61%
Don't know	76	10%
Total	733	

Which statement best describes the number of mature bucks over the past five years in the Zone 3A area you hunted?

Fewer mature bucks now than 5 years ago	202	25%
About the same number of mature bucks now as 5 years ago	343	42%
More mature bucks now than 5 years ago	136	17%
Don't know	131	16%
Total	812	

Please indicate your level of agreement with the following statement, "If the 3A and 3B deer seasons were consolidated into one 16-day season, I (and\or my hunting party) would lose access to the property we currently hunt".

Strongly Disagree	292	36%
Slightly Disagree	70	9%
Slightly Agree	98	12%
Strongly Agee	234	29%
Don't Know	121	15%
Total	815	

Beginning in 2003, the DNR has made several changes to the 3A and 3B seasons. Please indicate your level of support for the changes that have already occurred.

	n	Strongly oppose	Slightly oppose	Neither	Slightly support	Strongly support	Don't know
Allowing youth to hunt both the 3A and 3B seasons	809	10%	7%	12%	15%	52%	4%
Shortening the 3A season by 2 days (total of 7 days)	809	31%	12%	16%	12%	25%	4%
Lengthening the 3B season by 2 days (total of 9 days)	813	18%	8%	16%	14%	42%	3%
Allowing antlerless harvest during the 3A season	816	19%	9%	11%	17%	41%	3%
The current season structure that is currently in place, which is the 7-day 3A and 9-day 3B season	818	19%	15%	13%	18%	33%	3%

Please indicate your support or opposition to the following statements about potential deer management changes. Responses of 'neither' mean you neither support nor oppose the proposed regulation and would continue to hunt your traditional location if regulations were changed. Please check one box on each line.

	n	Oppose	Support	Neither
DNR should enact regulations that protect a majority of yearling bucks and increase the proportion of mature bucks in the deer population	803	30%	52%	18%
Eliminate buck cross-tagging (both seasons)	822	38%	51%	11%
Delay the 3A deer season one week	824	55%	32%	13%
Delay the 3A deer season until late November	824	72%	17%	11%
Consolidate the 3A and 3B deer seasons	825	57%	32%	11%
Institute an antler point restriction (both seasons)	819	41%	45%	14%
Conduct a 4-day youth-only season in mid-October	817	29%	54%	17%
DNR should restore the 3A season to 9 days	820	38%	42%	20%
DNR should restore the 3B season to 7 days	820	43%	33%	24%

Profile of Respondents – Zone 3B Deer Hunters

Demographic Characteristics	*n*	Percent
Did you hunt during the 2008 firearm deer season?		
Yes	780	98%
No	14	2%
Total	794	
Did you hunt another season in 2008?		
No	561	68%
Yes – Archery	130	16%
Yes – Muzzleloader	58	7%
Yes – All	80	10%
Total	829	
Which one permit area did you hunt most often during the firearms deer season?		
338	44	6%
339	23	3%
341	111	15%
342	90	12%
343	76	10%
344	45	6%
345	52	7%
346	75	10%
347	58	8%
348	58	8%
349	92	13%
Total	724	
During 2008, how many days did you hunt during the firearms season?		
1	9	1%
2	90	12%
3	118	15%
4	141	18%
5	149	19%
6	78	10%
7	84	11%
8	39	5%
9	68	9%
Average	5	
Total	776	

Including 2008, how many years have you been hunting deer?

1-10	150	22%
11-20	176	26%
21-30	155	22%
31-40	122	18%
>40	86	12%
Average	24	
Total	689	

Including 2008, how many years have you been hunting deer in Zone 3?

1-10	283	35%
11-20	230	29%
21-30	151	19%
31-40	84	10%
>40	53	7%
Average	18	2%
Total	801	

Are you a member of an organized deer hunting group? If yes, which group(s) do you belong to?

No	717	88%
Yes	100	12%
MDHA	30	30%
BWA	9	9%
QDMA	2	2%
Other	20	20%
Total	817	

How much of your deer hunting did you do on each of the following types of land during the 2008 firearm deer hunting season?

	n	None	Some	Most	All
Land I own	823	70%	9%	8%	12%
Land I lease	824	96%	1%	1%	2%
Neither own\lease	826	22%	14%	19%	46%
Public land	822	70%	17%	4%	8%

Hunting Behavior and Attitudes

	n	Percent
If you hunted on private land that you did not own during the 3B season, to the best of your knowledge who hunted that land during the 3A season:		
Landowner and\or immediate family	257	37%
Hunter who is not a relative of the landowner	272	40%
Hunter who is a relative of the landowner	156	23%
Nobody else hunts on this property during 3B	92	13%
Don't know	124	18%
Total	688	
If you hunted on private land that you own during the 3B season, who hunted on your property during the 3A season?		
Another party that also owns the property	23	7%
Friends that do not own the property	100	31%
Family that does not own the property	99	31%
Any hunter who asks permission	37	12%
Nobody else hunts my property during the 3B season	86	27%
Don't know	60	19%
Total	319	
Please indicate if any voluntary harvest restrictions apply to the property you hunted.		
Antlerless harvest restricted but can take any buck	25	4%
Buck harvest restricted to only large antlered bucks but can take any antlerless deer	143	20%
Buck harvest restricted to only large antlered bucks and antlerless harvest restricted	22	3%
No restrictions on the type of deer that can be harvested	442	62%
Don't know	80	11%
Total	712	
Which statement best describes the number of mature bucks over the past five years in the Zone 3B area you hunted?		
Fewer mature bucks now than 5 years ago	191	24%
About the same number of mature bucks now as 5 years ago	318	40%
More mature bucks now than 5 years ago	134	17%
Don't know	146	19%
Total	789	
Please indicate your level of agreement with the following statement, "If the 3A and 3B deer seasons were consolidated into one 16-day season, I (and\or my hunting party) would lose access to the property we currently hunt".		
Strongly Disagree	258	33%
Slightly Disagree	80	10%
Slightly Agree	111	14%
Strongly Agree	218	28%

Don't Know	116	15%
Total	783	

Beginning in 2003, the DNR has made several changes to the 3A and 3B seasons. Please indicate your level of support for the changes that have already occurred.

	n	Strongly oppose	Slightly oppose	Neither	Slightly support	Strongly support	Don't know
Allowing youth to hunt both the 3A and 3B seasons	783	9%	8%	16%	17%	48%	3%
Shortening the 3A season by 2 days (total of 7 days)	776	32%	13%	16%	13%	22%	5%
Lengthening the 3B season by 2 days (total of 9 days)	774	17%	8%	17%	14%	40%	4%
Allowing antlerless harvest during the 3A season	778	17%	10%	13%	16%	42%	2%
The current season structure that is currently in place, which is the 7-day 3A and 9-day 3B season	779	19%	15%	14%	16%	32%	3%

Please indicate your support or opposition to the following statements about potential deer management changes. Responses of 'neither' mean you neither support nor oppose the proposed regulation and would continue to hunt your traditional location if regulations were changed. Please check one box on each line.

	n	Oppose	Support	Neither
DNR should enact regulations that protect a majority of yearling bucks and increase the proportion of mature bucks in the deer population	765	27%	53%	20%
Eliminate buck cross-tagging (both seasons)	788	41%	50%	9%
Delay the 3A deer season one week	784	56%	31%	14%
Delay the 3A deer season until late November	786	72%	18%	10%
Consolidate the 3A and 3B deer seasons	779	56%	33%	11%
Institute an antler point restriction (both seasons)	780	37%	48%	14%
Conduct a 4-day youth-only season in mid-October	776	30%	54%	16%
DNR should restore the 3A season to 9 days	776	40%	41%	19%
DNR should restore the 3B season to 7 days	773	43%	34%	23%

Profile of Respondents – Zone 2 Deer Hunters

Demographic Characteristics	*n*	Percent
Did you hunt during the 2009 firearm deer season?		
Yes	1,981	99%
No	15	1%
Total	1,996	
Did you hunt another season in 2009?		
No	1,406	71%
Yes – Archery	237	12%
Yes – Muzzleloader	202	10%
Yes – All	136	7%
Total	1,981	
Which one permit area did you hunt most often during the firearms deer season?		
240	725	39%
241	383	20%
243	303	16%
244	408	22%
Other	53	3%
Total	1,872	
During 2009, how many days did you hunt during the firearms season?		
1	66	3%
2	323	17%
3	227	12%
4	306	16%
5	304	16%
6	211	11%
7	152	8%
8	72	4%
9	242	13%
Average	5	
Total	1,903	
Including 2009, how many years have you been hunting deer with a firearm?		
1-10	391	20%
11-20	437	22%
21-30	438	22%
31-40	374	19%
>40	319	16%
Average	26	
Total	1,959	

Including 2009, how many years have you hunted deer in the area listed in 1C?

1-10	792	42%
11-20	475	25%
21-30	305	16%
31-40	188	10%
>40	111	6%
Average	17	
Total	1,871	

Are you a member of an organized deer hunting group? If yes, which group(s) do you belong to?

No	1,744	88%
Yes	239	12%
MDHA	153	64%
BWA	0	0%
QDMA	65	27%
Other	34	14%
Total	1,983	

How much of your deer hunting did you do on each of the following types of land during the 2009 firearm deer hunting season?

	n	None	Some	Most	All
Land I own	1962	54%	6%	11%	30%
Land I lease	1959	95%	2%	1%	2%
Neither own\lease	1965	40%	11%	9%	40%
Public land	1959	79%	13%	3%	5%

Hunting Behavior and Attitudes

If you hunted on private land that you did not own during the firearm season, to the best of your knowledge who hunted that land the firearm season:

	n	Percent
Landowner and\or immediate family	1,009	75%
Hunter who is not a relative of the landowner	396	30%
Hunter who is a relative of the landowner	464	35%
Nobody else hunts on this property during 3B	111	8%
Don’t know	47	4%
Total	1,339	

If you hunted on private land that you own during the firearm season, who hunted on your property?

Another party that also owns the property	259	27%
Friends that do not own the property	378	39%
Family that does not own the property	596	61%
Any hunter who asks permission	36	4%
Nobody else hunts my property during the 3B season	139	14%
Don’t know	33	3%
Total	971	

Please indicate if any voluntary harvest restrictions apply to the property you hunted.

Antlerless harvest restricted but can take any buck	217	12%
Buck harvest restricted to only large antlered bucks but can take any antlerless deer	360	20%
Buck harvest restricted to only large antlered bucks and antlerless harvest restricted	125	7%
No restrictions on the type of deer that can be harvested	934	52%
Don't know	158	9%
Total	1,794	

Which statement best describes your perception of deer population size over the past five years in the deer area you hunted?

Fewer deer now than 5 years ago	973	49%
About the same number of deer now as 5 years ago	707	36%
More deer now than 5 years ago	186	9%
Don't know	115	6%
Total	1,981	

Which statement best describes the number of mature bucks over the past 5 years in the deer area you hunted?

Fewer mature bucks now than 5 years ago	682	34%
About the same number of mature bucks now as 5 years ago	790	40%
More mature bucks now than 5 years ago	291	15%
Don't know	222	11%
Total	1,985	

Do you believe the deer population in the area you hunt is,

Too Low	627	32%
About right	1,078	54%
Too High	139	7%
Don't know	138	7%
Total	1,982	

Overall, how satisfied were you with your 2009 firearms deer hunt?

Very Dissatisfied	274	14%
Slightly Dissatisfied	334	17%
Neither	390	20%
Slightly Satisfied	457	23%
Very Satisfied	503	26%
Total	1,958	

Please indicate your support or opposition to the following statements about potential deer management changes. Responses of ‘neither’ mean you neither support nor oppose the proposed regulation and would continue to hunt your traditional location if regulations were changed. Please check one box on each line.

	n	Oppose	Support	Neither
In general, would you oppose or support a regulation that would increase the proportion of antlered bucks in the area you hunt most often	1,946	23%	56%	21%
DNR should enact regulations that protect a majority of yearling bucks and increase the proportion of mature bucks in the deer population	1,940	31%	52%	17%
Eliminate buck cross-tagging	1,977	45%	45%	10%
Delay the firearm deer season one week	1,981	56%	31%	12%
Delay the firearm deer season until late November	1,978	82%	10%	8%
Institute an antler point restriction (both seasons)	1,971	46%	43%	11%
Conduct a 4-day youth-only season in mid-October	1,971	44%	41%	15%
Limit the number of buck licenses	1,973	69%	21%	10%

Wildlife Health Program

Carlos Avery Wildlife Management Area
5463 – C West Broadway
Forest Lake, Minnesota 55025
(651) 296-5200

SURVEILLANCE FOR HIGHLY PATHOGENIC AVIAN INFLUENZA IN MINNESOTA'S MIGRATORY BIRDS FROM 2006–2010

Erik Hildebrand[1], Michelle Carstensen, and Erika Butler

SUMMARY OF FINDINGS

As part of a national strategy for early detection of highly pathogenic avian influenza (HPAI) in North America, the Minnesota Department of Natural Resources (MNDNR) and the United States Department of Agriculture's Wildlife Services (USDA-WS) has been conducting surveillance for the virus in waterfowl in the state since 2006. In 2010, 1,016 birds were sampled for HPAI and no positive cases were detected; however, 57 strains of low pathogenic avian influenza (LPAI) were identified. From 2006 to 2010, a total of 9,017 wild birds have been sampled for HPAI throughout Minnesota; no HPAI was detected. Nationwide, approximately 410,600 wild birds have been sampled during 2006–2010, with no evidence of disease, yet this virus remains a major concern in many parts of the world, because of its zoonotic potential and threat to the domestic livestock industry. One particular strain of HPAI, called H5N1, has affected millions of birds and hundreds of people in parts of Asia, Europe, and Africa, and concerns about this strain developing into a worldwide pandemic remain. While concern about the virus entering North America through movements of infected poultry, poultry products, or migrations of wild birds continues, large-scale surveillance in wild bird populations in the United States has been discontinued. Minnesota will continue to monitor the health of wild birds by investigating morbidity and mortality events, and screening for HPAI when appropriate.

INTRODUCTION

Avian Influenza (AI) is a viral infection that occurs naturally in wild birds, especially waterfowl, gulls, and shorebirds. It is caused by type A influenza viruses that have 2 important surface antigens, hemagglutinin (H) and neuraminidase (N), that give rise to 144 possible virus subtypes. Influenza viruses vary widely in pathogenicity and ability to spread among birds. The emergence of an Asian strain HPAI H5N1 virus in 1996 and subsequent spread of the virus in Asia, Africa, and Europe has killed thousands of wild birds and millions of domestic poultry. In 1997, HPAI H5N1 became zoonotic in Hong Kong and to-date has infected at least 552 humans in Eurasia and Africa, resulting in over 322 deaths. The migratory movements of waterfowl and other shorebirds such as from Asia into North America, heightens concern for surveillance of HPAI H5N1, although movements of domestic poultry or contaminated poultry products, both legally and illegally, are believed to be the major driving force in this virus' spread.

Following the spread of HPAI H5N1 from Asia to Europe and Africa in 2006, the National Strategic Plan for early detection of HPAI H5N1 introduction into North America by wild birds was developed. This plan outlined a surveillance strategy that focused on sampling wild bird species in North America that have the highest risk of being exposed to or infected with HPAI H5N1, because of their migratory movement patterns. This includes birds that migrate directly between Asia and North America, birds that may be in contact with species from areas in Asia with reported outbreaks, or birds that are known to be reservoirs of AI.

Recognition that ducks, geese, and swans of the order *Anseriformes* are a primary reservoir for AI, reaffirmed the need for surveillance of these populations to understand the potential for the emergence of pathogenic human and avian strains (Hanson et al. 2003). This risk concern is not focused just on domestic or wild birds in the U. S., but includes the possibility of a worldwide pandemic. Minnesota is rated as a Level 1 state by the Implementation Plan for HPAI Surveillance in the U. S., because of its historic LPAI prevalence, species-specific migratory pathways, geographic size and location, wetland habitat and amount of shoreline, and band recovery information. This means Minnesota was awarded funds to collect an assigned

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[1] Corresponding author e-mail: erik.hildebrand@state.mn.us

number of wild bird species samples for HPAI H5N1 in cooperation with the USDA-WS.

Since 2006, the MNDNR has been working with USDA-WS to collect samples from wild birds for HPAI H5N1 testing. Last year (2010) marked the final year of this surveillance program. In total, $430,000 in federal funds were awarded to Minnesota to collect approximately 7,900 wild bird samples. Sampling goals were as follows: in 2006, 2,000 samples collected under an agreement of $100,000; in 2007, 1,500 samples collected under a $100,000 agreement; in 2008, 1,600 samples collected under a $90,000 agreement; in 2009, 1,400 samples collected under a $70,000 agreement, and again in 2010, 1,400 samples collected under a $70,000 agreement.

METHODS

In 2010, the MNDNR's surveillance goals included 50 common goldeneye (*Bucephala clangula),* 50 ring-neck ducks (*Aythya collaris),* 50 mallards (*Anas platyrhynchos),* and 30 blue-winged teal (*Anas discors)* to sample during the summer months, primarily in conjunction with planned banding activities. In the fall, hunter-harvested surveillance was used to obtain samples from approximately 80 northern pintails (*Anas acuta)*, 80 mallards, 80 American green-winged teal (*Anas crecca*), 80 blue-winged teal, 50 northern shovelers (*Anas clypeata*), and 50 American wigeons (*Anas Americana*). Focus was directed more on fall surveillance, because the prevalence of AI peaks in late summer and early fall, whereas infection rates are often lower than 1% outside of this period (Halvorson et. al 1985).

The USDA-WS planned to sample a similar number of ducks within the species mentioned above, as well as 100 Canada geese (*Branta canadensis).* If sampling goals per species could not be met, other waterfowl species within the same functional group (e.g., dabblers, divers) could be sampled and counted toward the state's total.

Sampling strategies were coordinated between the MNDNR and USDA-WS to maximize access to bird species through handling of live wild-caught birds from waterfowl banding programs, fall hunter-harvested birds at various sites, agency (USDA-WS) harvested birds, and mortality/ morbidity events. Sampling consisted of obtaining 2 cotton swabs cloacal and oropharyngeal for each bird. Both swabs were placed into a vial containing prepared brain heart infusion (BHI) media. These samples were then submitted to the University of Minnesota's Veterinary Diagnostic Laboratory in St. Paul for initial screening for the virus. If positive for AI, samples were forwarded to the National Veterinary Services Laboratories in Ames, Iowa for strain-typing. Environmental (fecal) samples were also collected from 2006 to 2008 in Minnesota and submitted for HPAI testing; this sampling method was suspended in 2009.

RESULTS AND DISCUSSION

From 1 April 2010 through 31 March 2011, the MNDNR and USDA-WS collected a combined total of 1,016 samples from wild birds. This included birds that were live-caught (*n* = 417), hunter-harvested (*n* = 552), agency-harvested (*n* = 40), and mortality/morbidity events (*n* = 7) throughout Minnesota (Table 1, Figure 1). No positive cases of HPAI H5N1 were identified; however, 7 American green-winged teal, 32 mallards, and 2 northern pintails tested positive for LPAI subtype H5 (Figure 2). The testing protocol limited the screening for H5, H7, and N1 subtypes only; however, in some cases other subtypes were identified and reported elsewhere (Table 2).

According to the latest numbers from the United States Geologic Survey's (USGS) website (http://wildlifedisease.nbii.gov/ai/), approximately 40,660 birds were sampled for HPAI H5N1 in the U. S. in 2010. No positive cases were found. From 2006 to 2010, over 410,000 wild birds have been sampled for HPAI H5N1 throughout the U. S., including 9,017 in Minnesota, and no HPAI H5N1 has ever been detected. Despite multiple wild bird mortality events in Asia and Europe, it does not appear that HPAI H5N1 has been introduced via migratory birds into the U. S.

From 2006 to 2010, of the 9,017 samples collected in Minnesota, there were 146 positive LPAI H5 subtypes and 7 LPAI N1 subtypes (Table 3). Approximately 26% of the total samples collected were in the summer months (presumably from resident/local birds), while 48% were from fall hunter-harvested birds that were migrating into Minnesota.

There has been additional AI research conducted by the Southeastern Cooperative Wildlife Disease Study (SCWDS) since 2006 in northwestern Minnesota. Primary focus areas include Roseau River Wildlife Management Area (WMA), Thief Lake WMA, and Agassiz National Wildlife Refuge (NWR). Sampling has also occurred at lakes around the Bemidji and Fosston areas. From 2006 to 2010, SCWDS sampled over 9,200 ducks, and based on virus isolation in embryonating chicken eggs, found 1,254 positive samples, of which 30 were LPAI H5 subtypes, and 20 LPAI H7 subtypes (Table 4). Throughout all testing, there was no HPAI H5N1 virus detected. Sampling in Minnesota will continue by SCWDS at least through 2013.

Other AI research has been conducted throughout the state by University of Minnesota (UMN) since 2008, mostly in conjunction with MNDNR's sampling efforts. From 2008 to 2010, the UMN sampled over 3,100 ducks, have analyzed 3,092 to-date, and used both a plaque reduction neutralization test (PRNT) and a virus isolation (VI) test; 72 LPAI isolates have been detected. Sub-types isolated by species to-date include LPAI H1N1, H6N1, H1N1, H3N8, and H3N2 in mallards; LPAI H4N8, H4N2, H3N8, H3, and H11N9 in blue-winged teal; and LPAI H4N8 in ring-necked ducks. No H5 or H7 LPAI or HPAI has been encountered to-date.

Federal AI funding for most wild bird surveillance in the U. S. is no longer available; however, federally-funded efforts to monitor for the disease in domestic poultry will likely continue. Even though USDA-WS and MNDNR will no longer be conducting large-scale surveillance for HPAI H5N1 in wild birds, AI samples will continue to be collected at all mortality/morbidity events involving wild birds in the state.

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LITERATURE CITED

Halvorson, D. A., C. J. Kelleher, and D. A. Senne. 1985. Epizootiology of avian influenza: effect of season on incidence in sentinel ducks and domestic turkeys in Minnesota. Applied and Environmental Microbiology 49:914-919.

Hanson, B. A., D. E. Stallknecht, D. E. Swayne, L. A. Lewis, and D. A. Senne. 2003. Avian influenza viruses in Minnesota ducks during 1998-2000. Avian Diseases 47:867-871.

Table 1. Bird species sampled in Minnesota for highly pathogenic avian influenza H5N1 by the Minnesota Department of Natural Resources (MNDNR) and United States Department of Agriculture-Wildlife Services (USDA-WS), 2010. These wild birds were live-caught, hunter-harvested, agency-harvested, or subjects of morbidity/mortality events.

Agency	Species sampled	*n*
MNDNR	Black duck (American)	2
	American green-winged teal	86
	American coot	6
	American wigeon	26
	Blue-winged teal	84
	Common goldeneye	50
	Common merganser	3
	Gadwall	4
	Greater scaup	2
	Hooded merganser	1
	Lesser scaup	40
	Mallard	218
	Northern pintail	39
	Northern shoveler	35
	Redhead	6
	Ring-necked duck	127
	Ruddy duck	1
	Wood duck	25
	Total	755
USDA-WS		
	American green-winged teal	2
	American wigeon	4
	Blue-winged teal	11
	Canada goose	84
	Double-crested Cormorant	57
	Mallard	82
	Northern shoveler	2
	Wood duck	19
	Total	261
Grand Total		1016

Table 2. Subtyping results of bird species sampled in Minnesota by the Minnesota Department of Natural Resources and United States Department of Agriculture-Wildlife Services, 2010.

Species	H10N7	H3N2	H3N8	H4N6	H5N2	H6N1	N2	N4	N8	TOTAL
American green-winged teal				1		1	1	1		4
Mallard	2	1		1	3	1	1		1	10
Northern pintail				1						1
Wood duck			1							1
Total	2	1	1	3	3	2	2	1	1	16

Table 3. Low pathogenic avian influenza strains detected in wild birds sampled in Minnesota by the Minnesota Department of Natural Resources and United States Department of Agriculture's Wildlife Services, 2006–2010.

Year	TotaL samples	Species	LPAI H5	LPAI N1
2006	2,065			
		American green-winged teal		1
		Northern pintail	1	
		Ring-necked duck	1	
		Total	2	1
2007	2,264			
		American green-winged teal	8	1
		American wigeon	5	
		Blue-winged teal	6	
		Lesser scaup	3	
		Mallard	8	1
		Northern pintail	9	1
		Northern shoveler	1	
		Total	40	3
2008	2,263			
		American green-winged teal	4	
		American wigeon	4	
		Bufflehead	1	
		Blue-winged teal	4	
		Gadwall	2	
		Lesser scaup	1	
		Mallard	24	1
		Northern pintail	2	
		Northern shoveler	1	
		Total	43	1
2009	1,409			
		American green-winged teal	3	
		American wigeon	1	
		Blue-winged teal	5	1
		Mallard	2	1
		Northern pintail	4	
		Ring-necked duck	4	
		Wood duck	1	
		Total	20	2
2010	1,016			
		American green-winged teal	7	
		Mallard	32	
		Northern pintail	2	
		Total	41	0
Total	9,017	Grand Total	146	7

Table 4. Avian influenza samples collected in Minnesota by the Southeastern Cooperative Wildlife Disease Study , Athens, Georgia, 2006–2010.

Year	Total samples	Positive (%)	# of Subtypes	LPAI H5	LPAI H7
2006	130	17 (13%)	4	0	0
2007	2,441	222 (9%)	27	2	15
2008	2,452	438 (18%)	31	16	2
2009	2,341	238 (10%)	Pending [a]	6	3
2010	1,896	339 (18%)	Pending [a]	6	0

[a] All H5 and H7 viruses recovered during these years have been tested by National Veterinary Services Laboratories.

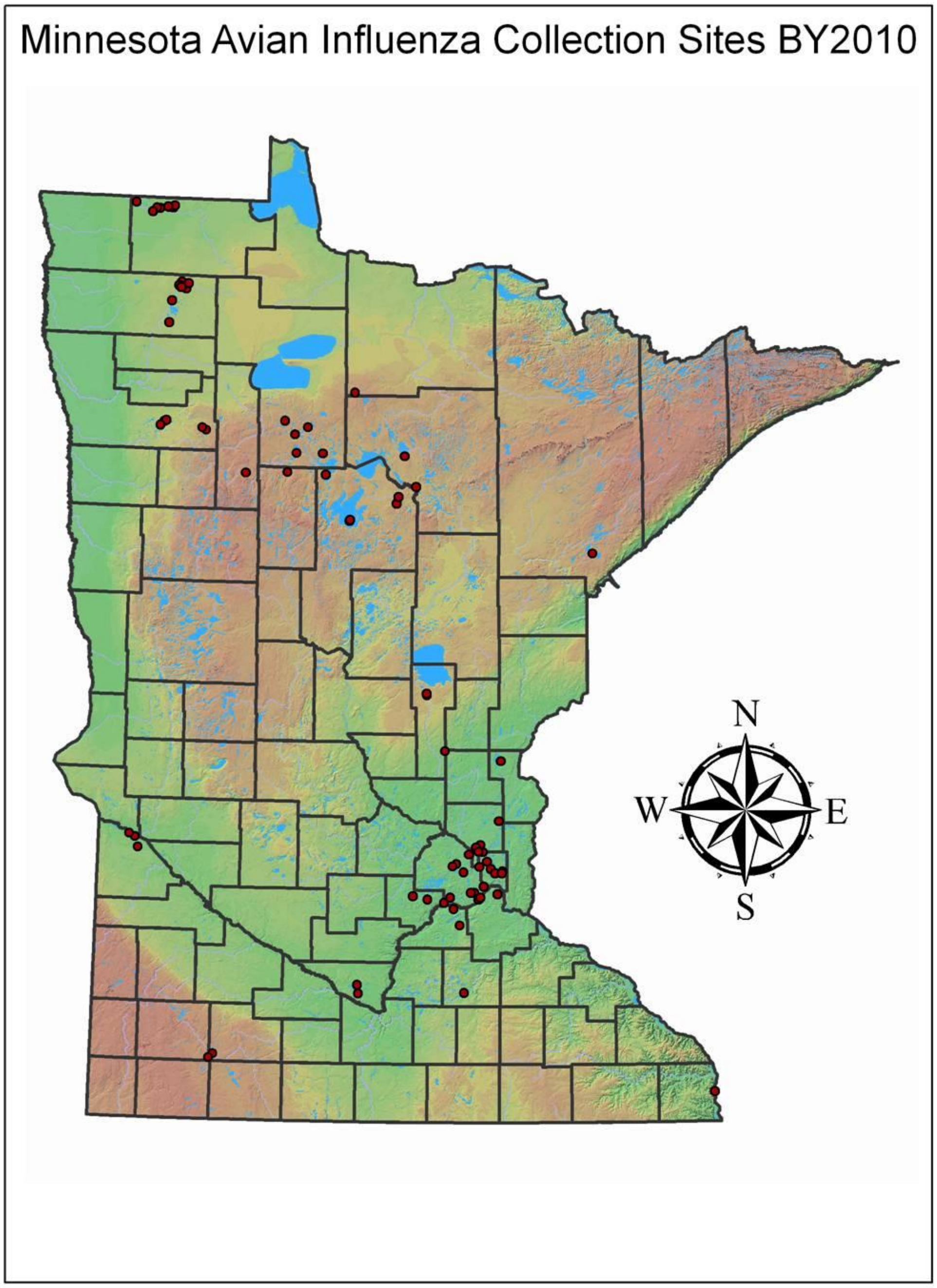

Figure 1. Sites in Minnesota from which wild bird samples (*n* = 1,016) were collected and tested for highly pathogenic avian influenza by the Minnesota Department of Natural Resources and United States Department of Agriculture's Wildlife Services, 2010.

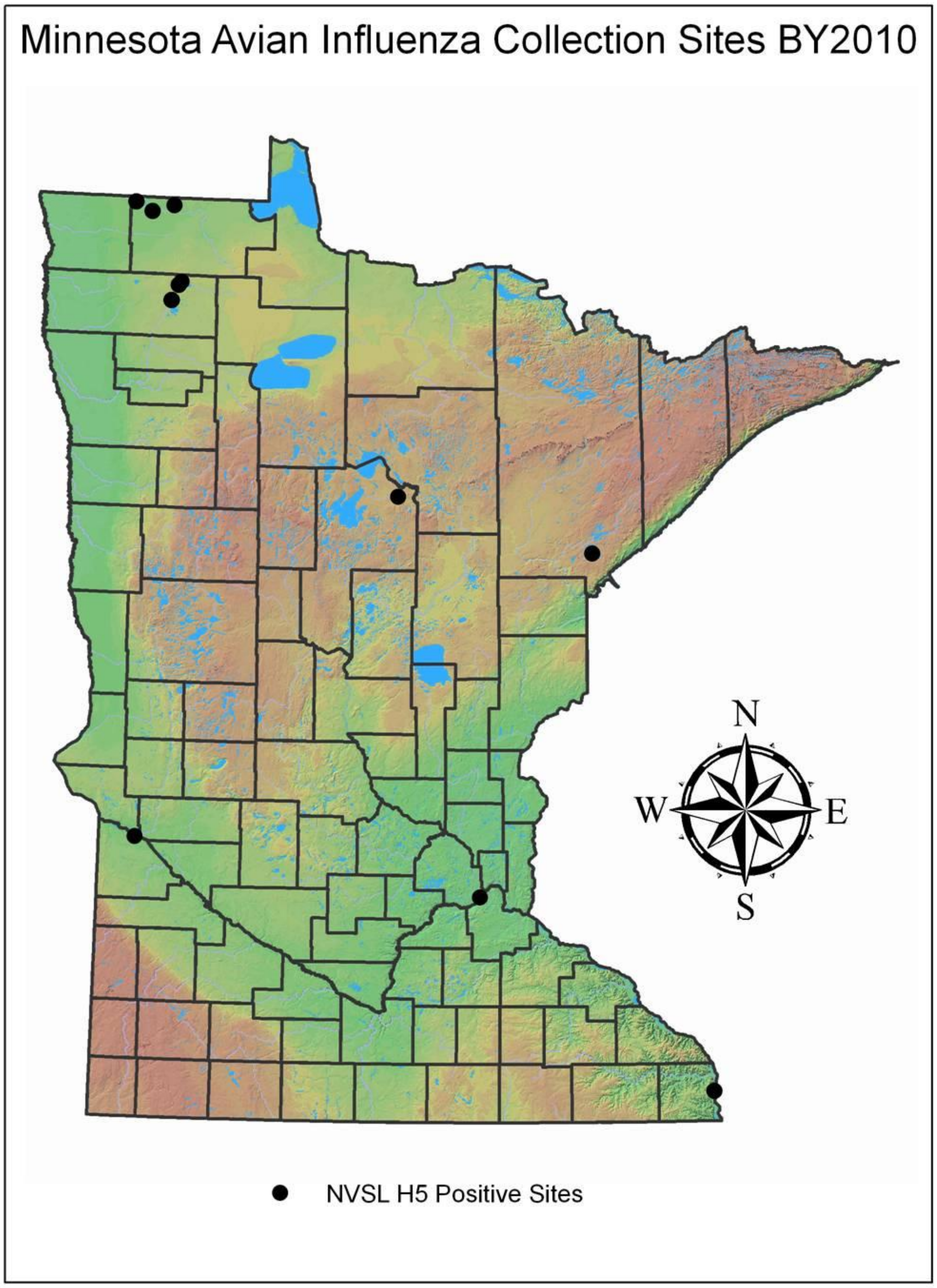

Figure 2. Collection sites in Minnesota where a low pathogenic avian influenza H5 strain was detected among the waterfowl (*n* = 41) sampled by the Minnesota Department of Natural Resources and United States Department of Agriculture's Wildlife Services, 2010.

