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 $\pm$ SE, range = 7−15, n = 12 nests with 109 eggs) and $86.6 \pm 0.1\%$ 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 $89.5 \pm 0.6\%$ 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 $84.5 \pm 0.1\%$ 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.

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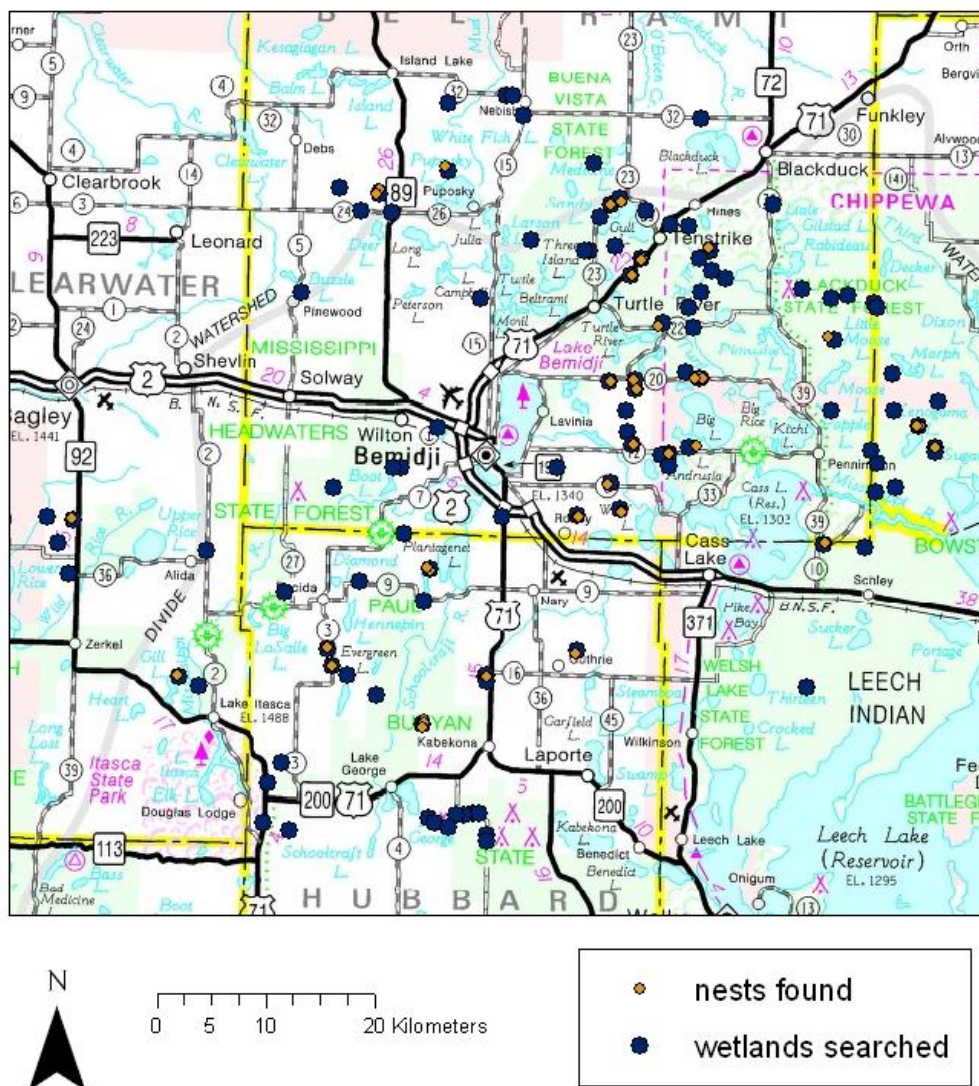
Figure 1. Wetlands searched for ring-necked duck nests in north-central Minnesota during 2008−2010.

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 $\pm$ 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.

[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.

ACKNOWLEDGMENTS

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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.