PRELIMINARY RESULTS FROM THE 2010 MOOSE HEALTH ASSESSMENT PROJECT

Erika Butler[1], Michelle Carstensen, and Erik Hildebrand

SUMMARY OF FINDINGS

This project represents the second phase of an assessment on the overall health of hunter-harvested moose (*Alces alces*) in northeastern Minnesota (MN), which began in 2007. The purpose of this project is to: (1) continue to screen hunter-harvested (and presumably healthy) moose from 2010 to 2012 for select disease agents to monitor changes in disease incidence or prevalence over time, (2) assess the clinical impacts of liver fluke (*Fascioloides magna*) infection on moose, and (3) determine the frequency of histological lesions consistent with brainworm (*Parelaphostrongylus tenuis*) infection. Samples were collected from 130 moose in 2010. Moose (*n* = 110) were screened for West Nile virus, eastern equine encephalitis, malignant catarrhal fever, borreliosis (*Borrelia burgdorfer*i), anaplasmosis (*Anaplasma phagocytopila*, formerly *Ehrlichia phagocytophila*) and 6 serovars of leptospirosis. There was evidence of exposure to West Nile Virus (29.1%), malignant catarrhal fever (3.6%), borreliosis (21.8%), and leptospirosis (0.9–9.2%). Whole livers and brains were collected and examined grossly and histologically for evidence of brainworm and liver flukes; both parasites were documented. Full serum chemistry profiles (*n* = 95) were used to determine if there is a correlation between liver fluke damage and serum liver enzymes. Whole blood samples (*n* = 109) were submitted for evaluation for tick-borne illnesses; anaplasmosis and piroplasma infections were documented.

INTRODUCTION

Several lines of evidence suggest the moose population in northeastern MN is declining. Since 2002, annual survival and reproductive rates were substantially lower than documented elsewhere in North America (Lenarz et al. 2007). Further, modeling based on these vital rates indicated that the population has been declining by approximately 15% per year since 2002 (Lenarz et al. 2010). Likewise, recruitment and twinning rates (1%) have steadily declined since 2002; recruitment was reported at its lowest rate in 2011. In 2011, the bull:cow ratio (0.64) was at the lowest value in the last 27 years. Lastly, hunter success rates have steadily decreased, from 84% in 1993 to 51% in 2010 (Lenarz 2011).

Previous and ongoing research has been unable to determine proximate and ultimate cause(s) of non-hunting moose mortality and the possible related impacts to the long-term viability of the northeastern MN population. In 2007, the MN Department of Natural Resources (MNDNR) began a 3-year moose health assessment project to determine which diseases northeastern MN moose are being exposed to and to establish baseline hepatic mineral levels. Results indicated that hunter-harvested moose in northeastern MN have been exposed to a variety of disease agents such as West Nile virus (WNV), eastern equine encephalitis (EEE), malignant catarrhal fever (MCF), anaplasmosis, borreliosis, and leptospirosis (Butler et al. 2010). While these findings were illuminating, there remained some key factors, the importance of which, we have been unable to determine, including: (1) the role liver damage (due to liver flukes) plays in non-hunting mortality, 2) the impact of arboviruses and how their incidences may be affected by changing climate, and (3) the impact of brainworm on moose survival, due to the difficulty in interpreting brain lesions caused by this parasite. To begin addressing these key factors, a second phase of the moose health assessment project was launched in 2010.

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[1] Corresponding author email: erika.butler@state.mn.us

Murray et al. (2006) concluded that moose in northwestern MN were dying from high liver fluke loads. However, assessing the extent of liver damage caused by flukes can be subjective. A large portion of the liver can be destroyed, yet have no clinical impact to the health of the animal. In order to determine if liver damage caused by flukes has clinical implications, serum liver enzymes should be evaluated. Beginning in 2009, we asked hunters to collect whole livers for evaluation and ranking of their liver fluke load by a board-certified veterinary pathologist. Results from this pilot year of liver examinations indicated that 35% of livers had fluke-induced lesions with some having nearly 100% of the liver parenchyma affected (Butler et al. 2010). However, poor blood collection techniques prevented assessment of the clinical impacts of the damage caused by the liver fluke infections. In 2010, we asked hunters to alter their blood collection strategies and began collecting both the whole liver and assessing serum liver enzymes, with the goal of determining whether results of gross evaluation of the liver correlated with liver function.

Our moose health assessment during 2007–2009 indicated that our moose are being exposed to a variety of arboviruses, including EEE, WNV, borreliosis, and anaplasmosis (Butler et al. 2010). As climate changes, the density annd distribution of capable vectors is expected to change as well. Climate is known to play a key role in determining the geographical and temporal distribution of arthropods, characteristics of arthropod lifecycles, dispersal patterns of associated arboviruses, evolution of arboviruses, and the efficiency with which they are transmitted from arthropods to vertebrate hosts (Gould and Higgs 2009). For example, there has been a substantial increase in tick-borne encephalitis in Sweden since the mid-1980s related to milder winters and earlier arrival of spring (Lindgren and Gustafson 2001). In Phase 2 of the moose health assessment study, serum will be screened for these arboviruses and a few additional select disease agents. Combined with results from our 2007–2009 sampling, we will have 6 years of data on the incidence of arbovirus exposure in our moose herd to evaluate any significant trends relative to fluctuations in climate.

Diagnostics have shown that moose are dying from brainworm in MN. It is also known that moose are able to survive low-dose infections of brainworm and even develop immunity to subsequent infections (Lankester 2002). Researchers have hypothesized that brainworm was responsible for historic declines in moose populations (Karns 1967, Prescott 1974, Lankester 1987), but it is questionable whether brainworm represents a major threat to the northeastern MN population. In 2008, we began collecting whole brains from hunter-harvested moose to determine the frequency of brain lesions consistent with past brainworm infections in presumably healthy moose. These data would allow for better interpretation of migration tracts and could prevent pathologists from wrongly assigning brainworm as the cause of death based solely on the presence of migration tracts. We will continue to collect whole brains to increase our sample and quantify the number of presumably healthy moose have migration tracts.

METHODS

Hunters (both tribal and state) were asked to collect whole livers, blood, hair, and a central incisor. State hunters were only allowed to harvest bulls while some tribal hunters were able to take either bulls or cows. Wildlife Health Program staff provided a presentation and instructions relative to the moose health assessment project at the mandatory MNDNR Moose Hunt Orientation Sessions and at tribal natural resource offices. Hunters were given a sampling kit with instructions at the sessions. Post-harvest, the sampling kits were dropped off at official registration stations by the hunters at the time of moose registration. We asked hunters to locate their kill-sites on maps we provided.

The MNDNR provided hunters with all the equipment needed for sample collection and preservation. Sampling kits included a cooler, 1-60-cc syringe for blood collection, 6-15-cc serum separator tubes, 2-5-cc ethylenediaminetetraacetic acid (EDTA) blood tubes for whole blood collection, 1 heavy-duty bag for liver storage, 2 coin envelopes for the tooth and hair collected, data sheet, protocol, Sharpie marker, 1 pair of large vinyl gloves, and 1 icepack.

Hunters collected blood using the 60-cc syringe after incising the jugular vein as soon after death as possible and recorded time of death and blood collection. Blood was placed in serum-separator tubes and in an EDTA tube and kept cool until they were delivered to official MNDNR registration stations or tribal natural resource offices. Livers were placed in heavy-duty, pre-labeled bags.

At the stations or offices, serum-separator tubes were centrifuged and the serum decanted. Blood spinning time was recorded. Portable refrigerators were located in advance at the registration stations to maintain the tissue samples. One whole blood sample (EDTA tube) and 1 mL of serum were refrigerated and submitted every 2–3 days to the University of MN(UMN)-College of Veterinary Medicine-Clinical Pathology Laboratory for a full large-animal serum chemistry profile. The remaining whole blood sample was submitted every 2–3 days to the UMN-Department of Entomology for testing for tick-borne illnesses. Remaining serum and the whole livers were frozen. Whole brains were removed with the hunter's permission and placed in formalin. The serum, whole liver, and whole brains were submitted to the UMN Veterinary Diagnostic Laboratory (UMN VDL, St. Paul, MN).

Serum was tested for WNV and EEE with a plaque reduction neutralization test (PRNT) at the National Veterinary Services Laboratory (NVSL) in Ames, Iowa. Serum was screened for leptospirosis (microscopic agglutination test), borreliosis (immunofluorescence assay), anaplasmosis (card test), and MCF via peroxidase-linked assay (PLA) with positive PLA tests further tested with a virus neutralization test (VN) at the UMN VDL. The livers were ranked by a board-certified veterinary pathologist based on parenchymal damage due to liver flukes; ranking included no fluke-induced lesions (no evidence of fluke migration), mild infection (< 15% of liver parenchyma is affected with mild prominence/fibrosis of bile ducts and few smaller nodules characterized by peripheral fibrosis and central presence of opaque brown pasty material), moderate infection (15–50% of the liver parenchyma affected by nodules and fibrosis), and marked infection (51–100% of the liver parenchyma affected with deformation of the entire liver by larger nodules with widespread fibrosis). Brains were examined histologically with 4 complete coronary brain, cerebellum, and brain stem sections processed from each moose. An average of 25 histological slides per animal were examined, including the frontal, temporal, parietal, and occipital lobes and the basal nuclei, thalamus, mesencephalon, and brain stem. Central incisors of moose were submitted to Mattson's Laboratory (Milltown, Montana) for aging by cementum annuli (Sergeant and Pimlott 1959).

RESULTS AND DISCUSSION

Samples from 130 moose (125 males, 4 females, 1 sex unknown) were submitted for diagnostic screening in 2010 (Figure 1). Exact age was determined for 124 of these moose (median = 4, range = 1–11 years old).

EEE

One hundred and ten serum samples were tested for EEE and all were negative. The absence of EEE exposure in these moose was unexpected as an average exposure rate of 6.1% of the population was documented during Phase 1 of this study (Butler et al. 2010).The continued surveillance for EEE in Phase 2 of this study may provide greater insight into the annual variation in apparent disease prevalence. Mosquitoes spread EEE, which can cause neurologic signs and often death. It poses a greater mortality threat for most species than WNV, although the effects of EEE infection have not been studied in moose.

WNV

Evidence of exposure to WNV was detected in 32 of 110 (29.1%) moose. These results were similar to those reported during the first 3 years of the study (34.8%, Butler et al. 2010). Positive results indicated that animals were exposed to the WNV, but does not necessarily indicate illness. A titer that is greater than 100 is considered a very strong positive and means that the serum was able to neutralize nearly 100% of the virus. Multiple animals had titers ≥100. Little is known about the effects of WNV in moose.

MCF

Evidence of exposure to MCF was detected in 4 of 110 (3.6%) moose sampled with PLA. Follow-up testing with VN was negative for 2 of the 4, and the remaining 2 were unsuitable for testing. These PLA results are markedly lower than what we reported from 2007 to 2009 (35%, Butler et al. 2010). The PLA test is more sensitive than VN, meaning it is much better at identifying true positives, whereas VN is more specific and thus better at identifying true negatives. The PLA reacts with multiple gammaherpes viruses (e.g., wildebeest strain of MCF, sheep strain of MCF, deer strain of MCF). A PLA positive does not indicate the strain of exposure. The VN test only screens for the wildebeest strain (which is exotic to the U.S.) and would be negative if other strains are present. This means a sample that was positive on PLA and negative on VN was likely exposed to a gammaherpes virus, but not the wildebeest strain.

We have been collaborating with researchers (Dr. Hong Li, Washington Animal Disease Diagnostic Laboratory) to determine the strain of MCF exposure in the northeastern MN moose population. To date, all attempts at strain-typing have been unsuccessful.

Gammaherpes viruses have been documented to cause serious illness and death in moose and other ruminants. The clinical symptoms can mimic brainworm infection, including neurological deficits, blindness, and thrashing on the ground prior to death. While infection with MCF frequently results in death, carrier status can occur and is identified with serology. Zarnke et al. (2002) found serologic evidence of exposure in numerous species across Alaska and reported 1% prevalence in moose.

Anaplasmosis

No evidence of exposure to anaplasmosis was detected in moose screened for this disease in 2010 (*n* = 100). These results are similar to the results of 2007−2009 screening (1/319, 0.3%; Butler et al. 2010), indicating that exposure to this bacterium is likely occurring, albeit at a low rate.

Moose are thought to be susceptible to infection with *A. phagocytophilum*. In Norway, anaplasmosis was diagnosed in a moose calf, which displayed apathy and paralysis of the hind-quarters (Jenkins et al. 2001). This moose was concurrently infected with *Klebseilla* pneumonia, to which the calf's death was attributed, though the *Klebseilla* infection was most likely secondary to and facilitated by the primary infection with *A. phagocytophilum*. In sheep, this disease produces significant effects on the immunological defense system, increasing their susceptibility to disease and secondary infections (Larsen et al. 1994).

Borreliosis

Evidence of exposure to borreliosis was detected in 24 of the 110 (21.8%) moose sampled. These results are similar to results from 2007 to 2009 (22.9%, Butler et al. 2010).

Borreliosis is a tick-borne bacterial disease that is maintained in a wildlife/tick cycle involving a variety of species, including mammals and birds. While evidence of natural infection in wildlife exists, there has been no documentation of clinical disease or lesions reported in wildlife species.

Leptospirosis

A total of 110 samples were screened for 6 serovars of *Leptospira interrogans*. Results per serovar are as follows:

- *L. bratislava*:
  - 1/109 (0.9%)
- *L. canicola*:
  - 0/109
- *L. grippothyphosa*:
  - 1/109 (0.9%)
- *L. hardjo*:
  - 0/109
- *L. interrogans* serovar *icterohaemorrhagicae*:
  - 0/109
- *L. pomona*:
  - 10/109 (9.2%)

While the prevalences are lower for most of the serovars compared with data from 2007–2009, the prevalence of *L. pomona* actually increased from 6.9% to 9.2% (Butler et al. 2010). Leptospirosis is a bacterial disease that can infect a wide variety of mammals, both domestic and wild. Moose could be at an increased risk for leptospirosis, as it is often propagated by mud and water contaminated with urine, not uncommon in moose habitat.

General Tick-Borne Illness Screening

Whole blood samples from 109 moose were submitted to the UMN Department of Entomology, where we are collaborating with Dr. Ulrike Munderloh to determine if hunter-harvested moose are infected with tick-borne illnesses. Samples were screened with a variety of polymerase chain reaction (PCR) techniques. Preliminary results indicate that 10.1% of the moose were infected with anaplasmosis and 32.1% were positive for prioplasma primers. Further analysis is pending.

Brain Histopathology

Forty whole brains were collected in 2010. Since 2008, a total of 87 whole brains have been collected and examined. No lesions were found in 74 (85.1%) of the brains, 9 (10.3%) had lymphocytic infiltration (unspecific chronic inflammatory lesion), and 4 (4.6%) had lesions consistent with larval migration tracts (mild to moderate meningitis, axonal degeneration, and secondary demyelination).

Whole Liver Evaluation

In 2010, 108 whole livers were collected. Combined with livers collected in 2009 (n = 57), 165 livers have been submitted for gross examination. Of the 165 livers examined, 120 (72.7%) had no fluke-induced lesions, 28 (16.9%) had mild infection, 15 (6.7%) had moderate infection, and 6 (3.6%) had marked infection. Collection of whole livers will continue in 2011–2012. Additionally, beginning in 2010, serum was submitted for a serum chemistry profile in an attempt to correlate serum liver enzyme levels with the level of fluke-induced damage. These results have not yet been analyzed.

Serum Chemistries

A total of 95 serum samples were submitted for a full large animal serum chemistry profile. Analysis of these results is pending. The purpose of collecting these data is to determine if there is a correlation between the liver ranking and serum liver enzymes, as well as to establish baseline "normals" for animals in this population.

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LITERATURE CITED

Butler, E., M. Carstensen, E. Hildebrand, J. Giudice, R. Wright, and M. Schrage. 2010. Preliminary results from the 2007-2009 moose herd health assessment project. Pages 150-163 *in* J. S. Lawrence, R. O. Kimmel, and M. S. Lenarz, eds. Summaries of wildlife research findings, 2009. Minnesota Department of Natural Resources, Wildlife Populations and Research Unit, St. Paul.

Gould, E. A., and S. Higgs. 2009. Impacts of climate change and other factors on emerging arbovirus disease. Royal Society of Tropical Medicine and Hygeine 103:109-121.

Jenkins, A., K. Handeland, S. Stuen, L. Schouls, I. Van de Pol, R. T. Meen, and B. E. Kristiansen. 2001. Ehrlichiosis in a Moose calf in Norway. Journal of Wildlife Diseases 37:201-203.

Karns, P. D. 1967. *Pneumostrongylus tenuis* in deer in Minnesota and implications for moose. Journal of Wildlife Management 32:299-303.

Lankester, M. W. 1987. Pests, parasites and diseases of moose (*Alces alces*) in North America. Swedish Wildlife Research Supplement 1:461-489.

Lankester, M. W. 2002. Low-dose meningeal worm (*Paralaphostrongylus tenuis*) infections in moose (*Alces alces*). Journal of Wildlife Disease 38:789-795.

Larsen, H. J. S., G. Overnes, H. Waldeland, and G. M. Johansen. 1994. Immunosuppression in sheep experimentally infected with *Ehrlichia phagocytophila*. Research in Veterinary Science 56:216-224.

Lenarz, M. S. 2011. 2011 Aerial moose survey. Unpublished report. Minnesota Department of Natural Resources, St. Paul

Lenarz, M. S., J. Fieberg, M. W. Schrage, and A. J. Edwards. 2010. Living on the edge: viability of moose in northeastern Minnesota. Journal of Wildlife Management 74:1013-1023.

Lenarz, M. S., M. W. Schrage, A. J. Edwards, and M. E. Nelson. 2007. Moose population dynamics in northeastern Minnesota. Pages 346-348 *in* M. W. DonCarlos, R. O. Kimmel, J. S. Lawrence, and M. S. Lenarz, eds. Summaries of wildlife research findings, 2005. Minnesota Department of Natural Resources, Wildlife Populations and Research Unit, St. Paul.

Lindgren, E., and R. Gustafson. 2001. Tick-born encephalitis in Sweden and climate change.

The Lancet 358:16-18.

Murray, D. L., E. W. Cox, W. B. Ballard, H. A. Whitlaw, M. S. Lenarz, T. W. Custer, T. Barrnett, T. K. Fuller. 2006. Pathogens, nutritional deficiency, and climate influence on a declining moose population. Wildlife Monographs 166.

Prescott, W. H. 1974. Interrelationships of moose and deer of the genus *Odocoileus*. Naturaliste Canadien 101:493-504.

Sergeant, D. W., and D. H. Pimlott. 1959. Age determinationin moose from sectioned incisor teeth. Journal of Wildlife Management 23:315-321.

Zarnke, R. L., H. Li, and T. B. Crawford. 2002. Serum antibody prevalence of malignant catarrhal fever viruses in seven wildlife species from Alaska. Journal of Wildlife Diseases 38:500-504.

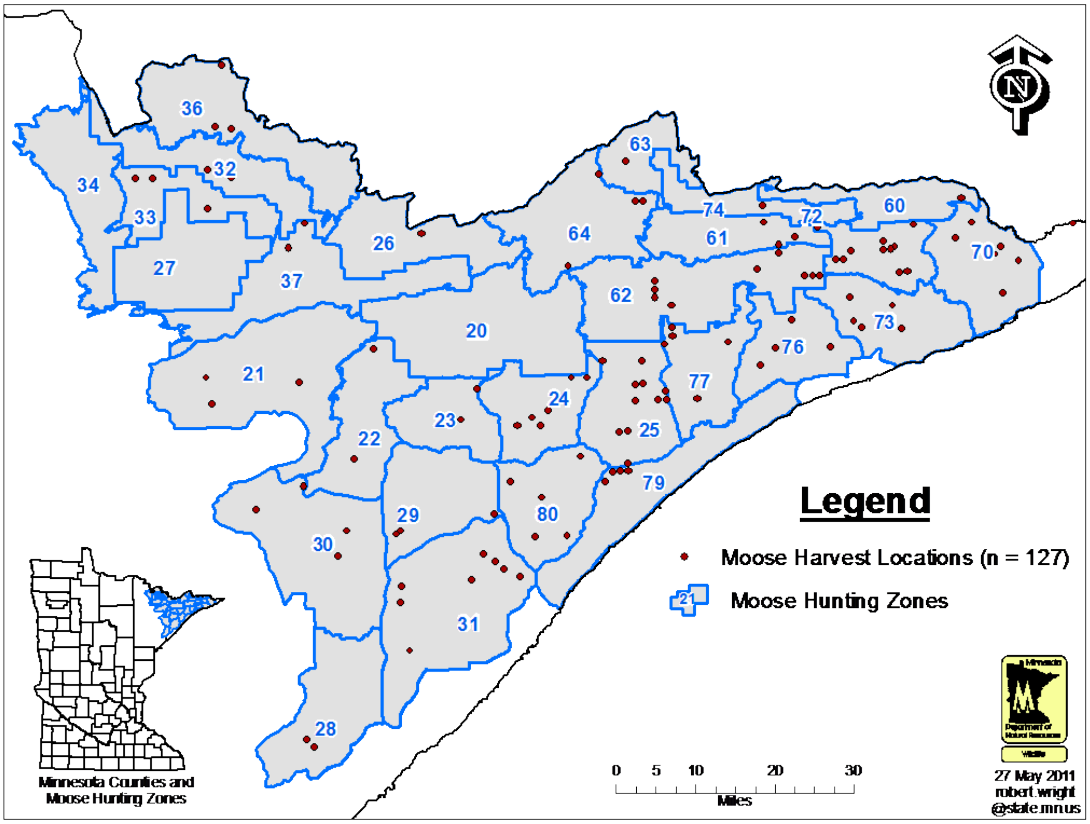

Figure 1. Locations of hunter-harvested moose (*n* = 127) included in the 2010 moose health assessment project, northeastern Minnesota.

MINNESOTA GRAY WOLF DISEASE SCREENING AND MORPHOLOGY

Michelle Carstensen[1], Erika Butler, Dan Stark, Erik Hildebrand, and John Erb

SUMMARY OF FINDINGS

A total of 206 wolves (*Canis lupus*) were included in the first year of a 2-year study to document the apparent prevalence of diseases and parasites in Minnesota's wolf population, as well as provide insight into their genetic makeup. Our results indicated serologic exposure of wolves to 8 diseases: canine parvovirus (75.4%), canine adenovirus (76.1%) canine distemper virus (16.4%), eastern equine encephalitis (2.8%), West Nile virus (13.7%), heartworm (9.6%), Lyme (94.5%), and neosporosis (83.6%). Parasites were discovered in 14.5% of fecal samples examined. Genetic analyses are pending.

INTRODUCTION

Minnesota's gray wolf population is currently managed under the authority of the U.S. Fish and Wildlife Service. Wolves in Minnesota are classified as threatened under the federal Endangered Species Act. The Minnesota Department of Natural Resources (MNDNR) anticipates a decision to delist gray wolves in Minnesota within the next year. Following that ruling, wolves will be managed in Minnesota by state statute, rule, and under a wolf management plan. This plan is designed to protect wolves and monitor the population while giving owners of livestock and domestic pets more flexibility in addressing wolf depredation. A primary component of monitoring the wolf population is to understand what diseases and parasites might be impacting them. Furthermore, the collection of morphological and genetic data will add current and more spatially comprehensive data to the ongoing debates regarding the genetic identity of wolves in Minnesota.

There are a number of diseases and parasites known to affect wolves that can have population-level impacts. Most notably, relatively high prevalence of canine parvovirus (CPV) has been reported in Minnesota and could be adversely impacting pup survival and limiting population growth (Mech et al. 2008). Other diseases, including canine distemper, adenovirus, and parasites may also kill infected wolves and impact population performance. Furthermore, some diseases, such as neosporosis, are of particular concern to livestock producers; gaining a more thorough understanding of the prevalence and distribution of this disease may benefit wolf management strategies.

There is uncertainty in the taxonomic and genetic identity of wolves in the Great Lakes Region (Leonard and Wayne 2008, Mech 2008, Koblmuller et al. 2009, Nowak 2009, Schwartz and Vucetich 2009, Wheeldon and White 2009, Mech 2010, vonHoldt et al. 2011). No one has attempted to relate Minnesota wolf morphology with genetics. This portion of the project will systematically assess both genetic and morphological characteristics of a large sample of wolves in Minnesota. Relating wolf morphology to genetics should help determine the taxonomic identity of wolves throughout Minnesota and reveal any potential geographic patterns of species or subspecies occurrence. Information will have both scientific and management value, and depending on the timing of the results, may better inform ongoing efforts to delist the wolf in Minnesota.

METHODS

The MNDNR entered into a contract with the United States Department of Agriculture (USDA)-Animal and Plant Health Inspection Service (APHIS)-Wildlife Services (WS) to collect biological samples from all dispatched wolves immediately after death. Researchers within the MNDNR, the United States Geological Survey (USGS), and tribal authorities that capture and radiocollar wolves also were involved in sample collections. Conservation officers and Area Wildlife staff assisted in collecting samples from vehicle-killed wolves. All key personnel were trained in proper sample collection and handling, as well as recording morphological measurements. Sampling kits provided to data collectors included the following items: soft-sided cooler, 1-20cc syringe for blood collection, 6-10-cc serum tubes for blood storage, 1-5-cc EDTA tube for whole blood, 1 whirlpak for fecal collection, 1 ear punch, 1 FTA card, 1-2-mL vial with 95% ethanol, 1-2-mL vial with desiccant, tape measure, caliper, data sheet, protocol, Sharpie, 1 pair of large vinyl gloves, and 1 icepack.

Our goal was to collect samples from wolves throughout the extent of their range in Minnesota; however, the vast majority of samples were collected by USDA-WS with an expected bias toward depredating wolves. Opportunistic sampling (e.g., vehicle kills) was encouraged to help increase sample size and provide a better distribution in more remote areas within wolf range.

Blood was collected from the jugular vein whenever possible (cephalic vein or saphenous vein are also options). For euthanized wolves, blood was collected from the site of a bullet wound, heart, or from the chest cavity as soon after death as possible. Blood was centrifuged and serum extracted. Whole blood samples were kept cool and sent to an entomologist at the University of Minnesota for tick-related disease research. Fecal samples were collected from the rectum and placed in a whirlpak bag. Heart and brain samples were also collected from euthanized wolves when possible. Once properly preserved, the serum and genetic samples collected during this study can be stored indefinitely.

Serums were screened for 8 diseases at the Veterinary Diagnostic Laboratory at the University of Minnesota (UMN-St. Paul) and the National Veterinary Services Laboratory (Ames, Iowa). The presence of CPV was confirmed using a hemagglutination inhibition (HI) test; titers ≥256 were considered positive. Exposure to canine adenovirus 1 (CAV 1) was confirmed using a serum neutralization test (SN); titers ≥8 were considered positive. Canine distemper virus (CDV) was also detected using a SN test; titers ≥25 were considered positive. A plaque reduction neutralization test (PRNT) was used to confirm exposure to eastern equine encephalitis (EEE) and West Nile Virus (WNV). Heartworm disease was detected by an antigen test. An immunofluorescence assay (IFA) was used for evidence of exposure to Lyme disease; titers ≥160 were considered positive. The MNDNR is collaborating with Dr. J. P. Dubey (USDA-Agriculture Research Service, Beltsville, Maryland) on a *Neospora* research project. Dr. Dubey used both a modified agglutination test (MAT) and a neospora agglutination test (NAT) on samples of serum, heart, brain, or feces to confirm neospora. A titer ≥25 on either the MAT or the NAT test was considered positive.

Morphological measurements of cleaned skulls from dead wolves followed procedures described in Nowak (1995). The 10 measurements were (1) greatest length of skull, (2) zygomatic width, (3) alveolar length from P1 to M2, (4) maximum width of rostrum across outer sides of P4, (5) palatal width between alveoli of P1, (6) width of frontal shield, (7) height from alveolus of M1 to most ventral point of orbit, (8) depth of jugal, (9) crown length of P4, and (10) greatest crown width of M2 (illustrations of the measurements and a more detailed explanation of statistical procedures is described in Nowak [1995]). For all wolves, including live captures, we recorded coat color, body weight, and measurements of ear length, shoulder height, body length, tail length, and foot length and width.

To assess mtDNA and microsatellites, genetic samples (e.g., ear punch, FTA® card, and muscle samples) were collected from each wolf. Muscle samples were preserved in both 95% ethanol and desiccant, and stored at room temperature. Genetic samples will be evaluated by the U. S. Fish and Wildlife Forensics Laboratory in Ashland, Wisconsin. Details of the statistical analyses used to identify or group individuals based on DNA or morphology will be outlined when results are available. Herein, we simply note that the focus will be on elucidating any spatial differences or patterns in molecular or morphological attributes, and on assessing whether any observed molecular patterns translate into meaningful morphological differences.

Anticipated project duration is 2 years. We hope to sample a minimum of 400 wolves over the 2-year study period, with samples distributed throughout wolf range. Detection of any disease will be assessed at an assumed prevalence level; >1% prevalence would be significant.

As a pilot study, the significance of any disease detection from this health survey would require more formal study to ascertain its significance relative to population demography.

RESULTS AND DISCUSSION

Samples from a total of 206 wolves (149 adults, 4 yearlings, 42 pups, and 11 of unknown age; 105 males, 100 females, and 1 unknown sex) were included in the first year of this study. These included wolves that were euthanized by USDS-WS (*n* = 103), live-caught research animals (*n* = 31), vehicle kills (*n* = 22), found dead (*n* = 45), and euthanized due to sickness (*n* = 5) (Figure 1). Genetic samples were obtained from all wolves; however, blood and fecal samples were not collected from wolves that had been dead for an extended period of time.

Serologic Disease Screening

Serological results indicated wolves were exposed to all 8 diseases included in our screening (Table 1). These tests only confirm past exposure, not current infection.

Our results indicated 75% of wolves have been exposed to CPV, which is similar to findings reported by Mech et al. (2008) for northeastern Minnesota. Canine parvovirus was first reported in 1967, but it wasn't until 1978 that a new variant of the virus was reportedly killing a high number of newborn wolf pups. It was theorized that this new variant of CPV was a mutation from feline parvovirus. This disease can infect most age classes of canids; however, mortality related to CPV in domestic canids has been primarily associated with younger animals (1–12 weeks of age). Little is known about the epidemiology of CPV in wild canid populations or its potential to negatively impact populations. Mech et al. (2008) reported that annual pup survival was reduced by 70% in northeastern Minnesota, and wolf population change was related to CPV antibody prevalence. These authors further speculated that CPV may reduce pup survival by 40–60% in the greater Minnesota population, and that this reduction limited rate of population increase to about 4% per year (compared with increases of 16–58% in other wolf populations). Canine parvovirus is transmitted through the fecal-oral route and causes diarrhea, fever, and dehydration. The disease can be fatal to wolves and is suspected of causing declines or attenuation of wolf populations in Wisconsin (Wydeven et al. 1995) and on Isle Royale, Michigan (Peterson et al. 1998).

Prevalence of CAV1 in wolves in our study was slightly less than the 96% reported in Yellowstone's adult wolf population (Almberg et al. 2009). Canine adenovirus 1 causes hepatitis, a disease of the liver and other body organs. The virus is found worldwide and is spread by body fluids including nasal discharge and urine. Canids of any age are susceptible to the disease. The incubation period is from 6 to 9 days, and signs include fever, loss of appetite, congested mucous membranes, and pain in the region of the liver. Reported mortality in dogs (*Canis familiaris*) is about 10%, and about 25% of the survivors develop a temporary corneal opacity (hepatitis blue eye). Chronic infection may occur, leading to cirrhosis of the liver. It remains unclear how endemic CAV 1 infection might impact wolf populations.

Wolves in Minnesota showed similar exposure to CDV as Spanish wolves (18.7%, Sobrino et al. 2007). Canine distemper virus is a *Morbillovirus* that infects a broad class of canids. Animals acquire CDV through inhalation or ingestion of airborne particles (Murray et al. 1999), and clinical signs include pneumonia, encephalitis, and death. Since CDV occurs in several carnivore taxas, there is concern about horizontal transmission among species. Outbreaks of CDV in 1999, 2002 and 2005 in free-ranging wolves within Yellowstone National Park were correlated with high pup mortality rates (Almberg et al. 2009). The CDV appears to be capable of causing dramatic population declines over a short time- frame.

Eastern equine encephalitis is a member of the genus *Alphavirus* in the family *Togaviridae*, which has been a source of epizootics in both domestic and wild animals since the 19th century. Outbreaks are typically concentrated around swampy areas and have been found primarily in the southeastern U. S., but also in Michigan and Wisconsin. Transmission by mosquitoes is thought to be the primary source of exposure; however, direct contact with contaminated blood, feces, vomitus, semen, or assassin bugs also can be a source of infection. Clinical signs vary depending on the species. Little is known about EEE infection in wolves; however, the disease has been documented in dogs (Farrar et al. 2005). Clinical signs in dogs were described as including pyrexia, depression, nystagmus, and lateral recumbency. Farrar et al. (2005) concluded that primarily young dogs are the most susceptible to EEE. This disease had not been known to occur in Minnesota prior to the MNDNR's moose health assessment project initiated in 2007, which discovered 6% of moose (*Alces alces*) in northeastern Minnesota have serological evidence of exposure to EEE (Butler et al. 2010). Our findings suggest northeastern wolves are also exposed to EEE, yet it is unclear what effect, if any, this may have on wolf survival.

West Nile virus is an avian virus that can cause fatal disease in some species of mammals, reptiles and birds. West Nile virus is an arbovirus in the *Flavivirus* genus of the family Flaviviridae. Until 1999, WNV was confined to the eastern hemisphere; however, it has since spread to North America and is now considered established in the U. S. and Canada. West Nile Virus is primarily transmitted by mosquitoes; 59 species are confirmed carriers in North America alone. A recent study of Minnesota's northeastern moose population found nearly 35% serologic prevalence (Butler et al. 2010), and their range overlaps with wolf range. While it remains unclear what effect WNV has on the nearly 14% of wolves that we documented were exposed to the disease, neurological signs have been reported from rare clinical cases in dogs and wolves. For example, a case of WNV was reported in a captive 4-month-old Arctic wolf pup (*C. lupus arctos*, Lanthier et al. 2004) and in a 3-month old wolf pup (Lichtensteiger et al. 2003). Both reportedly exhibited vomiting, anorexia and ataxia prior to death, which occurred 24–48 hours after the onset of neurological signs.

Results from nearly 10% of wolves in our study indicated exposure to heartworm, which has been previously documented in Minnesota wolves by Mech and Fritts (1987). Mosquitoes are the major vector of dog heartworm, *Dirofilaris immitis*. Once the worms end up in a canine, they will mature and grow on the right side of the animal's heart and pulmonary arteries. Initial symptoms include detectable heart murmurs and pulse deficits. As the problem progresses, the animal's heart may become enlarged and if the infection becomes severe (up to 200 worms have been found in some animals), blood flow will be blocked. Heart failure may result from a major infection. Heartworm has not been reported in Canada or Alaska, as the mosquitoes that carry it prefer warmer climates.

Our findings indicated a significantly higher prevalence of Lyme disease than 2.5%, which was previously reported in wolves in Minnesota and Wisconsin (Thieking et al. 1992). Lyme disease is caused by the bacterium *Borrelia burgdorferi*, and can affect dogs, horses and humans. The disease was first discovered in New England in 1975, and has since been reported in at least 43 states and eastern Canada. Infection typically results from bites from infected *Ixodes scapularis* ticks (deer ticks). White-tailed deer (*Odocoileus virginianus*) are the major hosts for the mature ticks, whereas small rodents are the hosts for the immature ticks. These hosts can become infected with *B. burgdorferi*, but never show symptoms of the disease. Wolves in Minnesota and Wisconsin have been found to be infected with the disease, but clinical Lyme disease has not yet been found in wild wolves. A wolf was experimentally infected with *B. burgdrferi* and showed some symptoms of the disease (lymphadenopathy), which suggests that wolves may be susceptible to it (Thieking et al. 1992).

Neospora caninum is a protozoal parasite, which is best known for causing abortion in cattle and neurological disease in dogs. Wild herbivores and canids also are thought to act as intermediate and definitive hosts, respectively (Gondim 2006, Dubey et al. 2009). While clinical disease due to infection is best described in domestic animals, reports of ill-effects due to *Neospora* infection in wildlife do exist. Gondim et al. (2004) reported that *N. caninum* antibody seroprevalence was detected in 39% of free-ranging gray wolves , 11% of coyotes (*Canis latrans*), 26% of white-tailed deer, and 13% of moose. These data are consistent with a sylvatic transmission cycle of *N. caninum* between cervids and canids. The authors speculated that hunting by humans favors the transmission of *N. caninum* from deer to canids, because deer carcasses are usually eviscerated in the field. Infection of canids, in turn, increases the risk of transmitting the parasite to domestic livestock.

Fecal Parasitology

A total of 62 fecal samples were examined by floatation for any evidence of ova or protozoal infection. Nine of the samples had hookworm ova, 2 had trematode ova, 13 had sarcocysts, and 2 were positive for both sarcocyts and hookworm. While this provides an idea of the types of parasites present in the wolf population, it does not provide an indication of parasite load or infection rate, as fecal-shedding does not correlate with severity of infection and shedding is often cyclical (Gondim 2006).

Wolves are susceptible to a variety of internal and external parasites. These include at least 24 species of nematodes (roundworms), 21 species of cestodes (tapeworms), 9 species of trematodes (flukes), heartworms, and 3 species of acanthocephalia (spiny-headed worms).

General Tick-borne Illness Screening

A total of 38 blood samples were submitted to the Department of Entomology (UMN), where we are collaborating with Dr. Ulrike Munderloh, to determine if wolves are infected with tick-borne illnesses. Whole blood samples were screened with a variety of polymerase chain reaction (PCR) techniques, which determine disease infection, not just disease exposure (which is detected through serology). Preliminary results indicate that 7.9% of the wolves were infected with Anaplasmosis, 39.5% were positive for prioplasma primers, and 5.3% were infected with Lyme disease. Further analysis is pending.

Morphology and Genetic Analysis

Although 206 skulls have been collected for taxonomic evaluation, presently, only about 15% have been cleaned. We have initiated a collaboration for preparing and curating skulls with Dr. Sharon Jansa at the Bell Museum of Natural History (UMN). As collection skulls are prepared for storage, measurements will be made as in Nowak (1995). Each skull will be permanently cataloged in the mammal collection at the Bell Museum.

Genetic samples are being stored until the National Wildlife Forensics Laboratory can conduct analyses, as in Fain et al. (2010). We intend to submit these samples for analyses in July 2011. New information has been presented in vonHoldt et al. (2011), which indicates wolves in Minnesota are predominantly gray wolves with admixture from coyotes that dates between 600–900 years ago. However, different sources have presented competing information about the genetic identity of wolves in Minnesota; consequently, additional analyses may be required to enhance our understanding of their genetic makeup. . Further, analysis of how skull morphology correlates to genetic identification may also contribute to our understanding of the taxonomic relationships of wolves in the region.

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LITERATURE CITED

Almberg, E. S., L. D. Mech, D. W. Smith, J. W. Sheldon, R. L. Crabtree. 2009. A serological survey of infectious disease in Yellowstone National Park's canid community. PLoS ONE 4:e7042.doi:10.1371/journal.pone.0007042.

Butler, E., M. Carstensen, E. Hildebrand, J. Giudice, R. Wright, and M. Schrage. 2010. Preliminary results from the 2007–2009 moose herd health assessment project. Pages 150-163 *in* J. S. Lawrence, R. O. Kimmel, and M. S. Lenarz, eds. Summaries of wildlife research findings, 2009, Minnesota Department of Natural Resources, Wildlife Populations and Research Unit, St. Paul.

Dubey, J. P., M. C. Jenkins, O. C. Kwok, R. L. Zink, M. L. Michalski, V. Ulrich, J. Gill, M. Carstensen, and P. Thulliez. 2009. Seroprevalence of neospora caninum and toxoplasma gondii antibodies in white-tailed deer (*Odocoileus virginianus*) from Iowa and Minnesota using four serologic tests. Veterinary Parasitology 161:330–334.

Fain, S. R., Straughan, R. J., and Taylor, B. F. 2010. Genetic outcomes of eastern timber wolf recovery in the western Great Lakes States. Conservation Genetics 11:1747–1765.

Farrar, M. D., D. L. Miller, C. A. Baldwin, S. L. Stiver, and C. L. Hall. 2005. Eastern equine encephalitis in dogs. Journal of Veterinary Diagnostics 17:614-617.

Gondim, L. F. P. 2006. *Neospora caninum* in wildlife. Trends in Parasitology 22:247–252.

Gondim, L. F. P., M. M. McAllister, N. E. Mateus-Pinilla, W. C. Pitt, L. D. Mech, and M. E. Nelson. 2004. Transmission of *neospora caninum* between wild and domestic animals. Journal of Parasitology 90:1361–1365.

Koblmuller, S., M. Nord, R. L. Wayne, and J. A. Leonard. 2009. Origin and status of the Great Lakes wolf. Molecular Ecology 18:2313–2326.

Lanthier, I., M. Hebert, D. Tremblay, J. Harel, A. D. Dallaire, and C. Girard. 2004. Natural West Nile virus infection in a captive juvenile arctic wolf (*Canis lupes*). Journal of Veterinary Diagnostics 16:326–329.

Leonard, J. A., and R. K. Wayne. 2008. Native Great Lakes wolves were not restored. Biological Letters 4:95–98. doi:10.1098/RSBI.2007.0354.

Lichtensteiger, C. A., K. Heinz-Taheny, T. S. Osborne, R. J. Novak, B. A. Lewis, and M. L. Firth. 2003. West Nile Virus encephalitis and myocarditis in a wolf and dog. Emerging Infectious Diseases 9:1303–1306.

Mech, L. D. 2008. Crying wolf: concluding that wolves were not restored. Biological Letters 5:65–66.

Mech, L.D. 2010. What is the taxonomic identity of Minnesota wolves? Canadian Journal Zoology 88:129–138.

Mech, L. D., and S. Fritts. 1987. Parvovirus and heartworm found in Minnesota wolves. *Endangered Species Technical Bulletin* 12:5–6.

Mech, L. D, S. M. Goyal, W. J. Paul, and W. E. Newton. 2008. Demographic effects of canine parvovirus on a free-ranging wolf population. Journal of Wildlife Diseases 44:824–836.

Murray, D. L., C. A. Kapke, J. F. Evermann, and T. K. Fuller. 1999. Infectious disease and the conservation of free-ranging large carnivores. Animal Conservation 2:241–54.

Nowak, R. M. 1995. Another look at wolf taxonomy. Pages 375-397 *in* L. N. Carbyn, S. H. Fritts, and D. R. Seip, eds. Ecology and conservation of wolves in a changing world. Proceedings of the Second North American symposium on wolves. University of Alberta, Edmonton, Canada.

Nowak, R. M. 2009. Taxonomy, morphology, and genetics of wolves in the Great Lakes Region. Pages 233-250 *in* A. P. Wydeven, T. R. Van Deelen, and E. J. Heske, eds. Recovery of gray wolves in the Great Lakes Region of the United States: a endangered species success story. Springer, New York.

Peterson, R. O., N. J. Thomas, J. M Thurber, J. A. Vucetich, and T. A. Waite. 1998. Population limitation and the wolves of Isle Royale. Journal of Mammalogy 79:828–841.

Schwartz, M. K., and J. A. Vucetich. 2009. Distinctness of the Great Lakes wolf. Molecular Ecology 18:2307–2309.

Sobrino, R., M. C. Arnal, D. F. Luco, and C. Gortazar. 2007. Prevalence of antibodies against canine distemper virus and canine parvovirus among foxes and wolves in Spain. Veterinary Microbiology 126:251–256.

Theiking, A., S. M. Goyal, R. F. Bey, K. I. Loken, L. D. Mech, R. P.Thiel, and T. P. O'Conner. 1992. Seroprevalence of Lyme disease in gray wolves from Minnesota and Wisconsin. Journal of Wildlife Diseases 28:177–182.

vonHoldt, B. M, J. P. Pollinger, D. A. Earl, J. C. Knowles, A. R. Boyko, H. Parker, E. Geffen, M. Pilot, W. Jedrzejewski, B. Jedrzejewski, V. Sidorovich, C. Greco, E. Randi, M. Musiani, R. Kayes, C. D. Bustamante, E. A. Ostrander, J. Novembre, and R. K. Wayne. 2011. A genome-wide perspective on the evolutionary history of enigmatic wolf-like canids. Genome Research, doi: 10.1101/gr.116301.110.

Wheeldon, T., and B. N. White. 2009. Genetic analysis of historic western Great Lakes region wolf samples reveals early *Canis lupus/lycaon* hybridization. Biological Letters 5: 101–104.

Wydeven, A. P., R. N. Schultz, and R. P. Thiel. 1995. Gray wolf monitoring in Wisconsin 1979–1991. Pages 147-156 *in* L. D. Carbyn, S. H. Fritts, and D. R. Seip, eds. Ecology and behavior of wolves in a changing world. Proceedings of the Second North American Symposium on Wolves. University of Alberta, Edmonton, Alberta, Canada.

Table 1. Serological results for disease screening of wolves sampled in Minnesota, January 2010–February 2011.

Disease	*n*	No. positives	Apparent prevalence (%)
Canine parvovirus	69	52	75.4
Canine adenovirus	71	54	76.1
Canine distemper virus	73	12	16.4
Eastern equine encephalitis	72	2	2.8
West Nile virus	73	10	13.7
Heartworm disease	73	7	9.6
Lyme disease	73	69	94.5
Neospora	73	61	83.6

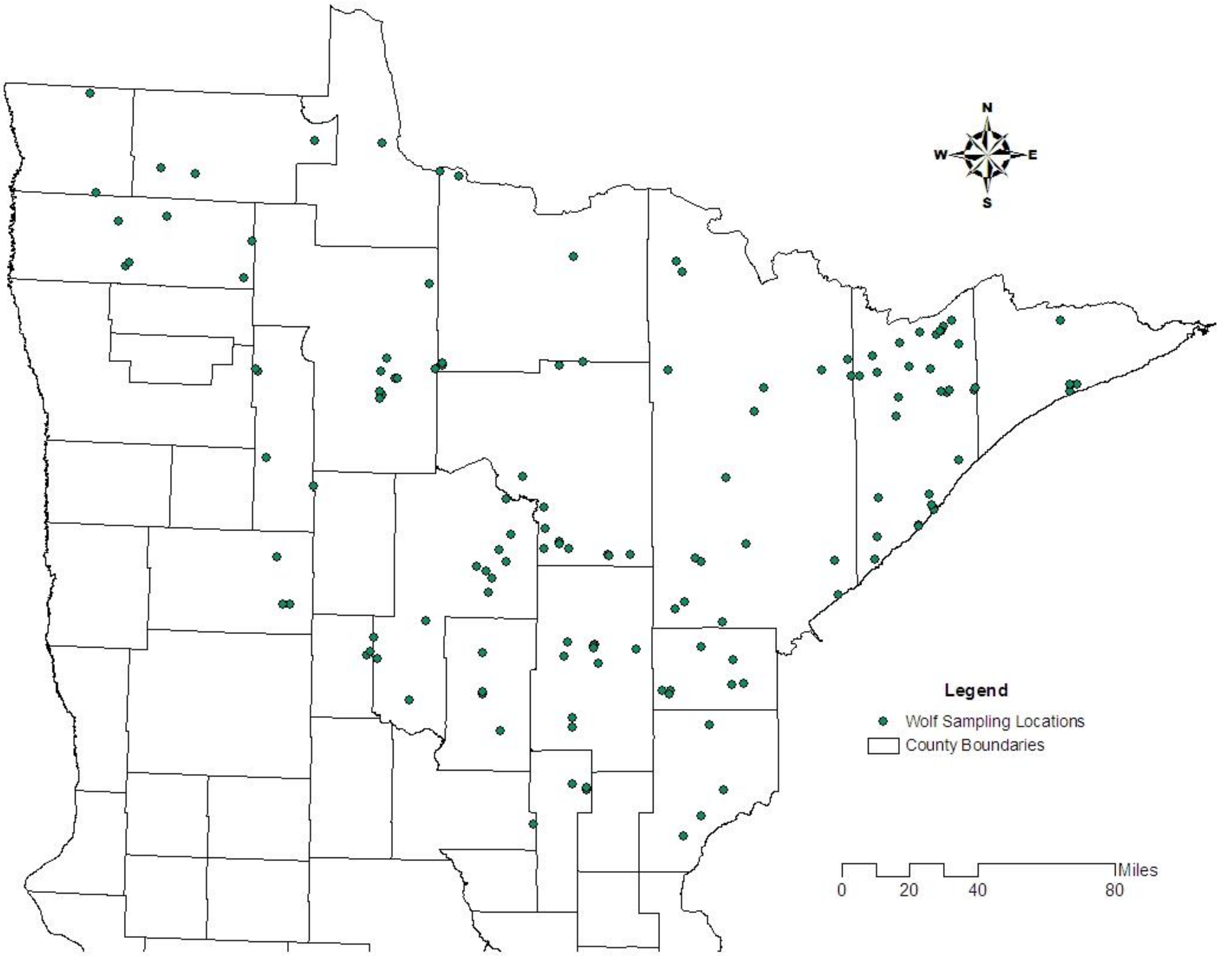

Figure 1. Sampling distribution of wolves (*n* = 206) during the first year of study of diseases and genetics of Minnesota's wolf population, 2010.

INTENSIFIED CHRONIC WASTING DISEASE SURVEILLANCE IN MINNESOTA'S SOUTHEASTERN WILD DEER HERD

Michelle Carstensen[1], Louis Cornicelli, David Pauly, Erik Hildebrand, and Erika Butler

SUMMARY OF FINDINGS

In November 2010, the Minnesota Department of Natural Resources (MNDNR) sampled 564 hunter-harvested white-tailed deer (*Odocoileus virginianus*) for chronic wasting disease (CWD) in southeastern Minnesota. This surveillance effort focused on a 32.2-km (20-mi) radius around a CWD-positive captive elk facility near Pine Island, discovered in 2009. One deer tested positive for CWD (0.2% apparent prevalence), marking the first detection of the disease in Minnesota's wild deer population. In response to this disease detection, MNDNR conducted a fixed-wing aerial deer survey in a 16.0-km (10-mi) radius of the index case in late January 2011 and estimated 6,200 deer (7.3 deer/km^2, 19 deer/mi^2). A supplemental surveillance effort was conducted in February–March 2011; 752 adult deer samples were collected and all tested CWD-negative. To prevent further disease spread, MNDNR banned recreational feeding of deer in a 4-county area in southeastern Minnesota. MNDNR will continue to conduct CWD surveillance of hunter-harvested deer in fall 2011.

INTRODUCTION

To date, CWD has been diagnosed in 3 captive elk (*Cervus elaphus*) herds and 1 captive white-tailed deer herd within the state of Minnesota. Two of the elk herds (Stearns and Aitkin counties) were discovered in 2002 and depopulated; no additional CWD-positive animals were found. In spring 2006, a captive white-tailed deer from a mixed deer/elk herd in Lac Qui Parle County was discovered to be infected with CWD. That herd was also depopulated without additional infection being detected. In early 2009, a third captive elk herd (Olmsted County) was found infected with CWD and, following depopulation of >600 animals, a total of 4 elk were confirmed with the disease. The United States Department of Agriculture's (USDA) indemnification document noted there was an apparent longstanding infection within this captive elk facility.

Overall, Minnesota has approximately 580 domestic cervid facilities with approximately 15,100 deer, elk, and other cervidae behind fences. As the current statewide population estimate of wild deer approaches one million, there is an element of inherent risk associated with disease transmission between domestic and wild cervids. Overall, risk is difficult to quantify as deer populations are unevenly distributed over the landscape and range in densities from (< 1-15 deer/km^2, 1–40 deer/mi^2. In addition, domestic facilities are sporadically distributed on the landscape and are mutually exclusive of deer densities.

In response to the discoveries of the first Minnesota CWD-positive captive elk herd in 2002 and CWD in wild Wisconsin white-tailed deer, the MNDNR developed a comprehensive wild deer CWD monitoring program. This included surveillance of targeted animals (e.g., suspect or potentially sick deer exhibiting clinical signs or symptoms consistent with CWD), opportunistic surveillance (e.g., vehicle-killed deer), and hunter-killed deer surveillance. During 2002–2004, nearly 28,000 deer were tested for CWD statewide with no positive results. Following completion of the statewide surveillance, the MNDNR scaled back surveillance efforts and sampled animals in response to elevated risk factors (e.g., detection of CWD-positive animals in captive cervid farms in Minnesota, or proximity of positive CWD cases in wild deer in neighboring states). From 2004 to 2009, an additional 5,200 hunter-harvested deer and over 500 targeted or opportunistic deer were tested for CWD, with no positives detected.

Chronic wasting disease belongs to a family of infectious diseases, called transmissible spongiform encephalopathies (TSEs), which alter the morphology of the central nervous system, resulting in a "sponge-like" appearance of this tissue. Chronic wasting disease only affects elk, mule deer (*O. hemionus*), white-tailed deer, and moose (*Alces alces*). The

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[1] Corresponding author e-mail: michelle.carstensen @.state.mn.us

etiological agent of CWD is an infectious protein, called a prion. Incubation time of the disease can range from 1.5 to nearly 3 years, although infected animals have been shown to shed prions in their feces up to a year before showing signs of illness (Tamguney et al. 2009). Clinical signs are non-specific and may include a loss of body condition and weight, excessive salivation, ataxia, and behavioral changes. There is no known treatment or vaccine for the disease and it is always fatal. Experimental and circumstantial evidence suggest that transmission of the disease is primarily through direct contact with infected animals or their infective saliva or excrement (Mathiason et al. 2006, Safar et al. 2008). However, persistence of prions in the environment and resulting indirect transmission has been shown to occur (Miller et al. 2004, Johnson et al. 2007, and Maluquer de Motes et al. 2008).

The Center for Disease Control (CDC) and other public health agencies have concluded there is no known link between CWD and any neurological disease in humans (MaWhinney et al. 2006). However, both the CDC and the World Health Organization (WHO) recommend that no part of a known positive animal should be consumed by humans. Additionally, there is no evidence that CWD can be naturally transmitted to species other than deer, elk, or moose.

METHODS

Hunter-harvested surveillance was conducted at deer registration stations during the regular firearm hunting season and first weekend of the muzzleloader season. Stations were staffed with MNDNR personnel and students (veterinary medicine and natural resources) trained in lymph node collection. Hunters were asked to voluntarily submit medial retropharyngeal lymph node samples for CWD-testing. All samples were inventoried, entered into a database, and sent to the University of Minnesota's Veterinary Diagnostic Laboratory (St. Paul, MN) for enzyme-linked immunosorbent assay (ELISA) testing. Positive samples from ELISA testing would be confirmed using immunohistochemistry (IHC) testing at the National Veterinary Services Laboratory in Ames, Iowa.

During fall 2010, registration stations were selected based on deer volume and distribution throughout the surveillance zone to meet a sampling goal of 500 deer within a 20-mile radius of the former CWD-positive elk farm near Pine Island. At the time of sample collection, hunter information was recorded, including the hunter's name, a telephone number, MNDNR number, and location of kill. Maps were provided to assist the hunters in identifying the location (Township, Range, and Section) of the kill. Cooperating hunters were given a cooperator's patch and entered into a raffle to win a firearm donated by the Minnesota Deer Hunter's Association.

MNDNR continued to sample deer exhibiting clinical symptoms consistent with CWD (targeted surveillance) statewide. Information has been disseminated to wildlife staff regarding what to look for regarding symptomatic deer. Staff were provided the necessary equipment and training for lymph node removal and data recording. The number of samples expected through targeted surveillance is estimated to be less than 100 animals annually, as few reports of sick deer are taken.

Additionally, MNDNR implemented efforts to obtain an additional 900 samples during winter 2011 in a 793-km^2 (306-mi^2) area surrounding a newly detected CWD-positive deer. Landowner shooting permits, agency-sponsored culling (conducted by USDA-Wildlife Services), and opportunistic sampling (e.g., vehicle-killed, sick or deer found dead) were used to collect samples from deer in this area. Landowner authorized by permit contacted trained MNDNR staff within 24 hours of harvesting deer; samples were collected in the field at private residences. All agency-harvested deer were transported intact to a central processing facility located within the winter CWD surveillance area. Sample collection and handling was similar to that described above. Carcasses were held in a refrigerated trailer at 33-35°F until test-negative results were reported (typically within 3 business days), then were salvaged for venison and made available to the public.

Prior to beginning the winter-sampling effort, MNDNR used a fixed-wing aircraft to conduct an aerial survey of the winter CWD surveillance area to assess deer numbers and distribution (Figure 1). A helicopter census of the CWD Core Area was conducted as well (Figure 2). This information was used to guide sharpshooting activities and estimate the percentage of deer removed from the area.

RESULTS AND DISCUSSION

During fall 2010, MNDNR sampled 438 hunter-harvested deer within 52 km^2 (20 mi^2) of the CWD-positive elk farm in Olmsted county, and an additional 86 deer in the periphery (Figure 3). In mid-January 2011, MNDNR was notified that an adult female harvested by a hunter on 28 November 2010, tested positive for CWD. This was the first case of CWD detected in a wild cervid in Minnesota. It was harvested approximately 4.8 km (3 mi) southwest of the former CWD-positive elk farm in Pine Island (Figure 4). Initial prevalence estimated the infection at < 0.2% of the local deer population. Further, over 3,200 deer were sampled in the southeast during falls 2009 and 2010 combined (Figure 4), which included about 400 deer within a 16-km (10-mi) radius of the index case.

From May 2010 to May 2011, MNDNR collected a total of 47 samples from targeted surveillance efforts. This included samples from 7 escaped captive cervids, 34 free-ranging sick deer, 2 free-ranging elk, 2 vehicle-killed deer, and 2 wild deer removed from within a captive cervid facility; all samples were negative for CWD.

Since discovery of our index case, the MNDNR has enacted its CWD Response Plan (http://files.dnr.state.mn.us/fish_wildlife/wildlife/disease/cwd/cwdresponseplan.pdf), which indentifies 4 primary goals for managing the disease:

1) determine and monitor the prevalence and geographic distribution of CWD in the infected area,
2) prevent or minimize further spread and new introductions of the disease,
3) support and conduct applied research on CWD and its epidemiology, and
4) provide accurate and current information about CWD to the public, constituent groups, and agency personnel.

As a first critical step in responding to CWD detection in the wild, the MNDNR conducted an aerial survey to gain an understanding of local deer abundance and distribution on the landscape. An aerial deer survey in late January-early February estimated 6,200 deer within the 793-km^2 (306-mi^2) winter surveillance area, equating to 7.3 deer/km^2 (19 deer/mi^2) density (Figures 1 and 2). Deer densities were highest within a 23-km^2 (9-mi^2) area surrounding the index case; 600 deer were counted with an estimate of >31 deer/km^2 (80 deer/mi^2) (Figure 2).

In order to gain further confidence in the apparent prevalence and geographic extent of the CWD infection in the local deer population, an additional 1,180 deer (752 adults, 428 fawns) were sampled within 16 km (10 miles) of the index case in winter 2011 (Figure 5); all deer were negative for the disease. Sampling included deer taken by landowner shooting permits (*n* = 491), agency-sponsored sharpshooting (*n* = 603), vehicle-kills (*n* = 59), and opportunistically (*n* = 27). Landowner shooting permits authorized landowners, or their designees, to take deer on their property. The permits had no bag limits and landowners were encouraged to take multiple deer. Ultimately, 323 landowner shooting permits were issued, and 47% of permit-holders harvested ≥1 deer. Overall, 57% of permitees took 1 or 2 deer and only 5% took >10 deer from their properties.

Another key step in preventing further spread of CWD was to ban the recreational feeding of deer. On February 14, MNDNR issued a special rule that made recreational deer feeding illegal in a 4-county area (Dodge, Goodhue, Olmsted and Wabesha), surrounding the location of the CWD-positive deer (Figure 6). The ban was aimed at reducing the potential for the disease spread by eliminating artificially-induced deer concentration sites. MNDNR Enforcement staff began educating and enforcing the new rule immediately and compliance was extremely high.

The estimated cost of the winter surveillance effort was $419,000. The majority ($229,000) resulted from the USDA sharpshooting contract, staff overtime ($82,000), and diagnostic testing ($30,000). The remaining expenditures were related to staff travel, building leases, and equipment leases or rentals.

Given the results of the CWD surveillance efforts of 2010 and winter 2011, evidence suggests that Minnesota is on the front end of a CWD outbreak in wild deer. The lack of detecting any additional infected deer in the immediate vicinity of the index case is encouraging.

It may be plausible that this disease is recent on the landscape and that few individuals have been exposed. Continued surveillance will be necessary to monitor this outbreak and determine what additional management actions may be needed to prevent CWD from becoming endemic in southeastern Minnesota.

ACKNOWLEDGEMENTS

We would especially like to recognize the tremendous amount of work and commitment by M. Dexter, J. Dippel, and H. Bloom for the massive surveillance effort this past winter. Equally important was excellent leadership provided by E. Boggess, M. DonCarlos, D. Simon, T. Bremicker, C. B. Bylander and G. Salo. We would like to thank all the MNDNR Area Wildlife and Enforcement staff from Regions 3 and 4, who volunteered to assist with this project in response to this disease outbreak. We also thank the students and faculty from the University of Minnesota, Colleges of Veterinary Medicine and Natural Resources, for assisting in our fall sampling efforts. Also, a special thanks to J. Adams and B. Wright for fulfilling our demanding GIS needs, J. Giudice, B. Haroldson, B. Moss, and T. Buker for conducting the aerial survey work, and P. Takash for his efforts in communications and internet-related work. We appreciate the support of the USDA-Wildlife Services program in their deer removal efforts (J. Hart and his team of wildlife professionals), as well as the Disease Program, who provided disease biologists (P. Wolf-MN, E. Blizzard-CO, J. Caudel-IN, and C. Hicks-OH). Their dedication and commitment to the successful completion of this project was tremendous.

LITERATURE CITED

Johnson, C. J., J. A. Pederson, R. J. Chappell, D. McKenzie, and J. M. Aiken. 2007. Oral transmissibility of prion disease is enhanced by binding to soil particles. PLoS Pathogens 3:e93.

Maluquer de Motes, C., M. J. Cano, M. Pumarola, and R. Girones. 2008. Detection and survival of prion agents in aquatic environments. Water Research 42:2465-2472.

Mathiason, C. K., J. G. Powers, S. J. Dahmes, D. A. Osborn, K. V. Miller, R. J. Warren, G. L. Mason, S. A. Hays, J. Hayes-Klug, D. M. Seelig, M. A. Wild, L. L. Wolfe, T. R. Spraker, M. W. Miller, C. J. Sigurdson, G. C. Telling, and E. A. Hoover. 2006. Infectious prion in the saliva and blood of deer with chronic wasting disease. Science 314:133-136.

MaWhinney, S., W. J. Pape, J. E. Forster, C. A. Anderson, P. Bosque, and M. W. Miller. 2006. Human prion disease and relative risk associated with chronic wasting disease. Emerging Infectious Diseases 12:1527-1535.

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 10:1003–1006.

Tamguney, G., M. W. Miller, L. L. Wolfe, T. M. Sirochmann, D. V. Glidden, C. Palmer, A. Lemus, S. J. DeArmond, and S. B. Prusiner. 2009. Asymptomatic deer excrete infectious prions in feces. Nature 461:529–532.

Safar, J. G., P. Lessard, G. Tamguney, and Y. Freyman. 2008. Transmission and detection of prions in feces. Journal of Infectious Diseases 198:81-89.

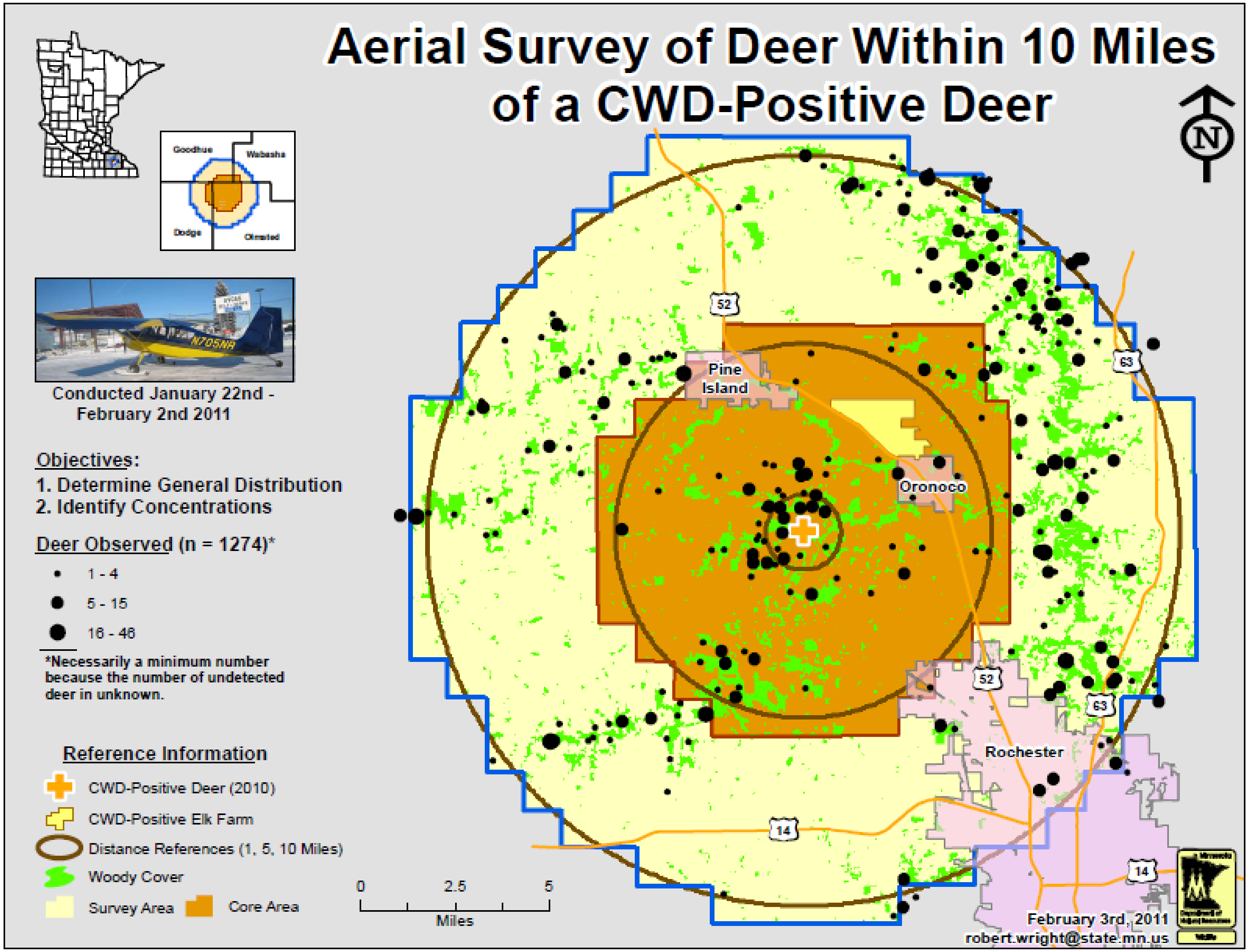


Figure 1. Fixed-wing, aerial survey results for 793-km^2 (306-mi^2) area surrounding the location of the white-tailed deer that tested positive for chronic wasting disease (CWD), southeastern Minnesota, January–February 2011.

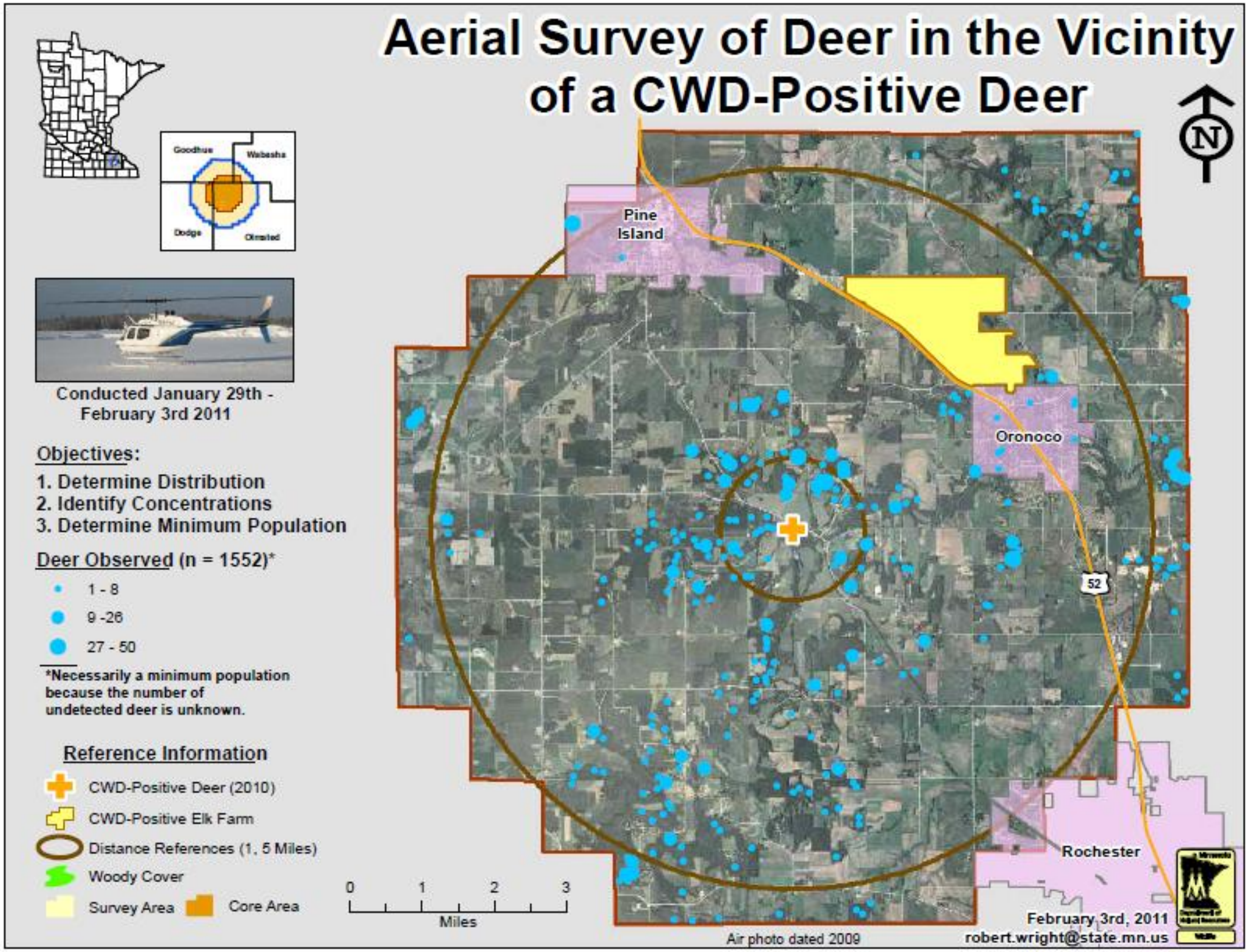


Figure 2. Helicopter white-tailed deer census for the 259-km^2 (100-mi^2) Core Area within the winter 2011 chronic wasting disease (CWD) surveillance area, southeastern Minnesota, January–February 2011.

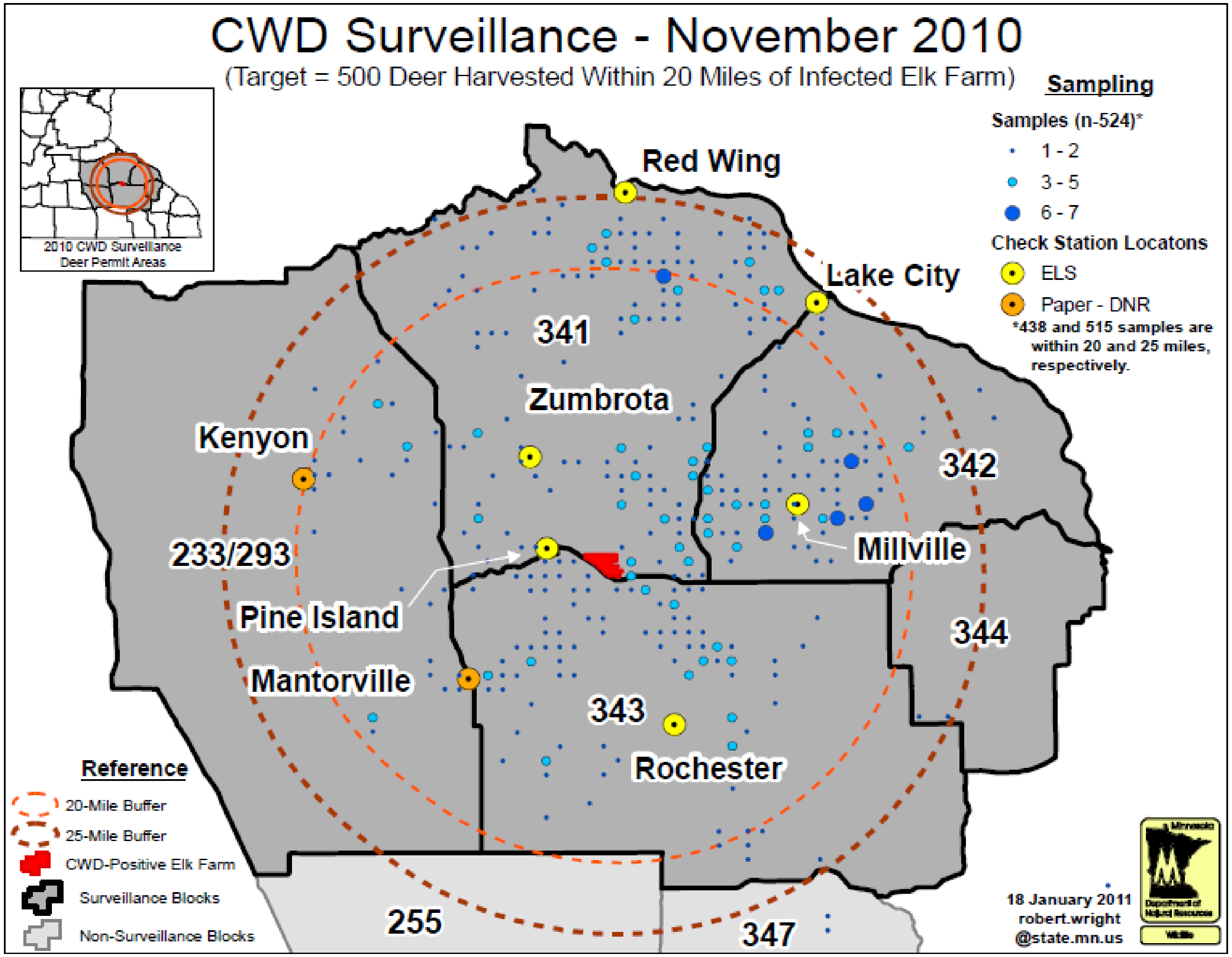


Figure 3. Sampling distribution for hunter-harvested white-tailed deer (*n* = 524) tested for chronic wasting disease (CWD) within 32 and 40 km (20 and 25 mi) of a formerly positive captive elk farm, southeastern Minnesota, fall 2010.

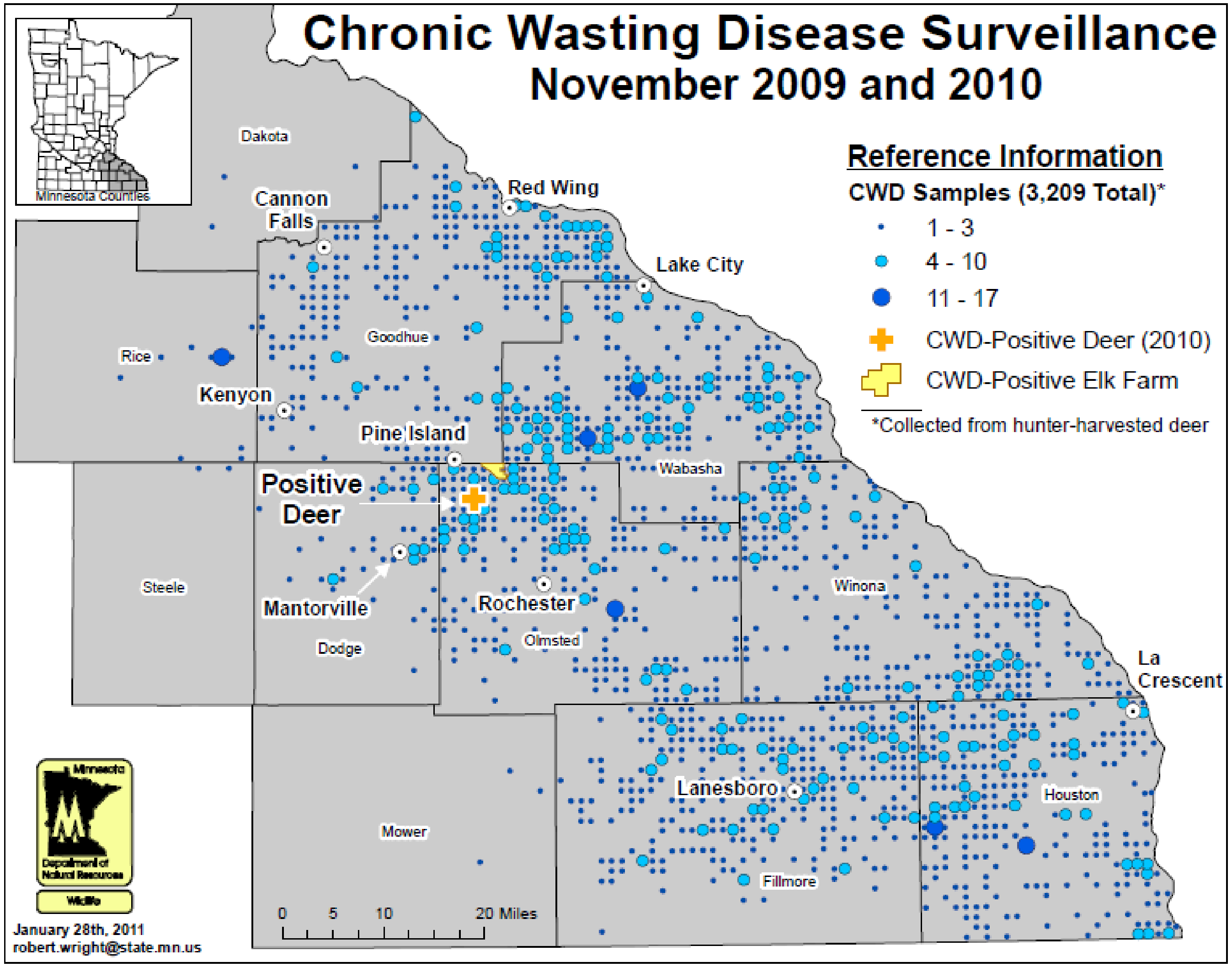
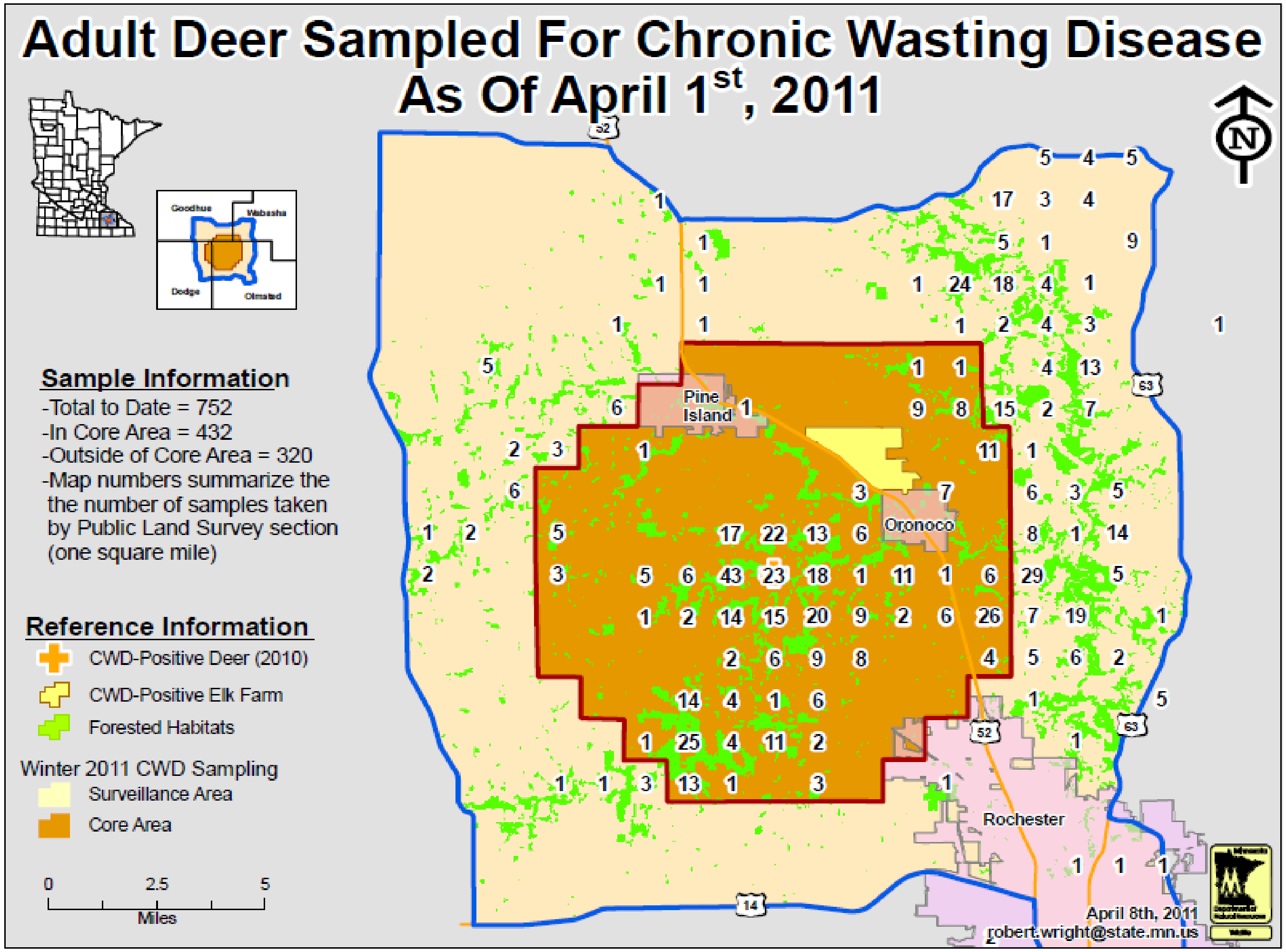


Figure 4. Sampling distribution for all hunter-harvested white-tailed deer (*n* = 3,209) tested for chronic wasting disease (CWD) in southeastern Minnesota, falls 2009 and 2010, in relation to the location of CWD-positive deer.



Adult Deer Sampled For Chronic Wasting Disease As Of April 1st, 2011

Sample Information
- Total to Date = 752
- In Core Area = 432
- Outside of Core Area = 320
- Map numbers summarize the the number of samples taken by Public Land Survey section (one square mile)

April 8th, 2011
robert.wright@state.mn.us

Figure 5. Section totals and distribution of white-tailed deer (*n* = 752) sampled for chronic wasting disease (CWD) during winter 2011, southeastern Minnesota.

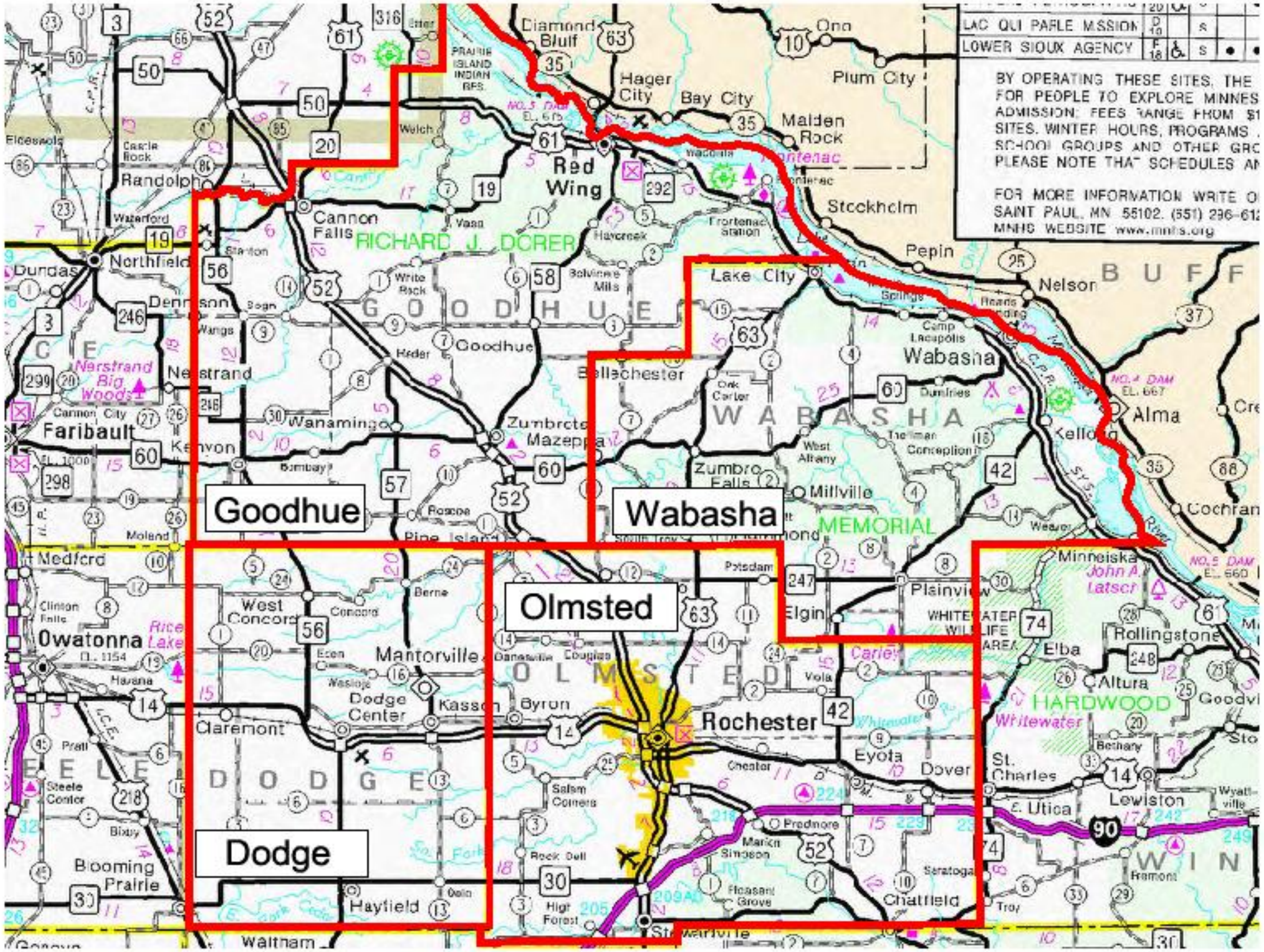


Figure 6. Four-county area in southeastern Minnesota where recreational feeding of wild white-tailed deer was banned in January 2011, following the discovery of chronic wasting disease in Olmsted County.

SPATIAL PATTERNS OF WHITE-TAILED DEER MOVEMENT RELATED TO BOVINE TUBERCULOSIS TRANSMISSION RISK IN NORTHWEST MINNESOTA

Michelle Carstensen[1], Robert Wright, Joao Ribeiro Lima[2], Louis Cornicelli, Eric Nelson, Scott Wells[2], and Marrett Grund

SUMMARY OF FINDINGS

The goal of this pilot research study is to provide a better understanding of white-tailed deer (*Odocoileus virginianus*) movements and habitat use in the transitional landscape of northwestern Minnesota, where a recent outbreak of bovine tuberculosis heightened awareness of disease transmission risks between deer and cattle. In January 2011, 16 deer were captured by helicopter net-gunning and fitted with satellite-linked global positioning system (GPS) collars. A second, ground-based capture effort in March added 5 deer to the study to compensate for a high winter mortality rate (47%), caused primarily by wolf predation. Preliminary findings for the first 5 months of this 15-month study indicated a mean winter home range size for deer (n = 19) from mid-January through mid-June of 19.9 km^2 (SE = 5.4) and a mean minimum cumulative distance traveled of 97 km (± 13). Serological screening of deer at capture for 9 common cattle diseases indicated exposure to bovine parainfluenza 3 virus (PI3, 24%), malignant catarrhal fever (MCF, 19%), and infectious bovine rhinotracheitis (9%). Fecal parasitology analyses indicated 13 (65%) of deer had evidence of liver fluke (*Fascioloides magna*) infection and strongyle-type ova was detected in 4 (20%) deer. Analysis of deer use of agricultural landscapes is pending.

INTRODUCTION

The Minnesota Department of Natural Resources (MNDNR) and the University of Minnesota (UMN) are collaborating on a 15-month pilot study to gain a better understanding of movements and habitat use by white-tailed deer in northwestern Minnesota. This is an area where continuous changes of forest into a more agricultural landscape and deer use of this "transitional" habitat are not partiularly well understood. The 2005 discovery of bovine tuberculosis (bTB) in wild deer in this area also increased concerns that a better understanding on how deer use such a diversified habitat is needed.

We are primarily interested in learning how deer use agricultural lands relative to state forest and wildlife management areas. In addition, we want to find out how farming practices, such as feed storage and animal husbandry, influence deer use of agricultural lands. This project intends to collect thousands of spatial locations of a small number of deer over 15 months. Utilizing this information to improve our understanding of how deer use farmed and pastured areas differently than natural habitats, we hope to gain greater insight into which practices may better minimize the risks of disease transmission between wild deer and cattle.

The UMN's Department of Veterinary Population Medicine previously developed a risk assessment process that was used by the Minnesota Board of Animal Health to evaluate the risk of deer and cattle interactions at farms within the bTB Management Zone (Knust et al. 2011). In this study, the UMN plans to quantify the microhabitat use of deer on farms, the potential for bTB transmission among cattle and deer, and to determine which herds are more likely to interact with deer as a consequence of the farm management practices. Further, we hope to leverage the results obtained in this study with another ongoing study evaluating cattle movements in northwestern Minnesota and possibly across the entire state. Combined, information generated from these studies should allow simulations of how bTB can spread across a network of farms where disease is introduced by infected cattle and spread by deer as a transmission vector. The research should also facilitate further understanding of steps that can be taken to mitigate these risks.

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[1] Corresponding author's email: michelle.carstensen@state.mn.us

[2] University of Minnesota, Department of Veterinary Population Medicine

Secondarily, the location data (“fixes”) stored on the radiocollars will allow the MNDNR to estimate home range sizes and dispersal rates, and describe migration patterns for the study animals. While we recognize that the results may not adequately represent the larger target deer population, they will provide wildlife managers and researchers with useful information and contribute to the design of a larger study in the future, should funding become available.

METHODS

The study area is approximately 360 km^2 and includes a mosaic of state forest and wildlife management lands, private recreational lands, and private farms (including row-crop agriculture, farmsteads, and stored forage). Within the area are >25 farms with a variety of livestock and agricultural uses (Figure 1). The study area lies just outside the southern boundary of the bTB Management Zone and contains 2 formerly bTB-infected cattle farms; however, the disease has not been detected in wild deer in this area. Deer density ranged from 15 to 20 deer/km^2. Major predators include gray wolves (*Canis lupus*), black bears (*Ursus americanus*), coyotes (*Canis latrans*), and bobcats (*Felis rufus*). Agricultural lands were surveyed to delineate and evaluate parameters that might attract deer to these areas (e.g., locations of stored forage, water sources, cattle pastures).

In winter 2011, deer were captured by helicopter net-gunning (Quicksilver Air, Inc., Fairbanks, Alaska) and Clover trap. We chemically immobilized (100 mg xylazine HCl, 400 mg ketamine HCl) captured deer and collected blood, urine, and fecal samples for health-screening. We also measured rump fat thickness by ultrasound and extracted a last lower incisor to determine exact age by counting cementum annuli (Mattson’s Laboratory, Milltown, Montana). We ear-tagged and fitted deer with a satellite-linked radiocollar (ARGOS, SirTrack, Hawkes Bay, New Zealand). Body temperature was monitored at 5-min intervals throughout the processing period. We administered a long-acting antibiotic (LA-200, oxytetracycline) intramuscularly (1 mL/10 kg body weight). Before release, we reversed anesthesia by intravenous injection of 15 mg/deer of yohimbine HCl. An observer monitored each deer’s recovery and recorded the time deer were up and moving away from the recovery area.

We programmed radiocollars to record locations every 90 minutes and transmit these “fixes” every 3 days through the ARGOS satellite system. Battery life of radiocollars is expected to be 15 months (to allow for 1 full year of seasonal movements). Collars were programmed to drop off in mid-April 2012. The research team will retrieve all collars and download the complete set of spatial data. In the interim, fixes are downloaded weekly and examined for temporal and spatial movement patterns to determine mortality, movements, and habitat use. For study animals that die during the study period, MNDNR wildlife staff investigate the cause of mortality, recover the collar, and collect medial retropharyngeal lymph node samples from the deer (when possible) for bTB testing.

Serums were tested for malignant catarrhal fever via peroxidase-linked assay (PLA); positive PLA tests werethen tested with a virus neutralization test (VN) at the National Veterinary Services Laboratory (Ames, Iowa). All other serology was conducted at the UMN’s Veterinary Diagnostic Laboratory (VDL) in St. Paul, Minnesota, which included screening for leptospirosis (6 serovars, microscopic agglutination test), anaplasmosis (card test), brucellosis (card test), and bovine parainfluenza 3 (hemagglutination inhibition test). Exposure to bluetongue virus and neosporosis were determined by enzyme-linked radioimmunoassay (ELISA). Exposure to bovine viral diarrhea (BVD, Types 1 and 2) and infectious bovine rhinotracheitis (IBR) were confirmed using serum neutralization tests (SN); titers ≥8 were considered positive. In addition, whole blood and serum were submitted to the UMN-College of Veterinary Medicine-Clinical Pathology Laboratory for a full large-animal serum chemistry profile and hematology; analyses of these results are pending.

We examined deer movements and made home range estimates using Home Range Tools (HRT) for ArcGIS® (Rodgers et al. 2007). Minimum convex polygons (MCPs) were constructing by connecting peripheral points containing 99% of available fixes (White and Garrott 1990, Rodgers et al. 2007).

For evaluation of deer use on the agricultural landscape, a descriptive analysis will be performed to evaluate patterns of deer visits to farms throughout the study period. This will include the number of visits to each farm by season and time of day, number of farms visited by each individual deer, differences in use of farm areas by age and sex of deer, and variations in home range characteristics of each deer during the study period. Also, a resource utilization model will be developed that compares characteristics of locations used by each deer to available locations that are not used; the intention is to identify higher risk areas for deer locations based on resource availability.

RESULTS AND DISCUSSION

Deer Capture and Handling

In January 2011, 16 deer (4 males, 12 females) were captured by helicopter net-gunning within (*n* = 11) and slightly northeast (*n* = 5) of the study area (Figure 2). Capture locations were driven by deer distribution at the time of capture and access to private land to process deer. Due to collar failure immediately following release, 1 deer (ID 519) was censored from the study and its fate remains unknown. As of June 2011, 7 of these remaining deer (47%) were killed by wolves (*n* = 6) or died from unknown causes (*n* = 1) (Table 1).

To compensate for the high winter mortality, the sample size was augmented with 5 deer (1 male, 4 females) captured using Clover-traps in March 2011 (Table 1). One of these deer (ID 577) was fitted with a test collar provided by SirTrack (Iridium satellite system prototype), and this collar failed to record or transmit locations immediately after the animal's release. Although this deer was censored from the study, it was killed by wolves in early April and the collar was recovered. A second deer (ID 447) from this group slipped its collar (likely caused by a premature triggering of the blow-off device) on 22 May 2011, and subsequently was censored from the study.

As of June 2011, 11 radiocollared deer remain in the study. The collars appear to be functioning well, as weekly satellite downloads of these animals obtained approximately one-third of recorded fixes (Table 2). This provided sufficient data to track major animal movements and monitor survival, yet preserves battery life by restricting the amount of time collars communicated with the satellite system. For collars that have been recovered, the success rate of obtaining fixes has been >95% (Table 3).

The number of mortalities we observed from February to April 2011, specifically due to wolf predation, was higher than expected. Winter conditions were moderately severe (Winter Severity Index [WSI] = 159, Red Lake Wildlife Management Area) in the study area, with prolonged snow cover of >36 cm from late-January through early April. In Minnesota's forest zone, DelGiudice et al. (2006) reported a 37% winter mortality rate for adult deer during the severe winter of 1995–1996 (WSI = 195), with wolves accounting for 63% of those deaths. During more moderately severe winters (WSI = 124 to 159) in north-central Minnesota, DelGiudice et al. (2006) reported winter mortality rates ranging from 7 to 19%, with wolf predation accounting for 50-80% of the deaths. In contrast, the winter mortality rate for adult female deer in Minnesota's farmland zone has been reported as only 5%; however, there is an absence of wolves and typically more mild winter conditions (Brinkman et al. 2004). Little information exists on winter mortality rates for deer in Minnesota's transition zone, and although the sample size was limited in this study, our preliminary findings suggest there might be factors in northwestern Minnesota imposing a unique influence on on deer population dynamics different than in the farmland and forest zones.

Disease Screening and Parasitology

Serological results indicated deer were exposed to bovine parainfluenza 3 virus (24%), MCF (19%), and infectious bovine rhinotracheitis (9%). There was no evidence of exposure to anaplasmosis, bluetongue virus, bovine viral diarrhea (Types 1 or 2), brucellosis, leptospirosis, or neosporosis. These tests only indicate deer have been exposed to these diseases, and thus, developed an immune response in which antibodies were detected through the various testing methods. We are not able to confirm current infection or illness from any of these diseases in these deer.

Exposure to PI3 in this study was not unexpected, as our prevalence was similar to the 20% reported by Ingebrigsten et al. (1986) for deer throughout Minnesota. Parainfluenza 3 virus is an RNA virus classified in the paramyxovirus family and is most commonly associated with cattle. Although PI3 is capable of causing disease, it is usually associated with mild to subclinical infections. The most important role of PI3 is to serve as an initiator that can lead to the development of secondary bacterial pneumonia. Little is known about PI3 infection in white-tailed deer. Thorsen et al. (1977) demonstrated PI3 was infective in both captive and free-ranging pronghorn (*Antilocapra americana*) in Alberta. In a serologic survey of wild cervids in national parks in the U. S., 58% of mule deer (*Odocoileus hemionus*) and 57% of elk (*Cervus elaphus*) were exposed to PI3 (Aguirre et al. 1995).

Our findings of 19% prevalence for MCF in deer is lower than what has been recently reported for wild elk in northwestern Minnesota (29%, Hildebrand et al. 2010) and northwestern moose (35%, Butler et al. 2010). Malignant catarrhal fever is caused by a Gammaherpes virus and affects many species in the family Artiodactyla (even-toed ungulates), including cattle, bison (*Bison bison*), deer, moose (*Alces alces*), exotic ruminants and pigs (*Sus scrofa domesticus*). At least 10 MCF viruses have been recognized worldwide, including 2 well-known viruses carried by sheep (*Ovis aries*) and wildebeest (*Connochaetes taurinus*); 5 MCF viruses have been linked to disease, while the others have been found, to date, only in asymptomatic carriers. The deer strain of MCF is typically carried asymptomatically, but it can cause disease in other susceptible species or in rare cases, in the reservoir host itself. In deer, MCF is usually acute and affected deer die within 1–2 days; however, more typically, MCF symptoms include corneal opacity, hemorrhagic diarrhea and bloody urine, shedding of the hoof in some animals, and death within 3 weeks of disease onset (Center for Food Security and Public Health 2008).

Infectious bovine rhinotracheitis is a highly contagious, infectious disease of cattle that is caused by bovine herpesvirus-1. Primarily a respiratory disease, IBR but can also cause conjunctivitis, abortions, encephalitis, and generalized system infections. Not much is known about IBR virus in deer. While we report a 9% prevalence, a higher prevalence (15%) was noted in a statewide serologic survey of Minnesota deer by Ingebrigsten et al. (1986). Infectious bovine rhinotracheitis exposure has also been reported in Minnesota's moose (Johnson et al. 1973). Sadi et al. (1991) reported a 57% prevalence of IBR in white-tailed deer on Anticosti Island (Quebec, Canada) and suggest it was the cause of an unusual mortality event among a 3–4 year-old cohort. While clinical signs associated with IBR in wild white-tailed deer are not known, researchers have demonstrated mild clinical signs in captive mule deer, including anorexia, depression, excessive salivation, increased respiratory rate, and occasional cough (Chow and Davis 1964).

Fecal samples from 20 deer were screened for evidence of parasites by fecal floatation. Thirteen (65%) of deer had evidence of liver fluke (*Fascioloides magna*) infection and strongyle-type ova were detected in 4 (20%) deer. Negative results do not necessarily mean the animal was parasite-free, only that it was not actively shedding at the time the feces were collected. Also, culture of fecal samples did not detect any evidence of Johne's disease (*Mycobacterium paratuberculosis*).

Home Range Size and Deer Movements

Mean home range size for all deer (n = 19) from mid-January through mid-June was 19.9 km^2 (SE = 5.4) and the mean cumulative distance traveled was 97 km (SE = 13). However, since deer were captured during mid-late winter, we are uncertain whether or not this represents solely winter ranges for these deer or also included all or portions of their spring-summer-fall ranges. Further, while deer that died (or slipped their collar) during the study had similar mean home range sizes to survivors (14 km^2 ± 6.7 and 24 km^2 ± 8.0, respectively;

Figures 3 and 4), the mean cumulative distance traveled by survivors was nearly twice as high as those that died (Tables 2 and 3), likely due primarily to being tracked over a longer time period. Six deer had home ranges >36 km^2,attributable to a few long-distance movements from one end of their range to the other (Figures 3 and 4). These movements began in late January for 3 deer, moving 11–21 km in a 2–3-day period. The other 3 deer moved 14–21 km in mid- to late March, again over a 2–3 day period. Of these 6 deer, 2 were killed, but the other 4 returned the same distance (in a 2–3 day period of travel) to the area in which they were originally captured in late March or early April. However, the majority of deer (63%) had home ranges ≤ 10 km^2.

Given the timing of deer capture (mid-January and early March), we assumed these animals were either on their winter range (if migratory) or were possibly year-round residents at the start of the study. Therefore, it is too early in the study for a thorough interpretation of the deer movement and home range data generated thus far. Brinkman et al. (2005) reported 78% of deer in Minnesota's farmland zone as migratory (43% obligate and 35% conditional migrators), with a mean migration distance of 10 km. Further, those authors determined mean winter and summer home ranges (95% MCPs) as 5.2 km^2 and 2.6 km^2, respectively. Conversely, forest zone deer in northeastern and north-central Minnesota were 89% and 68% migratory, respectively (Nelson 1995, Fieberg et al. 2008). Further, migration distances were most typically 10–14 km, but ranged from 2–135 km; onset of migrations varied annually, but ranged from early November to January (Fieberg et al. 2008). In both studies of forest zone deer, severe winters coincided with a higher number of conditional migrators making movements to a distinct winter range (Nelson 1995, Fieberg et al. 2008).

Deer Use of the Agricultural Landscape

No results have been generated yet, as only 5 months of spatial data are available. Analysis will occur at the completion of the 15-month project.

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LITERATURE CITED

Aguirre, A. A., D. E. Hansen, E. E. Starkey, and R. G. McLean. 1995. Serologic survey of wild cervids for potential disease agents in selected national parks in the United States. Preventive Veterinary Medicine 21:313–322.

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 32:1–7.

Brinkman, T. J., C. S. Deperno, J. A. Jenks, B. S. Haroldson, and R. G. Osborn. 2005. Movement of female white-tailed deer: effects of climate and intensive row-crop agriculture. Journal of Wildlife Management 69:1099–1111.

Butler, E., M. Carstensen, E. Hildebrand, J. Giudice, R. Wright, and M. Schrage. 2010.

Preliminary results from the 2007–2009 moose herd health assessment project. Pages 150–163 *in* J. S. Lawrence, R. O. Kimmel, and M. S. Lenarz, eds. Summaries of wildlife research findings, 2009. Minnesota Department of Natural Resources, Wildlife Populations and Research Unit, St. Paul.

Center for Food Security and Public Health. 2008. Malignant catarrhal fever. College of Veterinary Medicine, Iowa State University, Ames http://www.cfsph.iastate.edu/Factsheets/pdfs/malignant_catarrhal_fever.pdf

Chow, T. L., and R. W. Davis. 1964. The susceptibility of mule deer to infectious bovine rhinotracheitis. American Journal of Veterinary Research 25:518–519.

DelGiudice, G. D., J. Fieberg, M. R. Riggs, M. Carstensen Powell, and W. Pan. 2006. A long-term age-specific survival analysis of female white-tailed deer. Journal of Wildlife Management 70:1556–1568.

Fieberg, J., D. W. Kuehn, and G. D. DelGiudice. 2008. Understanding variation in autumn migration of northern white-tailed deer by long-term study. Journal of Mammalogy 89:1529–1539.

Hildebrand, E., M. Carstensen, E. Butler, and L. Cornicelli. 2010. Preliminary results of herd health assessment for northwestern free-ranging elk in Minnesota from 2004–2009. Pages 135–149 *in* J. S. Lawrence, R. O. Kimmel, and M. S. Lenarz, eds. Summaries of wildlife research findings, 2009. Minnesota Department of Natural Resources, Wildlife Populations and Research Unit, St. Paul.

Ingebrigsten, D. K., J. R. Ludwig, and A. W. McClurkin. 1986. Occurrence of antibodies to the etiologic agents of infectious bovine rhinotracheitis, parainfluenza 3, leptospirosis, and brucellosis in white-tailed deer in Minnesota. Journal of Wildlife Diseases 22: 83–86.

Johnson, D. W., M. Wheelock, B. Kolstad, and P. D. Karns. 1973. Serologic survey of Minnesota moose for infectious bovine rhinotracheitis (IBR), bovine viral diarrhea (BVD), parainfluenza 3 (PI3), and blue tongue (BT) antibodies. Proceedings of Annual Meeting, U.S. Animal Health Association 76:702–710.

Knust, B. M., P. C. Wolf, S. J. Wells. 2011. Characterization of the risk of deer-cattle interactions in Minnesota by use of an on-farm environmental assessment tool. American Journal of Veterinary Research 72:1–8.

Nelson, M. E. 1995. Winter range arrival and departure of white-tailed deer in northeastern Minnesota. Canadian Journal of Zoology 73:1069–1076.

Rodgers, A.R., A.P. Carr, H.L. Beyer, L. Smith, and J.G. Kie. 2007. HRT: Home Range Tools for ArcGIS. Version 1.1. Ontario Ministry of Natural Resources, Centre for Northern Forest Ecosystem Research, Thunder Bay, Canada.

Rodgers, A. R., A. P. Carr, H. L. Beyer, L. Smith, and J. G. Kie. 2007. HRT: Home Range Tools for ArcGIS. Version 1.1. Ontario Ministry of Natural Resources, Centre for Northern Forest Ecosystem Research, Thunder Bay, Canada.

Sadi, L., R. Joyal, M. St-Georges, and L. Lamontagne. 1991. Serologic survey of white-tailed deer on Anacosti Island, Quebec for bovine herpesvirus 1, bovine viral diarrhea, and parainfluenza 3. Journal of Wildlife Diseases 27: 569–577.

Thorsen, J., L. Karlstad, M. W. Barrett, and G. A. Chalmers. 1977. Viruses isolated from captive and free-ranging wild ruminants in Alberta. Journal of Wildlife Diseases 13:74–79.

White, G. C., and R. A Garrott. 1990. Analysis of wildlife radio-tracking data. Academic Press, Inc., San Diego, California. 383pp.

Table 1. Current status and fate of free-ranging white-tailed deer (*n* = 21) captured and radiocollared in January and March 2011, northwestern Minnesota.

Deer ID	**Capture Date**	**Method**	**Age Class**	**Age[1] (yr)**	**Sex[2]**	**Fate**	**Cause**	**Estimated Mortality Date**
469	1/15/11	Helicopter	Adult	4.5	F	Alive		
461	1/15/11	Helicopter	Yearling	1.5	F	Dead	wolf-kill	3/31/11
497	1/15/11	Helicopter	Yearling	1.5	F	Alive		
467	1/15/11	Helicopter	Yearling	1.5	M	Dead	wolf-kill	2/18/11
466	1/15/11	Helicopter	Adult	8.5	F	Alive		
496	1/15/11	Helicopter	Adult	2.5	F	Dead	unknown	2/23/11
472	1/15/11	Helicopter	Adult	5.5	F	Alive		
524	1/15/11	Helicopter	Adult	6.5	F	Dead	wolf-kill	3/10/11
473	1/15/11	Helicopter	Adult	4.5	M	Alive		
495	1/15/11	Helicopter	Adult	2.5	M	Alive		
471	1/15/11	Helicopter	Yearling	1.5	F	Dead	wolf-kill	4/5/11
491	1/16/11	Helicopter	Yearling	1.5	F	Alive		
348	1/16/11	Helicopter	Adult	9.5	F	Dead	wolf-kill	2/12/11
460	1/16/11	Helicopter	Adult	2.5	F	Dead	wolf-kill	2/10/11
519	1/16/11	Helicopter	Adult	3.5	M	Unknown	collar malfunction	
350	1/16/11	Helicopter	Adult	11.5	F	Alive		
336	3/7/11	Clover-trap	Yearling		M	Alive		
578	3/8/11	Clover-trap	Adult		F	Alive		
577[3]	3/8/11	Clover-trap	Adult		F	Dead	wolf-kill	4/10/11
579	3/8/11	Clover-trap	Adult		F	Alive		
447	3/10/11	Clover-trap	Adult		F	Unknown	slipped collar	

[1] Age (in years) was determined by cementum annuli. Analysis for deer captured in March is pending.

[2] F = female, M = male

[3] Deer 577 was fitted with a SirTrack test-collar (Iridium satellite system) and no movement data was recovered; mortality date are based on a public report of a severely injured deer and carcass remains.

Table 2. Fix success rates, home range size, and cumulative distance traveled by free-ranging deer (*n* = 11) remaining in the study, as of June 14, 2011, northwestern Minnesota.

Deer ID	Days on air	No. fixes[1]	Fix success rate[2] (%)	99% MCP[3] (km^2)	Cumulative distance traveled (km)
469	150	941	39.2	38.1	209.5
497	150	748	31.2	84.4	162.7
466	150	635	26.5	1.0	87.4
472	150	983	41.0	18.4	151.1
473	150	616	25.7	5.7	82.9
495	150	627	26.1	10.4	123.4
491	149	905	38.0	54.9	137.8
350	149	900	37.8	36.5	111.2
336	99	397	25.1	8.5	128.3
578	98	500	31.9	2.1	121.3
579	98	526	33.5	8.3	67.8
Mean	136	707	32.4	24.3	125.7
SE	7	60	1.8	8.0	12.1

[1] Total number of fixes included only data downloaded from the satellite system from deployment through June 14, 2011.

[2] Fix success rate was calculated by number of locations received through the satellite divided by the number of available locations, assuming collars recorded 16 locations/day.

[3] MCP = minimum convex polygon, contained 99% of all locations.

Table 3. Fix success rates, home range size, and distance traveled by free-ranging deer (*n* = 8) that had either died or slipped their collar during the study.

Deer ID	Days on air	No. successful fixes[1]	No. failed fixes	Success rate (%)	99% MCP[2] (km^2)	Cumulative distance traveled (km)
461	77	1325	8	99.4	40.1	111
467	43	774	7	99.1	1.0	18
496	43	773	17	97.8	0.5	14
524	61	1124	4	99.6	8.0	53
471	90	1693	82	95.2	10.4	128
348	28	517	13	97.5	47.9	29
460	43	763	24	96.8	0.3	9
447	89	1641	68	95.9	4.1	100
Mean	59	1076	28	97.7	14.0	58
SE	8	156	11	0.6	6.7	17.1

[1] Total number of successful fixes included all data from deployment until collar was recovered from the field, which extended beyond the estimated mortality dates.

[2] MCP = minimum convex polygon, contained 99% of all locations.

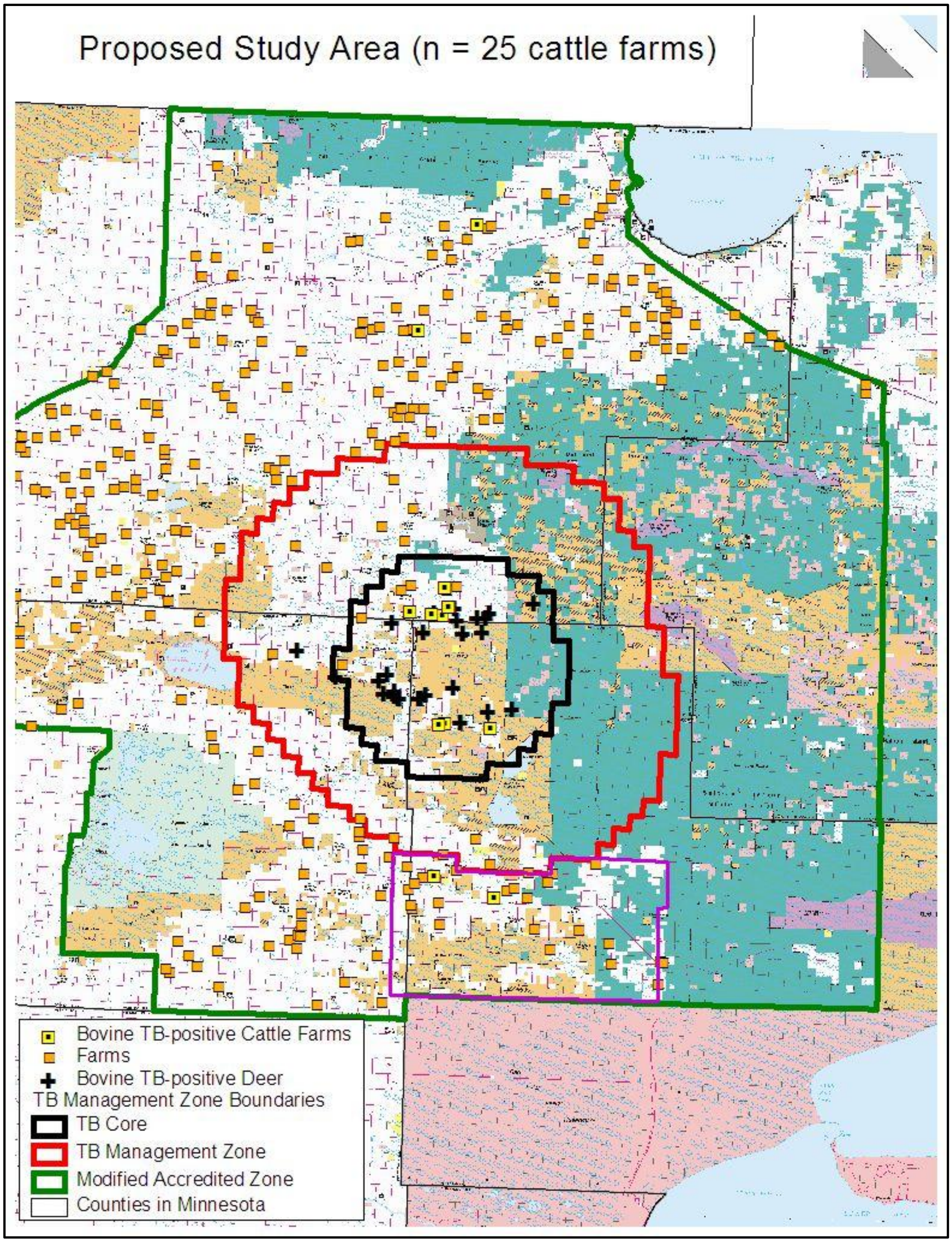

Figure 1. The 360-km^2-study area (outlined in purple) contains >25 cattle farms including 2 previously infected with bovine tuberculosis. The study area is immediately south of the Bovine Tuberculosis Management Zone, where 27 deer and 8 cattle farms tested positive for the disease.

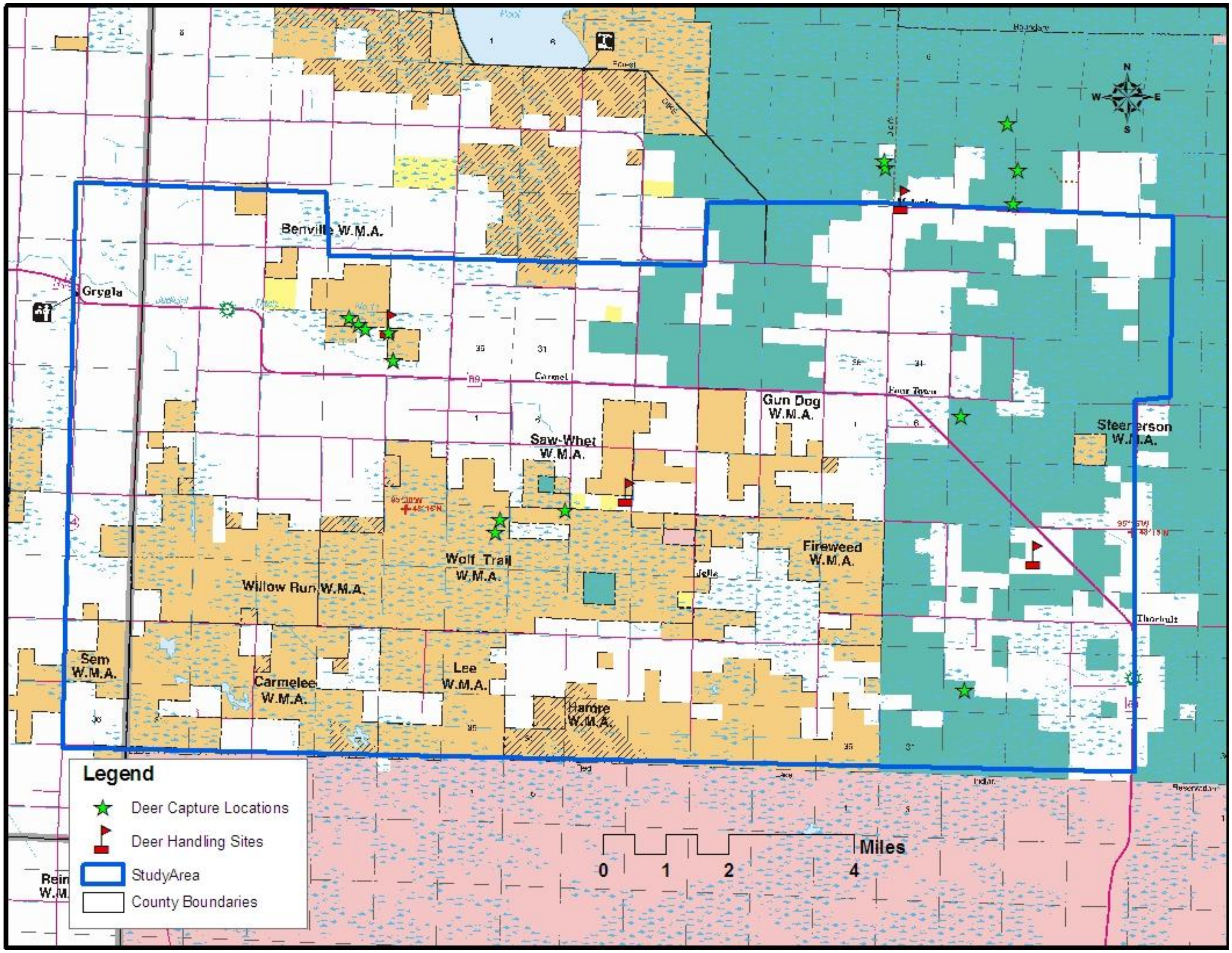

Figure 2. Capture locations and handling sites for free-ranging white-tailed deer (*n* = 16) captured by helicopter net-gunning in January 2011, northwestern Minnesota.

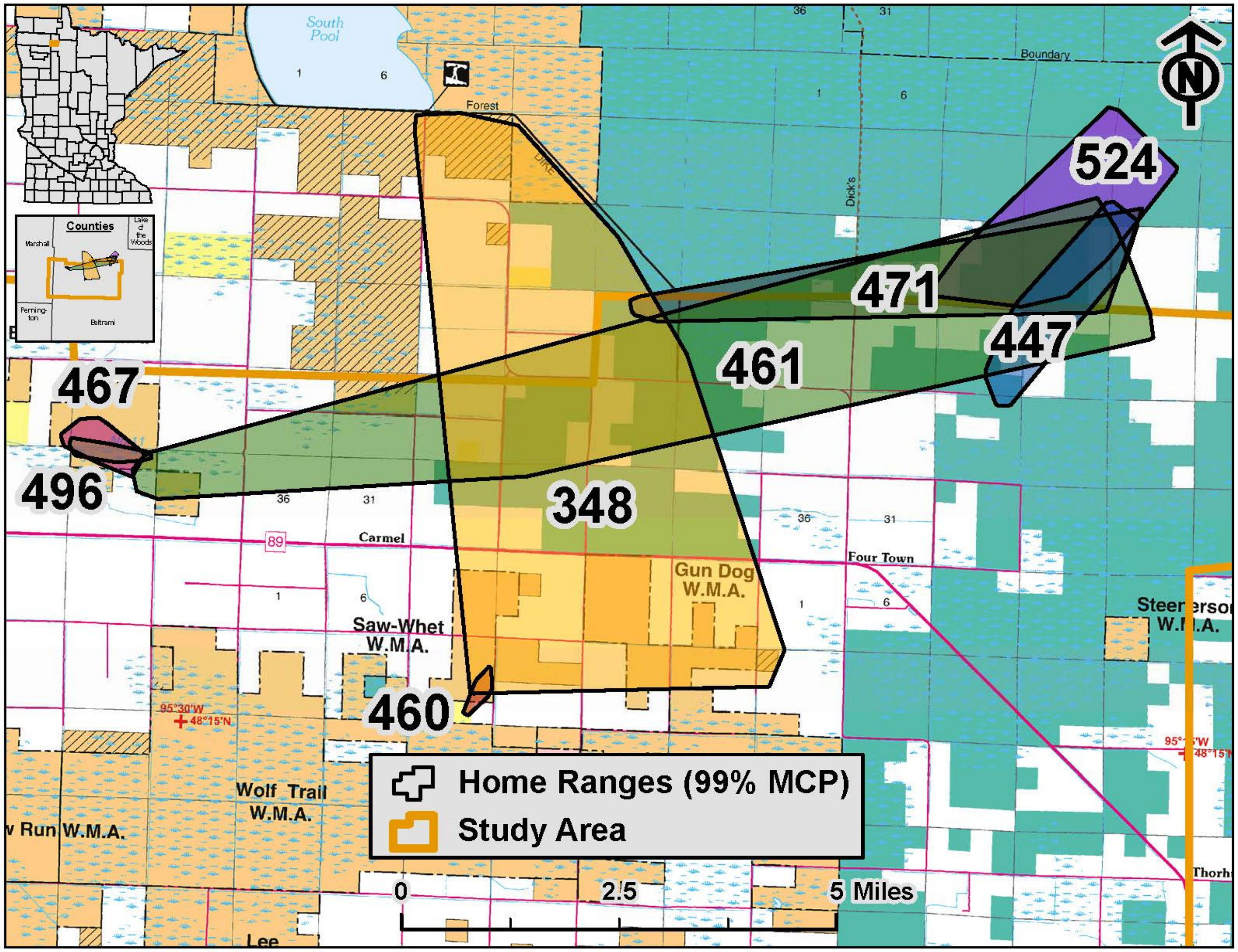

Figure 3. Home ranges, determined by 99% minimum convex polygons, for white-tailed deer (*n* = 8) that died or slipped their radiocollar during the study period, January–April 2011, northwestern Minnesota.

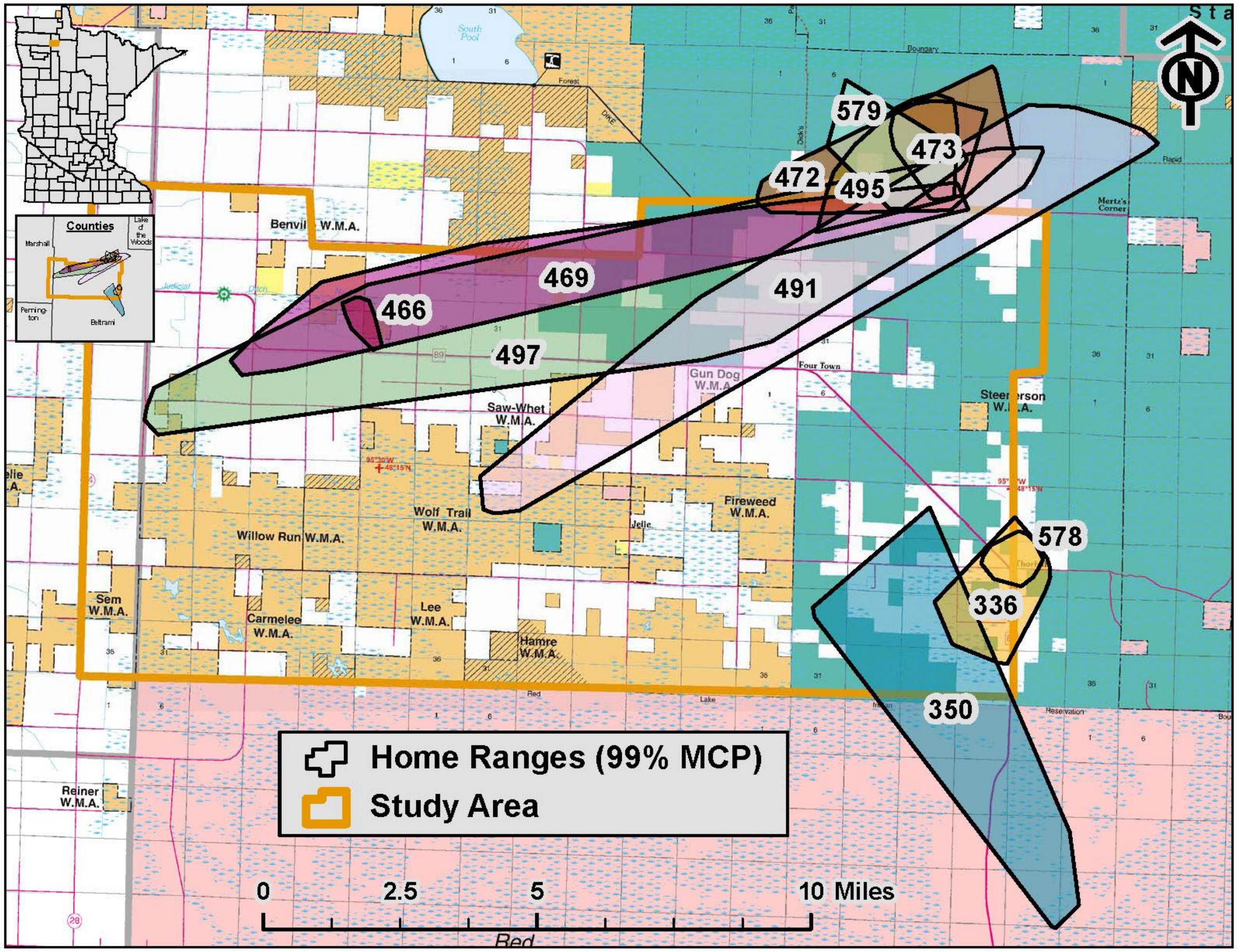

Figure 4. Home ranges, determined by 99% minimum convex polygons, for white-tailed deer (*n* = 11) alive as of 14 June 2011, northwest Minnesota.

MANAGING BOVINE TUBERCULOSIS IN WHITE-TAILED DEER IN NORTHWESTERN MINNESOTA: A 2010 PROGRESS REPORT

Michelle Carstensen[1], Erika Butler, Erik Hildebrand, and Louis Cornicelli

SUMMARY OF FINDINGS

Bovine tuberculosis (bTB), first detected in northwestern Minnesota in 2005, has since been found in 12 cattle operations and 27 free-ranging white-tailed deer (*Odocoileus virginianus*). Both deer and cattle have the same strain of bTB, which has been identified as one that is consistent with the disease found in cattle in the southwestern United States and Mexico. The Board of Animal Health (BAH) has been leading efforts to eradicate the disease in Minnesota's cattle, which have included the depopulation of all infected herds, a buy-out program that removed 6,200 cattle from the affected area, and mandatory fencing of stored feeds on remaining farms. In response to the disease being detected in cattle, the Minnesota Department of Natural Resources (MNDNR) began surveillance efforts in free-ranging white-tailed deer within a 24-km (15-mi) radius of the infected farms in fall 2005. To date, 26 of the 27 deer infected with bTB were sampled within a 425-km^2 (164-mi$^{2)}$ area, called the bTB Core, which is centered in Skime, Minnesota, and encompasses 8 of the previously infected cattle farms. In total, 1,639 hunter-harvested deer were tested for bTB in northwestern Minnesota during fall 2010, with no positive cases detected. This marks the first year that no new infected cases were detected in wild deer. An aerial survey estimated the population of the bTB Core to be 531 (SE = 95) deer in January 2011. The absence of new infected deer resulted in a suspension of targeted removal operations using ground sharpshooting over winter. A recreational feeding ban, instituted in November 2006 in a 10,360-km^2 (4,000-mi^2) region in northwestern MN to help reduce the risk of deer to deer transmission of the disease, remains in effect. Under a current agreement among the United States Department of Agriculture (USDA), BAH, and MNDNR, hunter-harvested deer surveillance will continue to monitor infection in the local deer population, and any further aggressive management actions (e.g., sharpshooting deer in key locations) will be dependent on future surveillance results.

INTRODUCTION

Bovine tuberculosis is an infectious disease that is caused by the bacterium *Mycobacterium bovis*. Bovine tuberculosis primarily affects cattle; however, other mammals may become infected. The disease was first discovered in 5 cattle operations in northwestern Minnesota in 2005. Since that time, 7 additional herds were found infected; resulting in a reduction of the state's bTB accreditation to Modified Accredited in early 2008. In fall 2008, Minnesota was granted a split-state status for bTB accreditation that maintained only a small area (6,915 km^2 [2,670 mi^2]) in northwestern Minnesota as "Modified Accredited," allowing the remainder of the state to advance to "Modified Accredited Advanced." To date, 27 wild deer have been found infected with the disease in northwestern MN, which can be attributed to a spillover of the disease from infected cattle. In 2010, The USDA upgraded Minnesota's bTB accreditation to Modified Accredited Advanced within the split-state zone and bTB-free throughout the remainder of the state. Although bTB was once relatively common in U. S. cattle, historically, it has been a very rare disease in wild deer. Prior to 1994, only 8 wild white-tailed and mule deer (*O. hemionus)* had been reported with bTB in North America. In 1995, bTB was detected in wild deer in Michigan and do serve as a reservoir of the disease in that state.

Bovine tuberculosis is a progressive, chronic disease. It is spread primarily through the exchange of respiratory secretions between infected and uninfected animals. This transmission usually happens when animals are in close contact with each other. Animals may also become infected with bTB by ingesting the bacteria from eating contaminated feed. Incubation periods

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[1] Corresponding author's email: michelle.carstensen@state.mn.us

can vary from months to years from time of infection to the development of clinical signs. The lymph nodes in the animal's head usually show infection first, and as the disease progresses, lesions (yellow or tan, pea-sized nodules) will begin to develop throughout the thoracic cavity. In severely infected deer, lesions can usually be found throughout the animal's entire body. Hunters do not always readily recognize small lesions in deer, as they may not be visible when field-dressing deer. In fact, most infected deer appear healthy. While it is possible to transmit bTB from animals to people, the likelihood is extremely low. Most human tuberculosis is caused by the bacteria *M. tuberculosis*, which is spread from person to person and rarely infects animals.

METHODS

In 2010, we developed a fall hunter-harvested surveillance strategy to meet the sampling goals established in a recently renegotiated Memorandum of Understanding (MOU) between the USDA and both the MNDNR and BAH. It requires 1,000 deer to be tested for bTB within the Modified Accredited Advanced Zone (MAAZ). Distribution of these samples was to include 500 from within the bTB Management Zone and 500 from the area outside the bTB Management Zone, but within the MAAZ. The MNDNR further defined these goals to specify that the 500-sample goal from within the bTB Management Zone must include at least 200 samples from the bTB Core Area.

At the registration stations, hunters were asked to voluntarily submit lymph node (LN) samples for bTB-testing. Hunter information was recorded, including the hunter's name, telephone number, MNDNR number, and location of the kill. Maps were provided to assist the hunters in identifying the location (township, range, section, and quarter-section) of the kill. Cooperating hunters were given a cooperator's patch and entered into a raffle for a firearm donated by the Minnesota Deer Hunter's Association (MDHA). In addition, the Roseau River chapter of MDHA raffled additional firearms and a life-time deer hunting license for hunters that submitted samples from within the bTB Management Zone or bTB Core Area.

Sampling procedures included a visual inspection of the chest cavity of the hunter-killed deer. Six cranial LNs (parotid, submandibular, and medial retropharyngeal) were visually inspected for presence of gross lesions and collected for further testing. Samples were submitted to the Veterinary Diagnostic Laboratory (VDL) at the University of Minnesota for histological examination and acid-fast staining. All samples were then pooled in groups of 5 and sent to the National Veterinary Services Laboratories (NVSL) in Ames, Iowa for culture. Any suspect carcasses (e.g., obvious lesions in chest cavity or head) were voluntarily surrendered at the registration stations and the hunter was issued a replacement deer license at no charge. Suspect carcasses were transported in their entirety to the VDL for further testing.

In early winter, MNDNR conducted an aerial survey of the bTB Core Area to assess deer numbers and distribution (Figure 1). This information was used to guide future management activities and estimate the percentage of deer removed from the area through hunting and agency culling.

RESULTS AND DISCUSSION

In fall 2010, we collected 1,639 samples from hunter-harvested deer; 1,437 samples from within the MAAZ and 202 samples outside the zone (Figure 2). Thus, MNDNR collected nearly 1.5x the overall sampling goal set forth by the MOU with USDA. Further, the sampling distribution met the guidelines of the MOU for samples collected within the bTB Management Zone (n = 575) and outside this zone, but within the MAAZ (n = 862) (Figure 2). The MNDNR achieved 92% of the specified goal of collecting at least 200 samples from within the bTB Core Area.

Testing of all lymph node samples at NVSL confirmed that there were no positive cases of bTB detected during the fall 2010 surveillance. Thus, 2010 marks the first complete year (including winter 2010 sharpshooting in the bTB Core Area) in which no new cases of the disease were detected in wild deer. Apparent prevalence of bTB in the local deer population, sampled throughout a 4,481–6,915-km^2 (1,730–2,670 mi^2) Surveillance Zone, indicates a significant decreasing trend from 2006 to 2010 (Table 1, Figure 3). Further, disease prevalence in the bTB Core Area has decreased dramatically from 2007 to 2010 (Table 1, Figure 3). Although disease prevalence estimates in the TB Core Area are biased due to the limited geographic distribution of TB-positive deer and the increased probability of detecting a positive individual, the decreasing trend is consistent with the large-scale surveillance of the local deer populations in the fall.

Aerial survey results from January 2011 estimated that the deer population in the bTB Core Area was a minimum of 531 (SE = 95 deer, Figure 1). This was slightly higher than the February 2010 population estimate of 422 (SE = 64; Figure 4, Table 2). Aggressive deer removal in the bTB Core Area by liberalized hunting, disease management permits, landowner shooting permits, and targeted sharpshooting allowed us to reduce the deer population in this 425-km^2 (164-mi^2) area by approximately 55% from 2006 to 2010, but clearly, maintaining deer numbers at a low level will remain difficult. It is likely that the bTB Core Area is home to both migratory and resident deer, some of which may move out of the zone to spring-summer-fall or winter ranges during the year. It is further likely that deer from the surrounding area are immigrating into the bTB Core Area as deer numbers are reduced and habitat availability increases. The moderately severe winter of 2010–2011 may have played a role in increased deer movement into the bTB Core Area, which provides good wintering habitat, and might explain the slight increase in estimated deer numbers.

The proximity of the TB-infected deer to infected cattle herds, the strain type, and the fact that disease prevalence (< 0.1%) is low, supports our theory that this disease spilled-over from cattle to wild deer in this area of the state. To date, we have sampled 9,783 deer in the northwest since 2005;27 were confirmed culture-positive deer (Figure 5). Further, the lack of infected yearlings or fawns and limited geographic distribution of infected adults further supports that deer are not a wildlife reservoir for this disease in Minnesota (Carstensen and DonCarlos, 2011). In November 2006, a ban on recreational feeding of deer and elk (*Cervus elaphus*) was instituted over a 10,360-km^2 (4,000-mi^2) area to help reduce the risk of disease transmission among deer and between deer and livestock (Figure 6). Enforcement officers continue to enforce this rule and compliance is very high within the bTB Management Zone.

As part of the requirements to regain bTB-Free accreditation, the USDA has required BAH to test all cattle herds within the Modified Accredited Advanced Zone annually, with additional movement restrictions for farms located within the bTB Management Zone. The BAH has submitted an application for status upgrade to USDA, and a decision is expected by September 2011. If approved, Minnesota would regain its bTB-free status throughout the entire state, removing our current split-state status entirely. What this will mean for continued surveillance in both cattle and deer is unknown. The MNDNR is committed to assisting BAH in regaining and maintaining Minnesota's bTB-free status. The MNDNR will conduct fall hunter-harvested surveillance in 2011, although surveillance goals and a timeline for continued surveillance beyond 2011 will likely be negotiated with USDA this fall.

ACKNOWLEDGMENTS

There is no way to complete a project of this scale without the assistance and leadership from St. Paul and regional staff, including E. Boggess, M. DonCarlos, D. Simon, D. Schad, P. Telander, J. Williams, and M. Carroll. For all the help with field collections, we'd like to thank area staff from Thief Lake, Red Lake, Norris Camp, and Thief River Falls, E. Hildebrand (Wildlife Health Specialist), M. Dexter (Fish and Wildlife Specialist), as well as students and faculty from the University of Minnesota, College of Veterinary Medicine. Also thanks to R. Wright, T. Erickson, and the Enforcement pilots (Tom Pfingsten and John Heineman) for conducting a deer survey within the bTB Management Zone, as well as identifying illegal deer feeding activities. Also thanks to J. Giudice for analyzing the survey data. We had an excellent team of GIS support, including S. Benson, J. Adams, R. Wright, and C. Scharenbroich. We also want to recognize the support from the USDA-WS Disease Program, including P. Wolf (St. Paul), D. Bruning (WA), D. Sinnett (AK), E. Blizzard (CO), J. Lujan (SC), D. Cavanaugh (GA), S. Goldstein (HI), A. Randall (NJ), B. Bodenstein (WI), and T. DeLiberto (CO).

LITERATURE CITED

Carstensen, M., and M. W. DonCarlos. 2011. Preventing the establishing of a wildlife disease reservoir: a case study of bovine tuberculosis in wild deer in Minnesota, USA. Veterinary Medicine International, Volume 2011, Article ID 413240, 10 pages, doi:10.4061/2011/413240.

Table 1. Number of deer sampled for bovine tuberculosis (TB) and testing results listed by sampling strategy, 2005 2010, northwestern Minnesota.

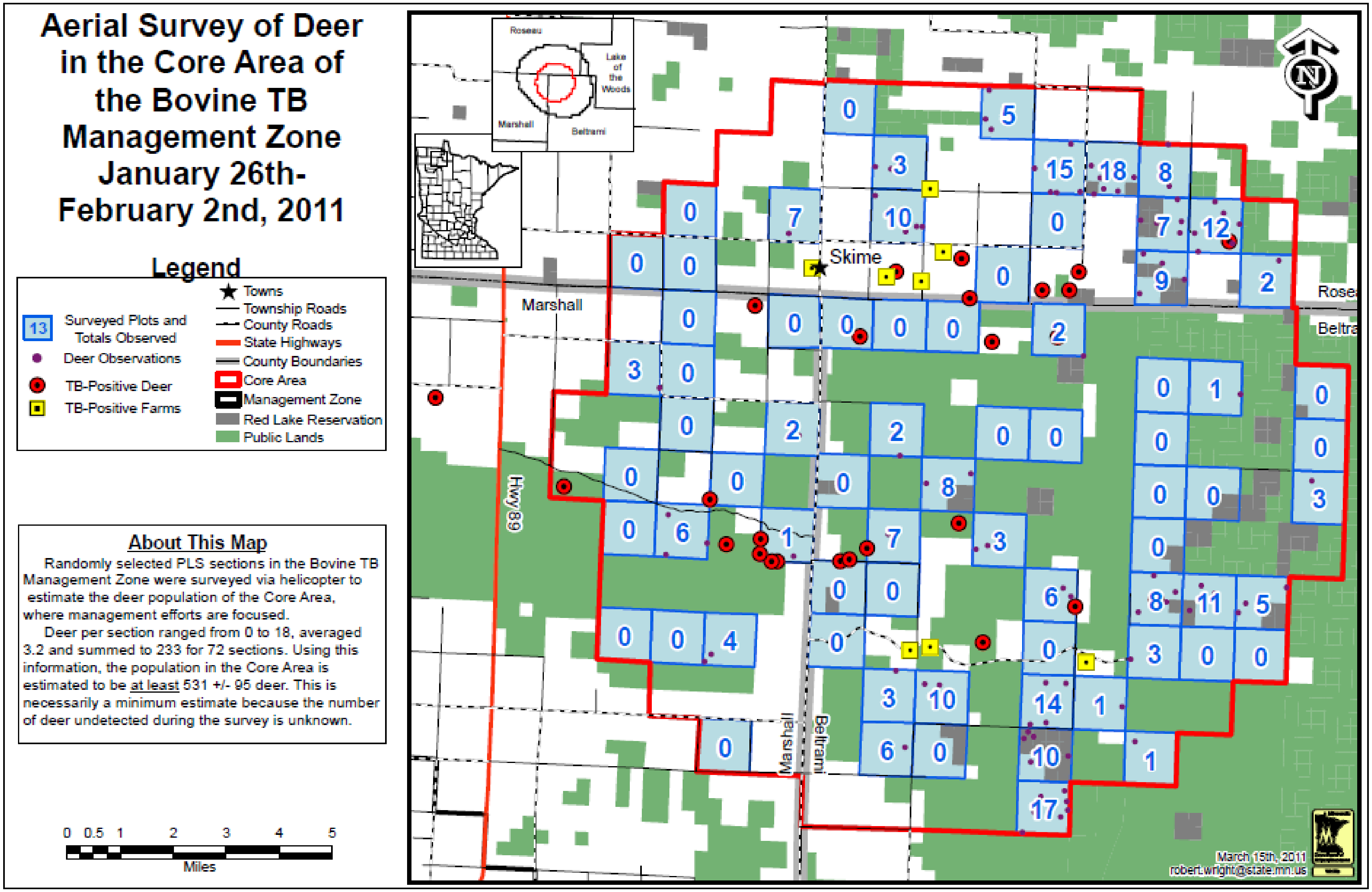
Sampling strategy	2005	2006	2007	2008	2009	2010	Totals
Hunter-harvested (Oct-Jan)	474	942	1,166	1,246	1,488	1,639	6,955
# TB-positive	1	5	5	0	1	0	
Apparent prevalence (%)	0.21	0.53	0.43	0.0	0.07	0.0	
Sharpshooting (Feb-May)	0	0	488	937	738	450	2,613
# TB-positive			6	6	2	0	
Apparent prevelance			1.23	0.64	0.27	0.0	
Landowner/tenant	0	90	0	125	0	0	215
# TB-positive		1		0			
Total deer tested	474	1,032	1,654	2,308	2,226	2,089	9,783
Total # TB-positive	1	6	11	6	3	0	27

Table 2. Population estimates and 95% confidence intervals of deer within the Bovine Tuberculosis Core Area, 2007–2011, northwestern Minnesota.[a,b]

Year	Aircraft	Design	Var.est	n	N	Srate	Svar	SE	Xbar	SE	95% CI	PopEst	SE	95% CI	CV(%)	RP(%)
2007	OH-58	StRS3	SRS	72	164	0.439	NA	NA	5.7	0.46	4.9–6.5	935	76.0	784–1086	8.1	16.2
2008	OH-58	GRTS.SRS	Local	72	164	0.439	21.94	4.53	4.9	0.56	3.8–6.0	807	75.2	659–954	9.3	18.3
2009	Enstrom	GRTS.StRS2	Local	79	164	0.482	20.63	2.56	4.1	0.27	3.5–4.6	664	44.4	577–751	6.7	13.1
2010	OH-58	GRTS.SRS	Local	72	164	0.439	29.30	6.70	2.6	0.39	1.8–3.3	422	64.4	296–548	15.3	30.0
2011	OH-58	GRTS.SRS	Local	72	164	0.439	21.01	2.70	3.2	0.30	2.7–3.8	531	48.6	436–627	9.2	18.0

[a] Population estimate = estimated *minimum* number of deer present during the sampling interval. Estimates are not adjusted for sightability (but intensive survey is designed to minimize visibility bias), and deer movement between sample plots is assumed to be minimal or accounted for via survey software.

[b] 95% confidence intervals (CI) are based on sampling variance only (adjusted for spatial correlation in 2008–2011); they do not include uncertainty associated with sightabilty or animal movements (temporal variation due to animals moving onto or off the study area).



Aerial Survey of Deer in the Core Area of the Bovine TB Management Zone January 26th-February 2nd, 2011

About This Map

Randomly selected PLS sections in the Bovine TB Management Zone were surveyed via helicopter to estimate the deer population of the Core Area, where management efforts are focused.

Deer per section ranged from 0 to 18, averaged 3.2 and summed to 233 for 72 sections. Using this information, the population in the Core Area is estimated to be at least 531 +/- 95 deer. This is necessarily a minimum estimate because the number of deer undetected during the survey is unknown.

Figure 1. Results of aerial white-tailed deer survey of the Bovine Tuberculosis Core Area in January 2011, northwestern Minnesota.

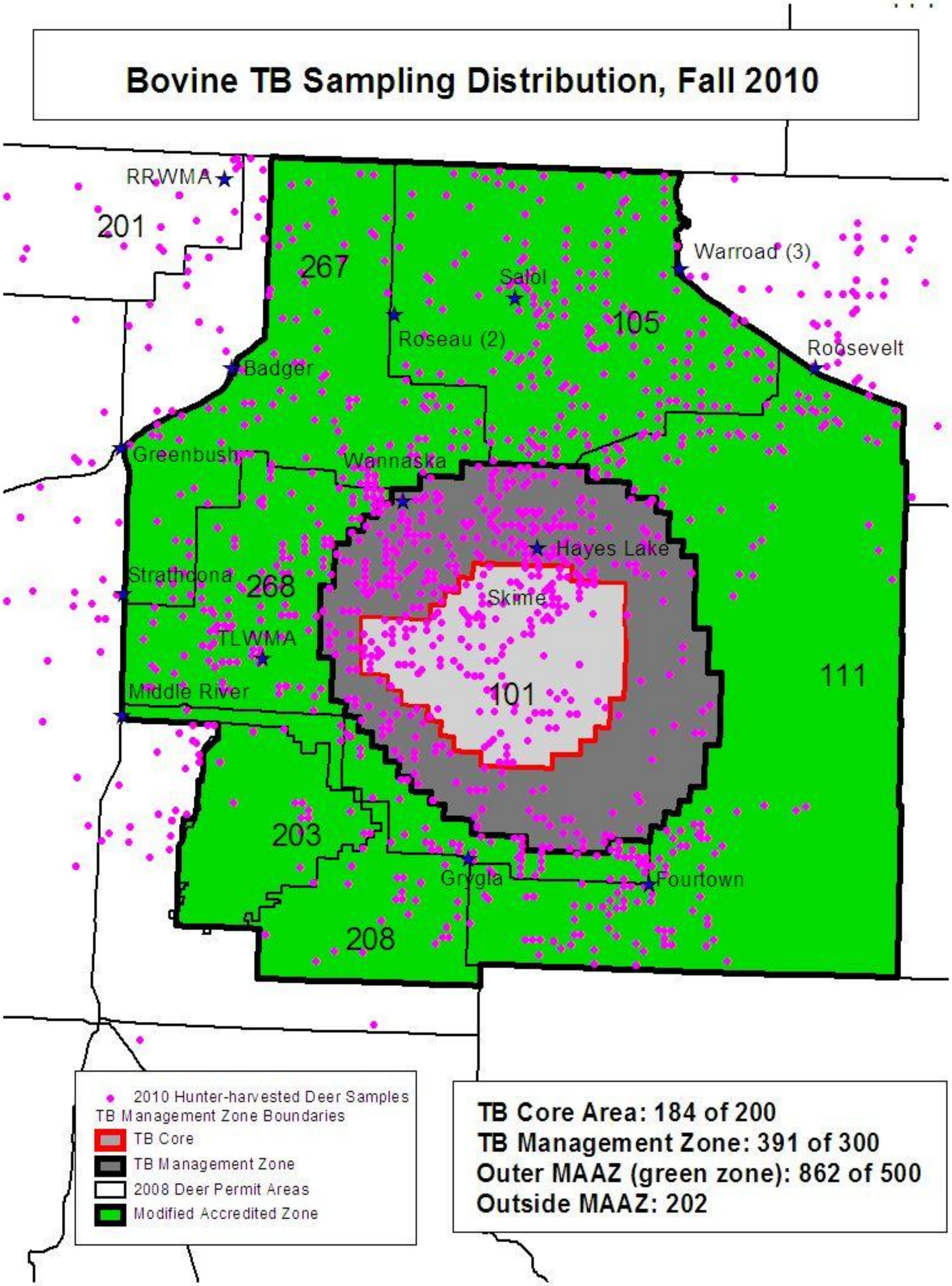


Figure 2. Locations of hunter-harvested deer (*n* = 1,639) sampled for bovine tuberculosis (TB) during fall 2010 in northwestern Minnesota.

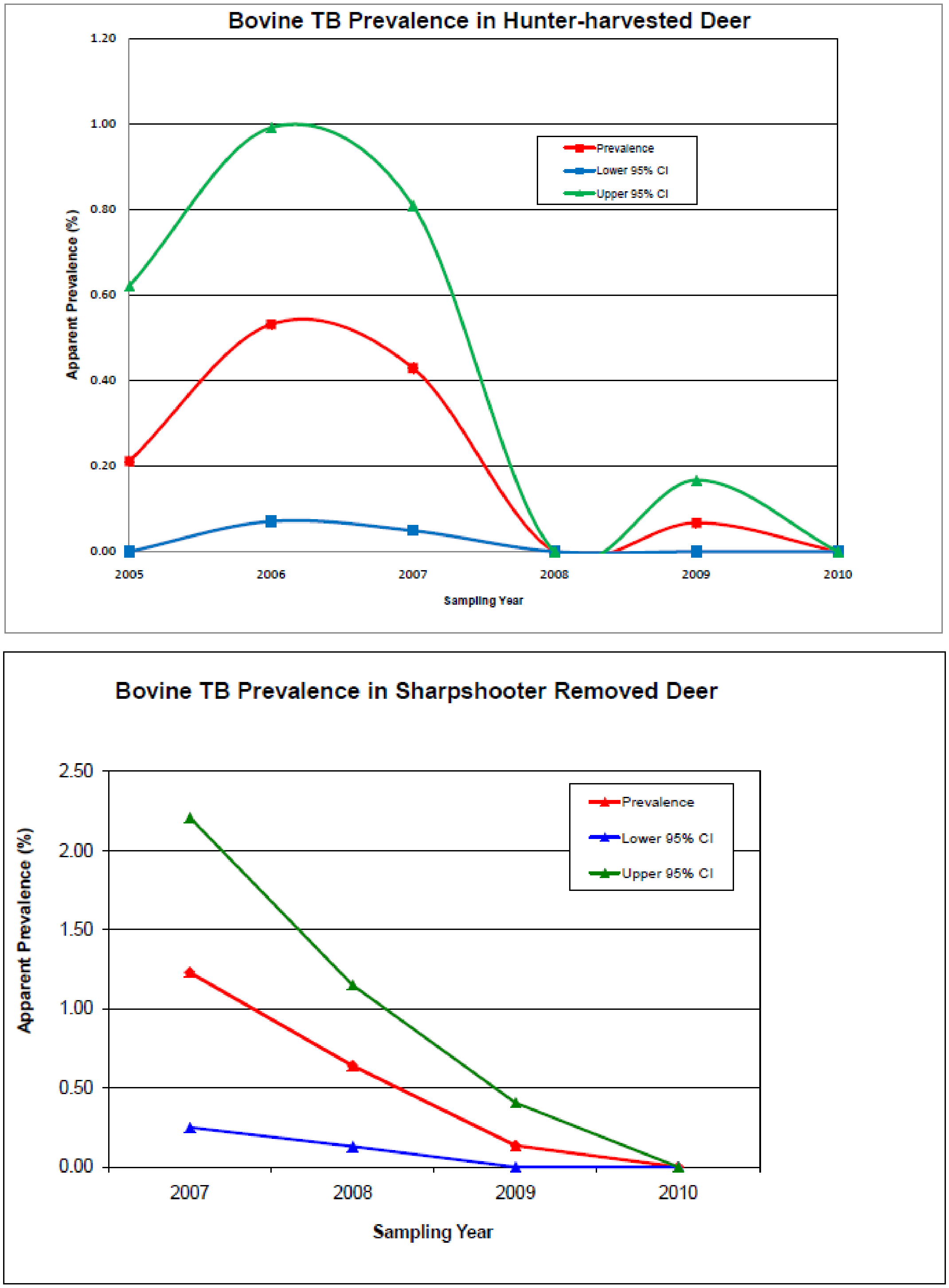


Figure 3. Prevalence of bovine tuberculosis (TB) in hunter-harvested deer from 2005 to 2010 in the BovineTB Surveillance Zone and disease prevalence from sharpshooter removed deer from 2007 to 2010 in the Bovine TB Core Area, northwestern Minnesota.

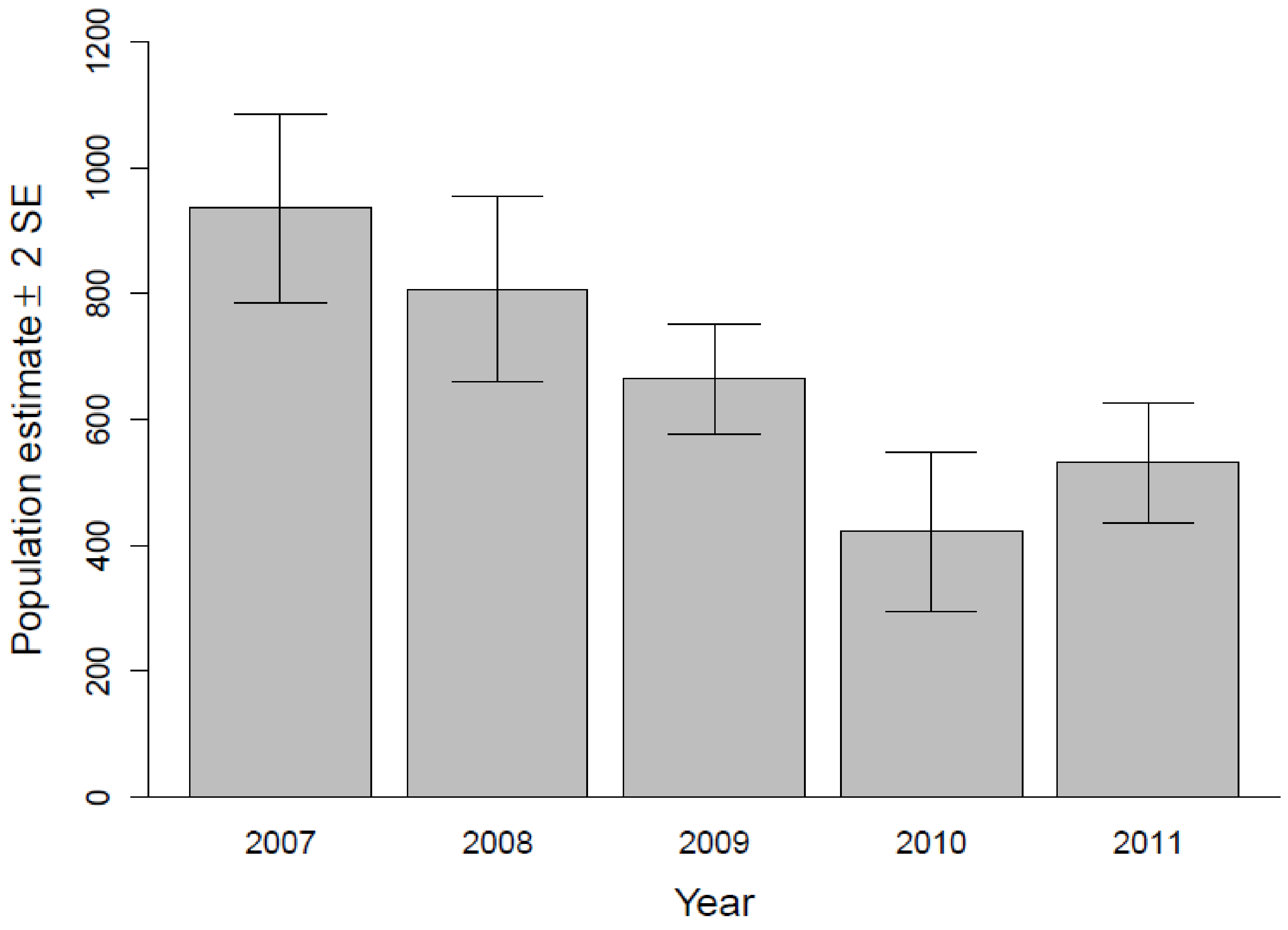


Figure 4. Population estimate of deer within the Bovine Tuberculosis Core Area, winters 2007–2011, northwestern Minnesota.

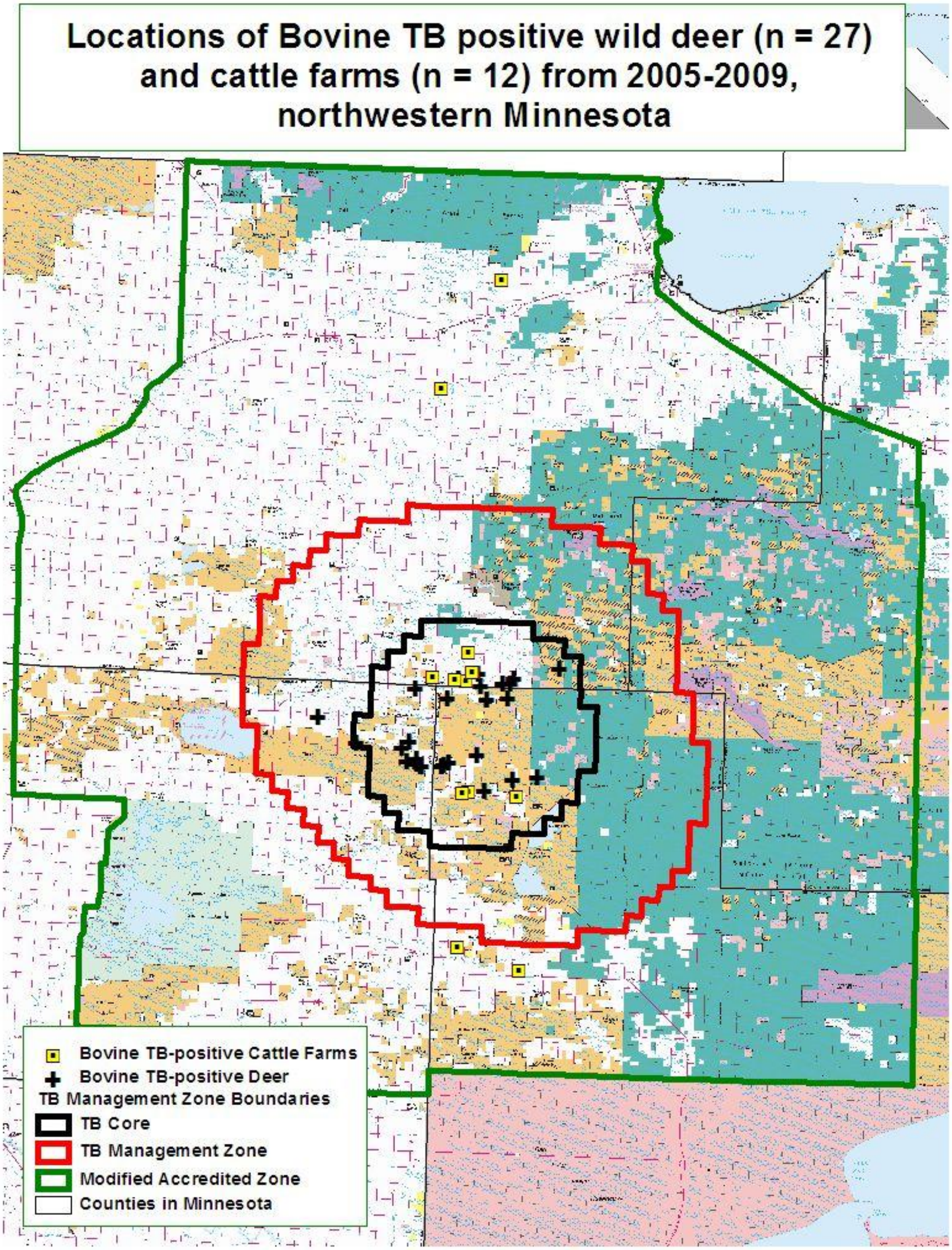
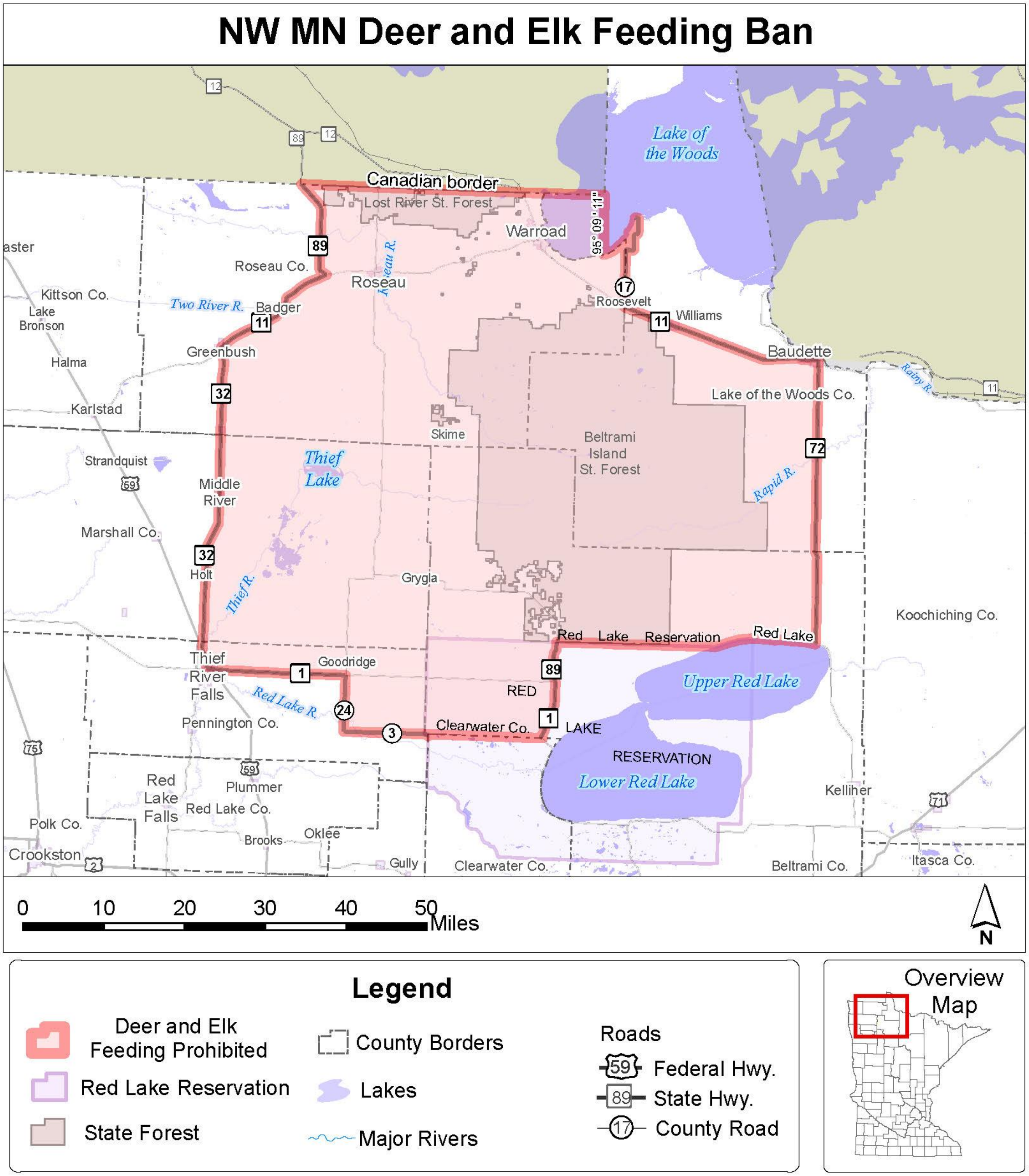


Figure 5. Locations of all white-tailed deer found infected (*n* = 27) with bovine tuberculosis (TB) since fall 2005 in northwestern Minnesota; the 12 previously-infected cattle operations are included.



Deer and Elk Feeding Ban Boundary Description

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

Figure 6. Area in northwestern Minnesota where recreational feeding of deer and elk was banned in November 2006, as a preventative measure to reduce risk of disease transmission.

PREVENTING THE ESTABLISHMENT OF A WILDLIFE DISEASE RESERVOIR: A CASE STUDY OF BOVINE TUBERCULOSIS IN WILD DEER IN MINNESOTA, USA[1]

Michelle Carstensen and Michael W. DonCarlos

ABSTRACT

Bovine tuberculosis (bTB) has been found in 12 cattle operations and 27 free-ranging white-tailed deer (*Odocoileus virginianus*) in northwestern Minnesota, following the state's most recent outbreak of the disease in 2005 in the northwestern part of the state. Both deer and cattle have the same strain of bTB. The Minnesota Board of Animal Health has been leading efforts to eradicate the disease in Minnesota's cattle, which have included the depopulation of all infected herds, a cattle buy-out program, and mandatory fencing of stored feeds. The Minnesota Department of Natural Resources (MNDNR) began surveillance efforts in free-ranging white-tailed deer in fall 2005. All bTB-infected deer have been found within a 16-km^2 area in direct association with infected cattle farms. Aggressive efforts to reduce deer densities through liberalized hunting and sharpshooting have resulted in a 55% decline in deer densities. Also, recreational feeding of wild deer has been banned. Disease prevalence in deer has decreased from 1.2% in 2005 to an undetectable level in 2010. Minnesota's primary goal has been the eradication of bTB from both deer and cattle. The aim of this paper is to describe the primary management strategies implemented by MNDNR to prevent the establishment of a wildlife disease reservoir in free-ranging white-tailed deer. These strategies included, (1) rapid response to initial disease detection, (2) follow-through on monitoring the outbreak with adequate surveillance, (3) recognizing when monitoring must switch to management, (4) aggressively reducing transmission potential by reducing deer densities, limiting recreational feeding and mitigating risks at the cattle-wildlife interface, and (5) evaluation of efforts and adjusting as needed.

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[1] From published paper: Carstensen, M., and M. W. DonCarlos. 2011. Preventing the establishment of a wildlife disease reservoir: a case study of bovine tuberculosis in wild deer in Minnesota, USA. Veterinary Medicine International, Volume 2011, Article ID 413240, 10 pages, doi:10.4061/2011/413240

PUBLIC ACCEPTANCE AS A DETERMINANT OF MANAGEMENT STRATEGIES FOR BOVINE TUBERCULOSIS IN FREE-RANGING U.S. WILDLIFE[1]

Michelle Carstensen, Daniel J. O'Brien[2], and Stephen M. Schmitt[2]

ABSTRACT

When bovine tuberculosis (bTB) is detected in free-ranging wildlife populations, preventing geographic spread and the establishment of a wildlife reservoir requires a rapid, often aggressive response. Public tolerance can exert a significant effect on potential control measures available to managers, and thus on the success of disease management efforts. Separate outbreaks of bTB in free-ranging white-tailed deer (*Odocoileus virginianus*) in 2 midwestern states provide a case study. In Minnesota, bTB was first discovered in cattle in 2005 and subsequently in deer. To date, 12 beef cattle farms and 26 white-tailed deer have been found infected with the disease. From 2005 to 2008, disease prevalence in deer has decreased from 0.4% (SE = 0.2%) to < 0.1% and remained confined to a small (< 425 km^2) geographic area. Deer population reduction through liberalized hunting and targeted culling by ground sharpshooting and aerial gunning, combined with a prohibition on baiting and recreational feeding, have likely been major drivers preventing disease spread thus far. Without support from cattle producers, deer hunters and the general public, as well as politicians, implementation of these aggressive strategies by state and federal authorities would not have been possible. In contrast, Michigan first discovered bovine bTB in free-ranging deer in 1975, and disease management efforts were not instituted until 1995. The first infected cattle herd was diagnosed in 1998. Since 1995, disease prevalence in free-ranging deer has decreased from 4.9% to 1.8% in the 1500-km^2 core outbreak area. Culture positive deer have been found as far as 188 km from the core area. Liberalized harvest and restrictions on baiting and feeding have facilitated substantial reductions in prevalence. However, there has been little support on the part of hunters, farmers or the general public for more aggressive population reduction measures such as culling, and compliance with baiting and feeding restrictions has been variable and often problematic. We compare and contrast the Minnesota and Michigan outbreaks with respect to temporal, social, economic, and logistical factors that shape public attitudes toward aggressive disease control strategies, the limitations these factors place on management, and the implications for bTB eradication from wildlife reservoirs in the U. S.

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[1] Abstract from published paper: Carstensen, M., D. J. O'Brien, and S. M. Schmitt. 2011. Public acceptance as a determinant of management strategies for bovine tuberculosis in free-ranging U. S. wildlife. Veterinary Microbiology 151:200–204, doi:10.1016/j.vetmic.2011.02.046

[2] Wildlife Disease Laboratory, Michigan Department of Natural Resources, 4125 Beaumont Rd., Room 250, Lansing, MI 48910

Wildlife Biometrics Unit
Carlos Avery Wildlife management Area
5463 – C West Broadway
Forest Lake, Minnesota 55025
(651) 296-5200

COMPARATIVE INTERPRETATION OF COUNT, PRESENCE-ABSENCE AND POINT METHODS FOR SPECIES DISTRIBUTION MODELS[1]

Geert Aarts[2], John Fieberg, and Jason Matthiopoulos[3]

ABSTRACT

1. The need to understand the processes shaping population distributions has resulted in a vast increase in the diversity of spatial wildlife data, leading to the development of many novel analytical techniques that are fit-for-purpose. One may aggregate location data into spatial units (e.g. grid cells), and model the resulting counts or presence-absences as a function of environmental covariates. Alternatively, the point data may be modeled directly, by combining the individual observations with a set of random or regular points reflecting habitat availability, a method known as a use-availability design (or, alternatively a presence–pseudo-absence or case-control design).

2. Although these spatial point, count and presence-absence methods are widely used, the ecological literature is not explicit about their connections and how their parameter estimates and predictions should be interpreted. The objective of this study is to recapitulate some recent statistical results and illustrate that under certain assumptions, each method can be motivated by the same underlying spatial Inhomogeneous Poisson point-process (IPP) model in which the intensity function is modeled as a log-linear function of covariates.

3. The Poisson likelihood used for count data is a discrete approximation of the IPP likelihood. Similarly, the presence-absence design will approximate the IPP likelihood, but only when spatial units (i.e., pixels) are extremely small (Baddeley et al., 2010). For larger pixel sizes, presence-absence designs do not differentiate between 1 or multiple observations within each pixel, hence leading to information loss.

4. Logistic regression is often used to estimate the parameters of the IPP model using point data. Although the response variable is defined as 0 for the availability points, these 0s do not serve as true absences as is often assumed; rather, their role is to approximate the integral of the denominator in the IPP likelihood (Warton and Shepherd 2010). Due to this common misconception, the estimated exponential function of the linear predictor (i.e., the resource selection function) is often assumed to be proportional to occupancy. Like IPP and count models, this function is proportional to the expected density of observations.

5. Understanding these (dis-)similarities between different species distribution modeling techniques should improve biological interpretation of spatial models, and therefore advance ecological and methodological cross-fertilization.

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[1] Abstract from paper accepted for publication in Methods in Ecology and Evolution

[2] Royal Netherlands Institute for Sea Research (NIOZ), P.O. Box 59, 1790 AB Den Burg, The Netherlands

[3] Scottish Oceans Institute, School of Biology, University of St. Andrews, East Sands, St. Andrews, Fife KY16 8LB Scotland, United Kingdom

A BAYESIAN HIERARCHICAL OCCUPANCY MODEL FOR TRACK SURVEYS CONDUCTED IN A SERIES OF LINEAR, SPATIALLY CORRELATED SITES[1]

Chrisna Aing[2], Sarah Halls[2], Kiva Oken[2], Robert Dobrow[2], and John Fieberg.

ABSTRACT

1. Natural resource agencies often rely on surveys of animal sign (e.g., scat, scent marks, tracks) for population assessment, with repeat surveys required to model and account for uncertain detection. Using river otter *Lontra canadensis* snow track survey data as a motivating example, we develop a 3-level occupancy model with parameters that describe (1) site-level occupancy probabilities, (2) otter movement (and thus, track availability), and (3) recorded presence-absence of tracks (conditional on the availability of tracks for detection).

2. We incorporated several recent developments in occupancy modeling, including the presence of both false negatives and false positives, spatial and temporal correlation, and repeated sampling across distinct observers.

3. We investigated optimal allocation of sampling effort (e.g., within and among snowfall events) using simulations. We also compared models that allowed site-level occupancy and track laying processes to be spatially correlated to models that assumed independence among sites.

4. Both types of models (independence and spatial) performed well across a range of simulated parameter values, but the spatial model resulted in more accurate point estimates for detection parameters and credibility intervals with better coverage rates when data were spatially correlated. When applied to real data, the spatial model resulted in a higher estimate of the occupancy rate ( ) than the baseline model (0.82 versus 0.59). A minimum of 15-20 helicopter flights, distributed among at least three unique snow events, were needed to meet precision goals (standard error < 0.05).

5. Synthesis and applications. We describe a flexible and robust occupancy modeling framework that accounts for heterogeneous detection rates in surveys of animal sign. The method allows for spatially correlated sites, and should have broad relevance to surveys conducted by many natural resource agencies.

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[1] Abstract from paper provisionally accepted for publication in the Journal of Applied Ecology

[2] Department of Mathematics, Carleton College, Northfield, Minnesota 55057

SPENDING DEGREES OF FREEDOM IN A POOR ECONOMY: A CASE STUDY OF BUILDING A SIGHTABILITY MODEL FOR MOOSE IN NORTHEASTERN MINNESOTA[1]

John Giudice, John Fieberg, Mark Lenarz

ABSTRACT

Sightability models are binary logistic-regression models used to estimate and adjust for visibility bias in wildlife-population surveys. Like many models in wildlife and ecology, sightability models are typically developed from small observational data sets with many candidate predictors. Aggressive model-selection methods are often employed to choose a ‘best’ model for prediction and effect estimation, despite evidence that such methods can lead to overfitting (i.e., selected models may describe random error or noise rather than true predictor-response curves) and poor predictive ability. We used moose-sightability data from northeastern Minnesota (2005–2007) as a case study to illustrate an alternative approach, which we refer to as degrees-of-freedom (df) spending: sample-size guidelines are used to determine an acceptable level of model complexity and then a pre-specified model is fit to the data and used for inference. For comparison, we also constructed sightability models using AIC step-down procedures and model averaging (based on a small set of models developed using df-spending guidelines). We used bootstrap procedures to mimic the process of model-fitting and prediction, and to compute an index of overfitting, expected predictive accuracy, and model-selection uncertainty. The index of overfitting increased 13% when the number of candidate predictors was increased from 3 to 8 and a ‘best’ model was selected using step-down procedures. Likewise, model-selection uncertainty increased when the number of candidate predictors increased. Model averaging (based on R = 30 models with 1–3 predictors) effectively “shrunk” regression coefficients toward zero and produced similar estimates of precision to our 3-df pre-specified model. As such, model averaging may help to guard against overfitting when too many predictors are considered (relative to available sample size). The set of candidate models will influence the extent to which coefficients are shrunk toward 0, which has implications for how 1 might apply model averaging to problems traditionally approached using variable-selection methods. We often recommend the df-spending approach in our consulting work, because it is easy to implement and it naturally forces investigators to think carefully about their models and predictors. Nonetheless, similar concepts should apply whether 1 is fitting 1 model or using multi-model inference. For example, model-building decisions should consider the effective sample size, and potential predictors should be screened (without looking at their relationship to the response) for missing data, narrow distributions, collinearity, potentially overly influential observations, and measurement errors (e.g., via logical error checks).

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[1] Abstract is from a paper that has been accepted for publication in the Journal of Wildlife Management

GENERALIZED FUNCTIONAL RESPONSES FOR SPECIES DISTRIBUTIONS[1]

Jason Matthiopoulos[2], Mark Hebblewhite[3], Geert Aarts[4], and John Fieberg.

ABSTRACT

Researchers employing resource selection functions (RSFs) and other related methods aim to detect correlates of space-use and mitigate against detrimental environmental change. However, an empirical model fit to data from 1 place or time is unlikely to capture species responses under different conditions, because organisms respond nonlinearly to changes in habitat availability. This phenomenon, known as a functional response in resource selection, has been debated extensively in the RSF literature, but continues to be ignored by practitioners for lack of a practical treatment. We therefore extend the RSF approach to enable it to estimate generalized functional responses (GFRs) from spatial data. GFRs employ data from several sampling instances characterized by diverse profiles of habitat availability. By modeling the regression coefficients of the underlying RSF as functions of availability, GFRs can account for environmental change and thus predict population distributions in new environments. We formulate the approach as a mixed-effects model so that it is estimable by readily available statistical software. We illustrate its application using (1) simulation and (2) wolf home-range telemetry. Our results indicate that GFRs can offer considerable improvements in estimation speed and predictive ability over existing mixed-effects approaches.

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[1] Abstract from published paper: Matthiopoulos, J., M. Hebblewhite, G. Aarts, and J. Fieberg. 2011. Generalized functional responses for species distributions. *Ecology* 92:583-589

[2] Scottish Oceans Institute, School of Biology, University of St. Andrews, East Sands, St. Andrews, Fife KY16 8LB Scotland, United Kingdom

[3] Wildlife Biology Program, Department of Ecosystem and Conservation Sciences, College of Forestry and Conservation, University of Montana, Missoula, Montana 59812

[4] Royal Netherlands Institute for Sea Research (NIOZ), P.O. Box 59, 1790 AB Den Burg, The Netherlands

INTEGRATED POPULATION MODELING OF BLACK BEARS IN MINNESOTA: IMPLICATIONS FOR MONITORING AND MANAGEMENT[1]

John Fieberg, Kyle W. Shertzer[2], Paul B. Conn[2], Karen V. Noyce, and Dave L. Garshelis.

ABSTRACT

Background: Wildlife populations are difficult to monitor directly because of costs and logistical challenges associated with collecting informative abundance data from live animals. By contrast, data on harvested individuals (e.g., age and sex) are often readily available. Increasingly, integrated population models are used for natural resource management, because they synthesize various relevant data into a single analysis.

Methodology/Principal Findings: We investigated the performance of integrated population models applied to black bears (*Ursus americanus*) in Minnesota, USA. Models were constructed using sex-specific age-at-harvest matrices (1980−2008), data on hunting effort and natural food supplies (which affects hunting success), and statewide mark–recapture estimates of abundance (1991, 1997, 2002). We compared this approach to Downing reconstruction, a commonly used population monitoring method that utilizes only age-at-harvest data. We first conducted a large-scale simulation study, in which our integrated models provided more accurate estimates of population trends than did Downing reconstruction. Estimates of trends were robust to various forms of model mis-specification, including incorrectly specified cub and yearling survival parameters, age-related reporting biases in harvest data, and unmodeled temporal variability in survival and harvest rates. When applied to actual data on Minnesota black bears, the model predicted that harvest rates were negatively correlated with food availability and positively correlated with hunting effort, consistent with independent telemetry data. With no direct data on fertility, the model also correctly predicted 2-point cycles in cub production. Model-derived estimates of abundance for the most recent years provided a reasonable match to an empirical population estimate obtained after modeling efforts were completed.

Conclusions/Significance: Integrated population modeling provided a reasonable framework for synthesizing age-at-harvest data, periodic large-scale abundance estimates, and measured covariates thought to affect harvest rates of black bears in Minnesota. Collection and analysis of these data appear to form the basis of a robust and viable population monitoring program.

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[1] Abstract from published paper: Fieberg, J., K.W. Shertzer, P. B. Conn, K. V. Noyce, and D. L. Garshelis. 2010. Integrated population modeling of black bears in Minnesota: implications for monitoring and management. *Plos One* 5(8): e12114. Doi:10.1371/journal.pone.0012114.

[2] Southeast Fisheries Science Center, National Marine Fisheries Service, Beaufort, North Carolina

CORRELATION AND STUDIES OF HABITAT SELECTION: PROBLEM, RED HERRING OR OPPORTUNITY?[1]

John Fieberg, Jason Matthiopoulos[2], Mark Hebblewhite[3], Mark S. Boyce[4] and Jacqueline L. Frair[5]

ABSTRACT

With the advent of new technologies, animal locations are being collected at ever finer spatiotemporal scales. We review analytical methods for dealing with correlated data in the context of resource selection, including post hoc variance inflation techniques, 'two-stage' approaches based on models fit to each individual, generalized estimating equations and hierarchical mixed-effects models. These methods are applicable to a wide range of correlated data problems, but can be difficult to apply and remain especially challenging for use–availability sampling designs, because the correlation structure for combinations of used and available points are not likely to follow common parametric forms. We also review emerging approaches to studying habitat selection that use fine-scale temporal data to arrive at biologically based definitions of available habitat, while naturally accounting for autocorrelation by modeling animal movement between telemetry locations. Sophisticated analyses that explicitly model correlation rather than consider it a nuisance, like mixed effects and state-space models, offer potentially novel insights into the process of resource selection, but additional work is needed to make them more generally applicable to large data sets based on the use–availability designs. Until then, variance inflation techniques and 2-stage approaches should offer pragmatic and flexible approaches to modeling correlated data.

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[1] Abstract from published paper: Fieberg, J., J. Matthiopoulos, M. Hebblewhite, M. S. Boyce, J. L. Frair. 2010. Correlation and studies of habitat selection: problem, red herring, or opportunity? *Philosophical Transactions of the Royal Society*, Series B 365:2233-2244

[2] NERC Sea Mammal Research Unit, Scottish Oceans Institute, University of St Andrews, St Andrews, Fife KY16 8LB, United Kingdom

[3] Wildlife Biology Program, College of Forestry and Conservation, University of Montana, Missoula, Montana 59812

[4] Department of Biological Sciences, University of Alberta, Edmonton, Alberta, Canada T6G 2E9

[5] State University of New York, College of Environmental Science and Forestry, Syracuse, New York 13210

RESOLVING ISSUES OF IMPRECISE AND HABITAT-BIASED LOCATIONS IN ECOLOGICAL ANALYSES USING GPS TELEMETRY DATA[1]

Jacqueline L. Frair[2], John Fieberg, Mark Hebblewhite[3], Francesca Cagnacci[4], Nicholas J. DeCesare[3], and Luca Pedrotti[5],

ABSTRACT

Global positioning system (GPS) technologies collect unprecedented volumes of animal location data, providing ever greater insight into animal behaviour. Despite a certain degree of inherent imprecision and bias in GPS locations, little synthesis regarding the predominant causes of these errors, their implications for ecological analysis or solutions exists. Terrestrial deployments report 37 per cent or less non-random data loss and location precision 30 m or less on average, with canopy closure having the predominant effect, and animal behaviour interacting with local habitat conditions to affect errors in unpredictable ways. Home range estimates appear generally robust to contemporary levels of location imprecision and bias, whereas movement paths and inferences of habitat selection may readily become misleading. There is a critical need for greater understanding of the additive or compounding effects of location imprecision, fix-rate bias, and, in the case of resource selection, map error on ecological insights. Technological advances will help, but at present, analysts have a suite of *ad hoc* statistical corrections and modeling approaches available—tools that vary greatly in analytical complexity and utility. The success of these solutions depends critically on understanding the error-inducing mechanisms, and the biggest gap in our current understanding involves species-specific behavioural effects on GPS performance.

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[1] Abstract from published paper: Frair, J. L., J. Fieberg, M. Hebblewhite, F. Cagnacci, N. DeCesare, and L Pedrotti. 2010. Resolving issues of imprecise and habitat-biased locations in ecological analyses using GPS telemetry data. *Philosophical Transactions of the Royal Society*, Series B 365:2187-2200.

[2] State University of New York, College of Environmental Science and Forestry, Syracuse, New York 13210

[3] Wildlife Biology Program, College of Forestry and Conservation, University of Montana, Missoula, Montana 59812

[4] Research and Innovation Centre, Environment and Natural Resources Area, Edmund Mach Foundation, 38010 San Michele all'Adige, Trento, Italy

[5] Stelvio National Park, 23032 Bormio, Sondrio, Italy

THE HOME-RANGE CONCEPT: ARE TRADITIONAL ESTIMATORS STILL RELEVANT WITH MODERN TELEMETRY TECHNOLOGY?[1]

John G. Kie[2], Jason Matthiopoulos[3], John Fieberg, Roger A. Powell[4], Francesca Cagnacci[5], Michael S. Mitchell[6], Jean-Michel Gaillard[7], and Paul R. Moorcroft[8]

ABSTRACT

Recent advances in animal tracking and telemetry technology have allowed the collection of location data at an ever-increasing rate and accuracy, and these advances have been accompanied by the development of new methods of data analysis for portraying space use, home ranges and utilization distributions. New statistical approaches include data-intensive techniques such as kriging and nonlinear generalized regression models for habitat use. In addition, mechanistic home range models, derived from models of animal movement behaviour, promise to offer new insights into how home ranges emerge as the result of specific patterns of movements by individuals in response to their environment. Traditional methods, such as kernel density estimators are likely to remain popular, because of their ease of use. Large data sets make it possible to apply these methods over relatively short periods of time, such as weeks or months, and these estimates may be analyzed using mixed-effects models, offering another approach to studying temporal variation in space-use patterns. Although new technologies open new avenues in ecological research, our knowledge of why animals use space in the ways we observe will only advance by researchers using these new technologies and asking new and innovative questions about the empirical patterns they observe.

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[1] Abstract from published paper: Kie, J. G., J. Matthiopoulos, J. Fieberg, M. S. Mitchell, R. A. Powell, F. Cagnacci, J-M. Gaillard, and P. Moorcroft. 2010. The home-range concept: are traditional estimators still relevant with modern telemetry technology? Philosophical *Transactions of the Royal Society*, Series B 365:2221-2231

[2] Department of Biological Sciences, Idaho State University, 921 South 8th Avenue, Stop 8007, Pocatello, Idaho 83209

[3] NERC Sea Mammal Research Unit, Scottish Oceans Institute, University of St. Andrews, St. Andrews, Fife KY16 8LB, United Kingdom

[4] Department of Zoology, North Carolina State University, Raleigh, North Carolina 27695

[5] Research and Innovation Centre, Environment and Natural Resources Area, Edmund Mach Foundation, 38010 San Michele all'Adige, Trento, Italy

[6] Montana Cooperative Wildlife Research Unit, University of Montana, 205 Natural Sciences Building, Missoula, Montana 59812

[7] UMR CNRS 5558 – LBBE, Biométrie et Biologie Évolutive, UCB Lyon 1 - Bât. Grégor Mendel, 43 bd du 11 novembre 1918, 69622 VILLEURBANNE cedex, France

[8] Department of Organismic and Evolutionary Biology, Harvard University, Cambridge, Massachusetts 02138

DNRSURVEY – MOVING-MAP SOFTWARE FOR AERIAL SURVEYS

Robert G. Wright, Brian S. Haroldson, and Chris Pouliot

SUMMARY OF FINDINGS

Advances in Global Position System (GPS), Geographic Information System (GIS), and computer technologies have enhanced our ability to navigate aerial wildlife surveys and capture observational data. We combined these technologies into a moving-map, aerial survey software program herein referred to as DNRSurvey, which allows users to display and record their position over digital aerial photos, navigate without reliance on ground features, and record animal locations directly to Environmental Systems Research Institute (ESRI, Inc., Redlands, California) shapefiles and Windows (Microsoft Corporation, Redmond, Washington) audio files. This program has improved the precision and efficiency of our aerial surveys and reduced data-entry transcription time and errors. Although originally designed for an aerial platform, DNRSurvey is equally applicable for vehicle-based observations.

INTRODUCTION

Aerial surveys are commonly used to estimate abundance of waterfowl, ungulates, and other large mammals. Navigation during these surveys, which began with map and compass, has improved with developments in technology, transitioning through land-based radio transmitters (e.g., long range navigation [LORAN]; Boer et al. 1989, Leptich et al. 1994) to global, satellite-based systems (e.g., GPS; Bobbe 1992, Leptich et al. 1994). Anthony and Stehn (1994) created a software program (GPSTRACK) which displayed real-time aircraft positions over pre-defined transects on a laptop computer and recorded locations of wildlife observations along transect lines. Within the last decade, advances in GPS, GIS, and computer hardware technologies have greatly enhanced our ability to navigate aerial surveys and capture observational data, independent of aircraft location. We combined these technologies into a moving-map, aerial survey software program referred to as DNRSurvey. Using a tablet computer connected to a GPS receiver, we are able to view and record our real-time position over digital air photos, navigate without reliance on ground features, and record animal observation data (e.g., location, count, age/sex, cover type) directly to ESRI shapefiles and Windows audio files. DNRSurvey is not a GIS, but a data collection tool that incorporates relevant GIS functionality. Use of this program has improved the precision and efficiency of our aerial surveys and reduced data-entry transcription time and errors.

SOFTWARE DEVELOPMENT

DNRSurvey was developed in Visual Basic (VB; Microsoft Corporation, Redmond, Washington) programming language and consists of 2 integrated components - Survey Editor (VB.NET) and MapView (VB 6.0). With Survey Editor, users create survey-specific data entry forms and shapefile attribute tables to record wildlife observations or other objects of interest (Figure 1). A variety of input controls, including textboxes, checkboxes, radio buttons, comboboxes, listboxes, and voice recording are available to customize data input (Figures 1 and 2). A spatial join feature allows attributes (e.g., public land survey features, plot number, acres) from another shapefile to be written to the survey shapefile when observations are recorded. Survey shapefile properties such as symbology and labeling can be pre-defined and a custom icon can be assigned to each survey form tool button (Figure 3).

MapView is the survey component of DNRSurvey and emulates a stripped-down version of an ESRI ArcMap data frame (Figure 4). It communicates with a GPS receiver via serial, USB or Bluetooth connection; displays a bread-crumb trail of positional fixes; and pans the display window as needed. Background image and shapefile layers such as aerial photos, management unit boundaries, and survey plot boundaries are managed in a Table of Contents window (Figure 4). Shapefile symbology and labeling can be customized and scale-dependent displays can be defined for all layers (Figure 5). Key functionalities, such as preset zoom scales, data backup, editing, and survey form activation are presented as toolbar buttons (Figure 4). Customized settings can be saved as a unique survey file (e.g., pa272_survey.lvs).

To begin collecting observations, users open the customized survey file, connect to the GPS receiver using the toolbar button, and select the data form tool button to make it active. The user records an observation by touching the screen where the object of interest is located and by completing the pop-up data form (Figure 4). Location coordinates and data form values are written directly to an output shapefile or audio file. Observations can be captured anywhere on the display or by accepting the current GPS position. The user edits an observation by selecting the Edit button and desired on-screen data point, and then by correcting erroneous data values in the pop-up data form. Pressing the data backup button copies all survey related data (e.g., observation shapefile, flight line shapefile, flight line text file) to a date/time-stamped working directory. The GPS coordinate properties (i.e., datum, coordinate system) are user-defined, but default to the North American Datum of 1983 (NAD83) and Universal Transverse Mercator (UTM) Zone 15N, respectively. In addition, the aircraft flight-line display and recording properties can be customized to meet the user's needs (Figure 6).

DNRSurvey works on tablet computers running Windows XP and Windows 7 operating systems (Microsoft Corporation, Redmond, Washington). We recommend a minimum computer configuration which includes: 80 GB hard drive; 3 GB RAM; 1 GHz processor; 550 nit daylight-readable display; serial port and/or Bluetooth data link; and integrated keyboard. DNRSurvey is compatible with GPS receivers using Garmin (Garmin International, Inc., Olathe, Kansas) or National Marine Electronics Association (NMEA) output formats.

For cockpit deployment, we currently use a wireless configuration consisting of a Panasonic CF-19 Toughbook tablet computer (Panasonic Corporation, Secaucus, New Jersey) communicating with an fTech Solarius BT-25 SR Solar Bluetooth GPS receiver (fTech Corporation, Tainan, Taiwan). This configuration is convenient and enhances cockpit safety by eliminating loose cables. A Garmin GPSMAP196 mounted in the aircraft serves as a backup receiver. The computer battery lasts >3 hours and is replaced during each fuel stop. Battery life for the solar Bluetooth GPS is sufficient to last all day on a single charge.

We are currently working on additional enhancements and expect to complete software development by December 2011. Although originally designed for an aerial platform, DNRSurvey is equally applicable for vehicle-based observations and will be available at www.dnr.state.mn.us.

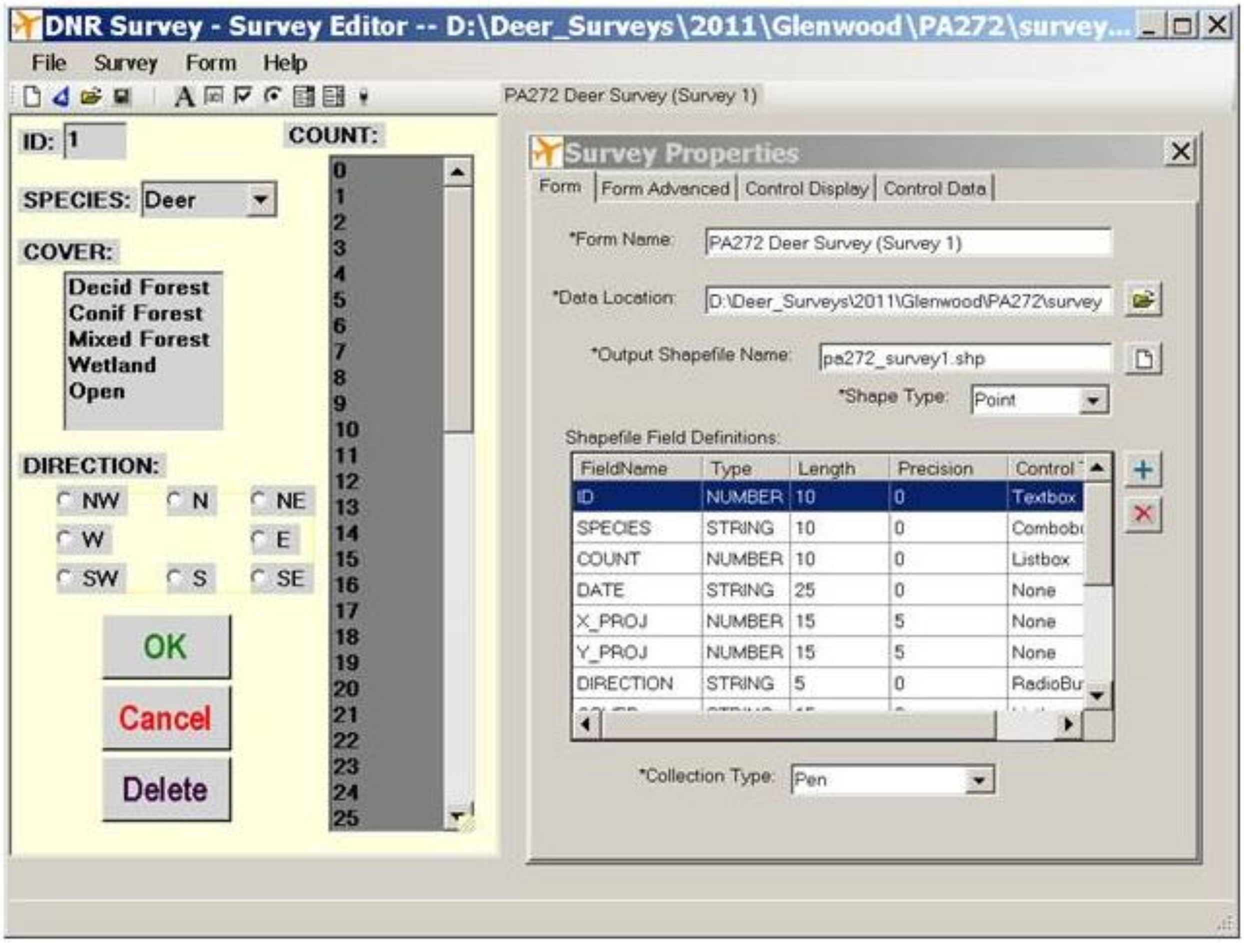
LITERATURE CITED

Anthony, R. M., and R. A. Stehn. 1994. Navigating aerial transects with a laptop computer map. Wildlife Society Bulletin 22:674-676.

Bobbe, T. 1992. Real-time differential GPS for aerial surveying and remote sensing. GPS World 3:18-22.

Boer, A. H., G. Redmond, and T. J. Pettigrew. 1989. LORAN-C: a navigation aid for aerial surveys. Journal of Wildlife Management 53:228-230.

Leptich, D. J., D. G. Beck, and D. E. Beaver. 1994. Aircraft-based LORAN-C and GPS accuracy for wildlife research on inland study sites. Wildlife Society Bulletin 22:561-565.



ID: 1

SPECIES: Deer

COVER:
Decid Forest
Conif Forest
Mixed Forest
Wetland
Open

*Form Name: PA272 Deer Survey (Survey 1)

*Data Location: D:\Deer_Surveys\2011\Glenwood\PA272\survey

*Output Shapefile Name: pa272_survey1.shp

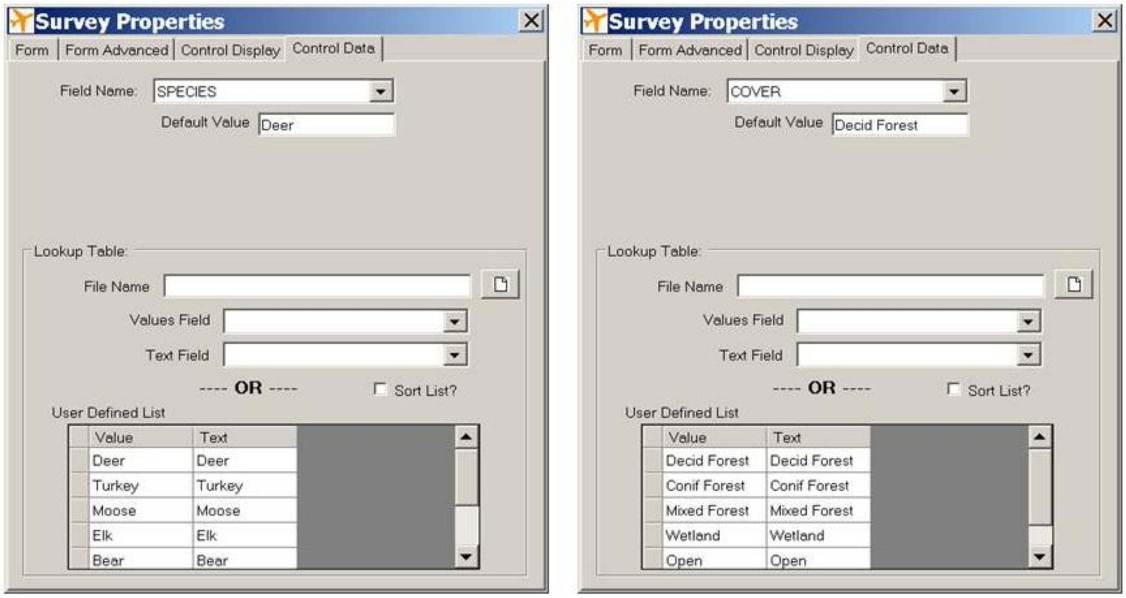
*Shape Type: Point

Shapefile Field Definitions:

FieldName	Type	Length	Precision	Control
ID	NUMBER	10	0	Textbox
SPECIES	STRING	10	0	Combob
COUNT	NUMBER	10	0	Listbox
DATE	STRING	25	0	None
X_PROJ	NUMBER	15	5	None
Y_PROJ	NUMBER	15	5	None
DIRECTION	STRING	5	0	RadioBu

*Collection Type: Pen

Figure 1. Survey Editor form building interface of DNRSurvey.



Field Name: SPECIES

Default Value: Deer

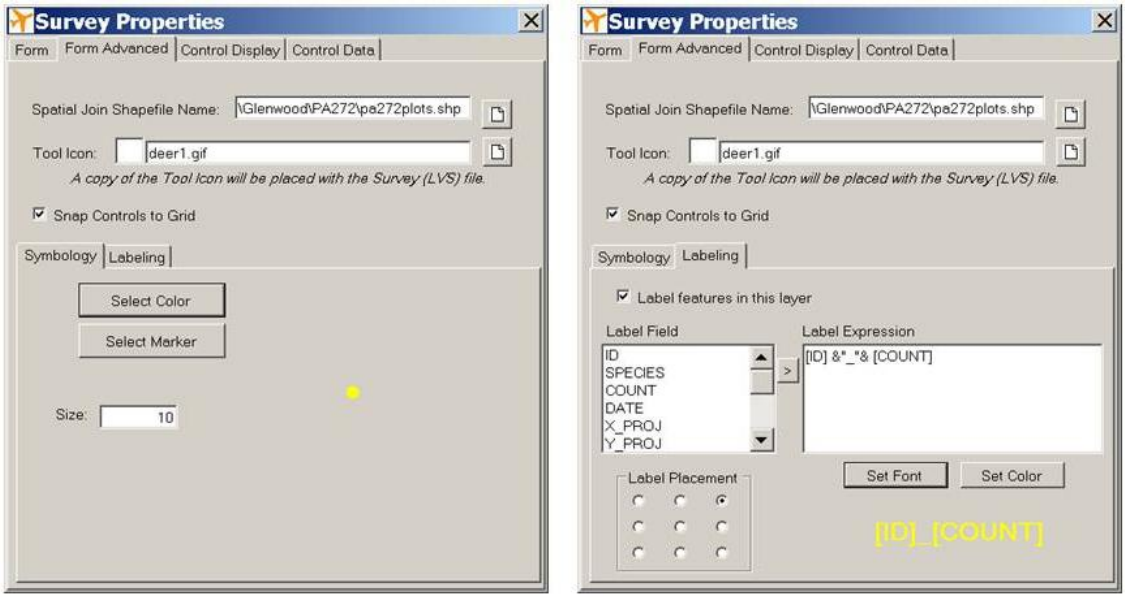
Value	Text
Deer	Deer
Turkey	Turkey
Moose	Moose
Elk	Elk
Bear	Bear

Field Name: COVER

Default Value: Decid Forest

Value	Text
Decid Forest	Decid Forest
Conif Forest	Conif Forest
Mixed Forest	Mixed Forest
Wetland	Wetland
Open	Open

Figure 2. Data input values for combobox (species) and listbox (cover) controls are defined using lookup or user-defined tables via drop-down menus and tabs within the Survey Editor component of DNRSurvey.



Spatial Join Shapefile Name: \Glenwood\PA272\pa272plots.shp

Tool Icon: deer1.gif

A copy of the Tool Icon will be placed with the Survey (LVS) file.

☑ Snap Controls to Grid

Size: 10

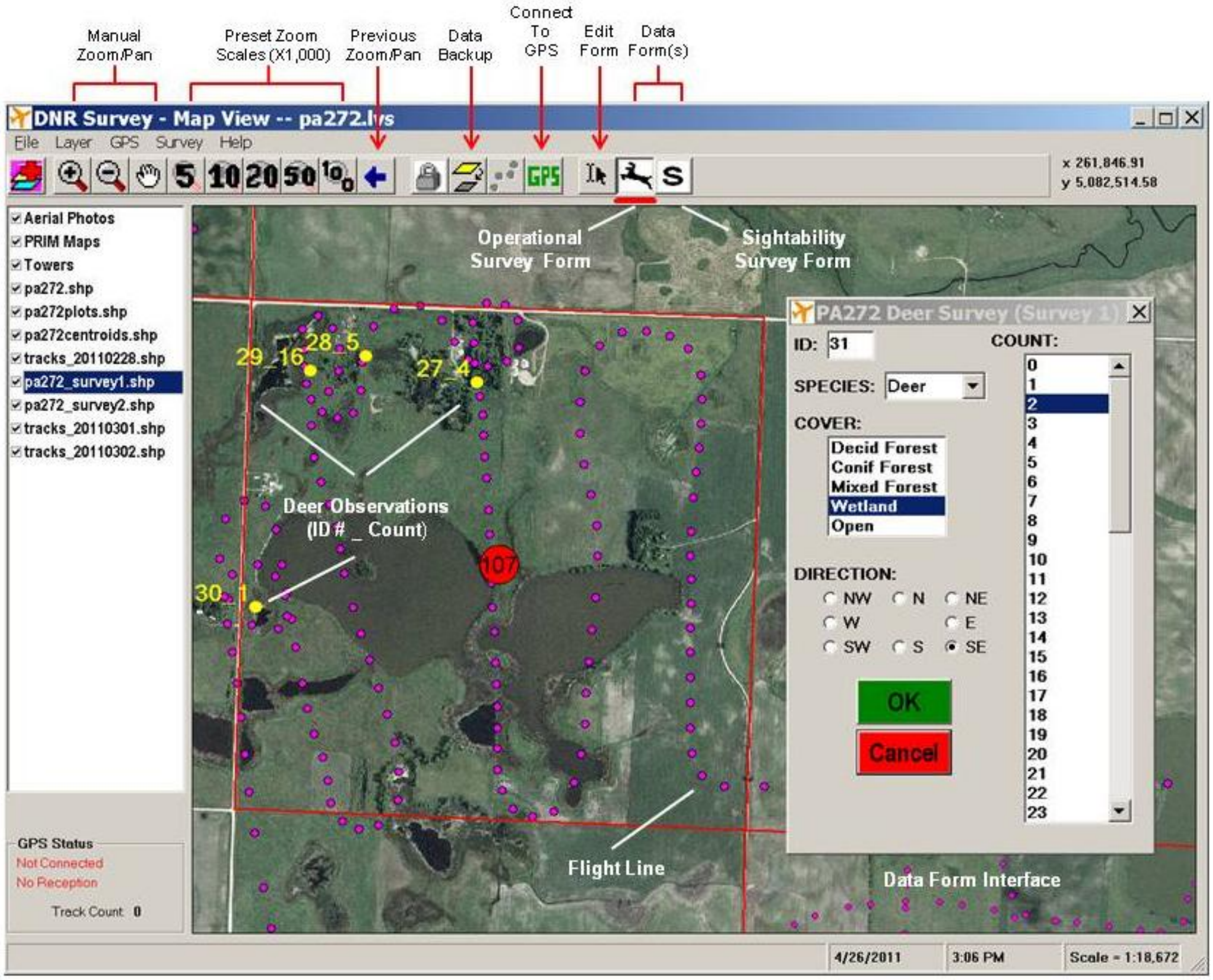
☑ Label features in this layer

Label Field: ID, SPECIES, COUNT, DATE, X_PROJ, Y_PROJ

Label Expression: [ID] &"_"& [COUNT]

[ID]_[COUNT]

Figure 3. Spatial join shapefiles, tool icons, and symbology and labeling properties are defined via drop-down menus and tabs within the Survey Editor component of DNRSurvey.



ID: 31

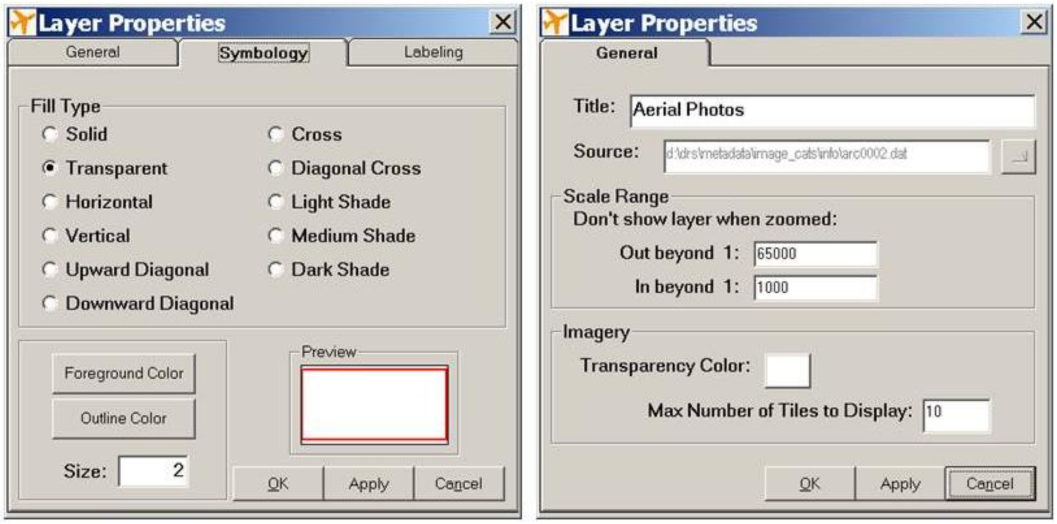
SPECIES: Deer

COVER: Decid Forest, Conif Forest, Mixed Forest, **Wetland**, Open

DIRECTION: ○ NW ○ N ○ NE ○ W ○ E ○ SW ○ S ● SE

COUNT: **2**

Figure 4. MapView interface component of DNRSurvey.



● Transparent

Size: 2

Title: Aerial Photos

Source: d:\drs\metadata\image_cats\info\arc0002.dat

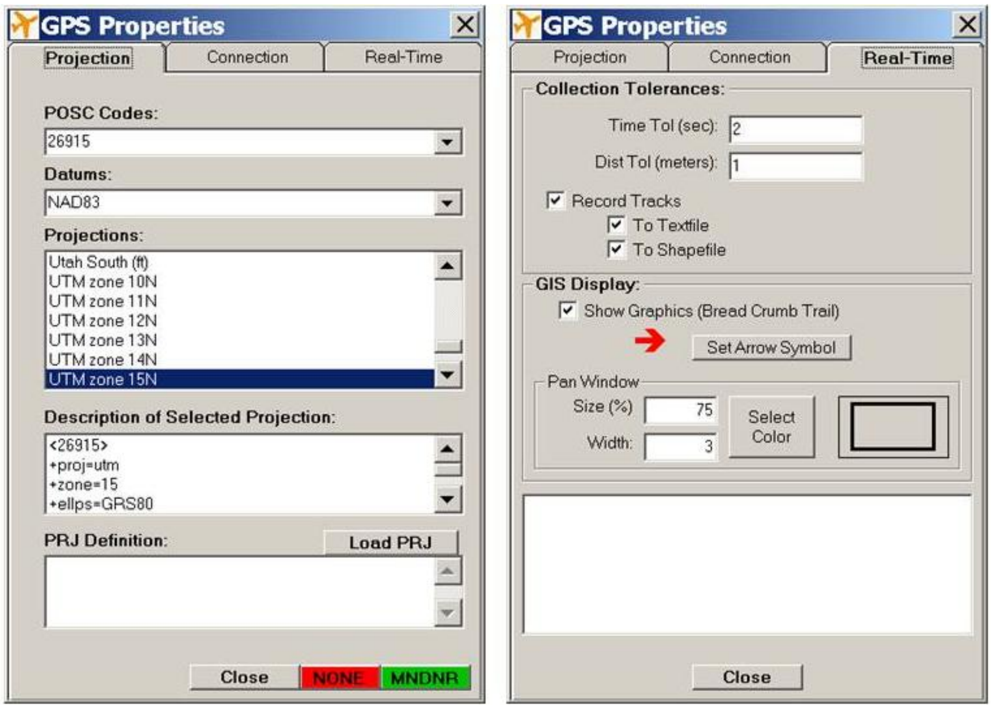
Don't show layer when zoomed:

Out beyond 1: 65000

In beyond 1: 1000

Max Number of Tiles to Display: 10

Figure 5. Symbology, labeling, and scale properties of background layers are defined via drop-down menus and tabs within the MapView component of DNRSurvey.



POSC Codes: 26915

Datums: NAD83

Projections: UTM zone 15N

Description of Selected Projection:
<26915>
+proj=utm
+zone=15
+ellps=GRS80

Time Tol (sec): 2

Dist Tol (meters): 1

☑ Record Tracks
  ☑ To Textfile
  ☑ To Shapefile

☑ Show Graphics (Bread Crumb Trail)

Size (%): 75

Width: 3

Figure 6. Datum and projection values, and flight-line display and recording properties are defined via drop-down menus and tabs within the MapView component of DNRSurvey.

Publications

The following is a list of scientific reports and other publications by personnel in the Wildlife Populations and Research Unit for the approximate period of March 2010 thru February 2011.

Some titles by Unit personnel pertain to work done while employed by the MNDNR, while other titles are from work done elsewhere (e.g. as graduate student, employed by another agency, while on leave of absence, etc.)

An asterisk (*) before an author's name indicates the report was listed as in press or in review in previous publication of the Summaries of Wildlife Research Findings.

Included under scientific reports are those that have not been published and those accepted for publication (in press).

Names in **bold** indicate a MNDNR employee.

Wetland Wildlife Populations and Research Group

Publication List (2010-2011)

Hanson, M. A., B. R. Herwig, K. D. Zimmer, **J. Fieberg**, S. R. Vaughn, R. G. Wright, and J. A. Younk. 2011. Factors influencing aquatic invertebrate populations in shallow lakes in prairie and parkland regions of Minnesota, USA. Wetlands: *In Review*.

Herwig, B. R., K. D. Zimmer, **M. A. Hanson**, M. L. Konsti, J. A. Younk, **R. W. Wright**, S. R. Vaughn, and M. E. Haustein. 2010. Factors influencing fish distributions in shallow lakes in prarie and prairie-parkland regions of Minnesota, USA. Wetlands 30:609-619.

Friederichs, S. J., K. D. Zimmer, B. R. Herwig, **M. A. Hanson**, and **J. Fieberg**. 2010. Total phosphorus and piscivore mass as drivers of food web characteristics in shallow lakes. Oikos 120:756-765.

Forest Wildlife Populations and Research Group

Publication List (2010-2011)

Baker, L. R., T. W. Arnold, O. S. Olubode, and **D. L. Garshelis**. 2011. Considerations for using occupancy surveys to monitor forest primates: a case study with Sclater's monkey (*Cercopithecus sclateri*). Population Ecology: In press.

***DelGiudice, G. D., B. A. Sampson, M. S. Lenarz,** M. W. Schrage, and A. J. Edwards. 2011. Winter body condition of moose (*Alces alces*) in a declining population in northeastern Minnesota. Journal of Wildlife Diseases 47:30-40.

Garshelis, D. L. 2011. Andean bear density and abundance estimates—how reliable and useful are they? Ursus 22:47–64.

*Hwang, M-H., **D. L. Garshelis**, Y-H. Wu, and Y. Wang. 2010. Home ranges of Asiatic black bears in the Central Mountains of Taiwan: gauging whether a reserve is big enough. Ursus 21:81–96.

*Laske, T. G., H. J. Harlow, **D. L. Garshelis**, and P. Iaizzo**.** 2010. Extreme respiratory sinus arrhythmia enables hibernating black bear survival - physiological insights and applications to human medicine. Journal of Cardiovascular Translational Research 3: 559–569.

Laske, T. G., **D. L. Garshelis**, and P. Iaizzo. 2011. Monitoring the wild black bear's reaction to human and environmental stressors. BMC Physiology: In press.

***Lenarz, M. S., J. Fieberg**., M. W. Schrage, and A. J. Edwards. 2010. Living on the edge: viability of moose in northeastern Minnesota. Journal of Wildlife Management 74:1013-1023.

Lenarz, M. S., **R. G. Wright**, M. S. Schrage, and A. J. Edwards. 2011. Compositional analysis of moose habitat in northeastern Minnesota. Alces: In review.

Liu, F., W. J. McShea, **D. L. Garshelis**, X. Zhu, D. Wang, and L. Shao. 2011. Human-wildlife conflicts influence attitudes but not necessarily behaviors: factors driving the poaching of bears in China. Biological Conservation 144:538–547.

Martin, D. J., B. R. McMillan, **J. D. Erb**, T. A. Gorman, and D. P. Walsh. 2010. Diel activity patterns of river otters (*Lutra canadensis*) in southeastern Minnesota. Journal of Mammalogy 91:1213-1224.

McGowan, C. P., M. C. Runge, and **M. A. Larson**. 2011. Incorporating parametric uncertainty into population viability analysis models. Biological Conservation 144: In press.

Noyce, K. V., and **D. L. Garshelis**. 2011. Seasonal migrations of black bears (*Ursus americanus*): causes and consequences. Behavioral Ecology and Sociobiology 65: 823-835.

Peacock, E., K. Titus, **D. L. Garshelis**, M. M. Peacock, and M. Kuc. 2011. Mark-recapture using tetracycline and genetics reveals record-high bear density. Journal of Wildlife Management: In press.

Reidy, J. L., F. R. Thompson, III, and **J. W. Bailey**. 2011. Comparison of methods for estimating density of forest songbirds from point counts. Journal of Wildlife Management 75:558-568.

*Steinmetz, R., and **D. L. Garshelis**. 2010. Estimating ages of bear claw marks in Southeast Asian tropical forests as an aid to population monitoring. Ursus 21:143–153.

Steinmetz, R., **D. L. Garshelis**, W. Chutipong, and N. Seuaturien. 2011. The shared preference niche of sympatric Asiatic black bears and sun bears in a tropical forest mosaic. PLoSONE 6(1):e14509.

*_In press_ last year.

Farmland Wildlife Populations and Research Group

Publication List (2010-2011)

Carbaugh, J. S., D. L. Combs, and **E. M. Dunton**. 2010. Nest site selection and nesting ecology of giant Canada geese in central Tennessee. Human-Wildlife Interactions 4:207-212.

Cornicelli, L., and **M. D. Grund**. 2011. Assessing deer hunter attitudes toward regulatory change using self-selected respondents. Human Dimensions of Wildlife 16:174-182.

Cornicelli, L., **M. D. Grund**, and **J. Fieberg**. 2011. Hunter perceptions and acceptance of alternative deer management regulations. Wildlife Society Bulletin: In press.

Dunton, E. M., and D. L. Combs. 2010. Movements, habitat selection, associations, and survival of giant Canada goose broods in central Tennessee. Human-Wildlife Interactions 4:192-201.

Dunton, E. M., **J. Fieberg**, and **K. J. Haroldson**. 2011. Living on the edge: diet selection and body condition of wild turkeys in northern Minnesota. Proceedings of the National Wild Turkey Symposium 10: In press.

Fieberg, J., **L. Cornicelli**, D. C. Fulton, and **M. D. Grund**. 2010. Design and analysis of simple choice surveys for natural resource management. Journal of Wildlife Management 74:871-879.

Grund, M. D., **L. Cornicelli**, L. T. Carlson, and **E. Butler**. 2010. Bullet fragmentation and lead deposition in white-tailed deer and domestic sheep. Human-Wildlife Interactions 4:257-265.

Grund, M. D. 2011. Survival analysis and computer simulations of lethal and contraceptive management strategies for urban deer. Human-Wildlife Interactions 5:23-31.

Harwood, A., B. McMillan, **K. Haroldson**, and **J. Giudice**. 2011. Probability of detection in crowing surveys of ring-necked pheasants. Journal of Wildlife Management: Submitted.

Parent, C. J., B. J. Goodwin, and **E. M. Dunton**. 2011. Survival of wild turkey hens transplanted beyond their ancestral range in northwestern Minnesota. Proceedings of the National Wild Turkey Symposium 10: In press.

Tranel, M. A. 2010. Preliminary findings of interseeding forbs in native grass stands. In D. W. Williams and D. D. Smith, editors. Proceedings of the North American Prairie Conference 22: In press.

Wildlife Biometrics Unit

Publication List (2010-2011)

Aing, C., S. Halls, K. Oken, R. Dobrow, and **J. Fieberg**. 2011. A Bayesian hierarchical occupancy model for track surveys conducted in a series of linear, spatially correlated sites. Journal of Applied Ecology: In press.

Giudice, J., **J. Fieberg**, and **M. Lenarz**. 2011. Spending degrees of freedom in a poor economy: a case study of building a sightability model for moose in northeastern Minnesota. Journal of Wildlife Management: In press.

Matthiopoulos, J., M. Hebblewhite, G. Aarts, and **J. Fieberg**. 2011. Generalized functional responses for species distributions. Ecology 92:583-589.

Fieberg, J., K. W. Shertzer, P. B. Conn, **K. V. Noyce**, and **D. L. Garshelis**. 2010. Integrated population modeling of black bears in Minnesota: implications for monitoring and management. *Plos One* 5(8): e12114. Doi:10.1371/journal.pone.0012114.

Fieberg, J., J. Matthiopoulos, M. Hebblewhite, M. S. Boyce, J. L. Frair. 2010. Correlation and studies of habitat selection: problem, red herring, or opportunity? Philosophical Transactions of the Royal Society, Series B 365:2233-2244.

Frair, J. L., **J. Fieberg**, M. Hebblewhite, F. Cagnacci, N. DeCesare, and L. Pedrotti. 2010. Resolving issues of imprecise and habitat-biased locations in ecological analyses using GPS telemetry data. Philosophical Transactions of the Royal Society, Series B 365:2187-2200.

Kie, J. G., Matthiopoulos, **J., Fieberg**, J., Mitchell, M.S., Powell, R. A., Cagnacci, F., Gaillard, J-M., and P. Moorcroft. 2010. The home-range concept: are traditional estimators still relevant with modern telemetry technology? Philosophical *Transactions of the Royal Society*, Series B 365:2221-2231.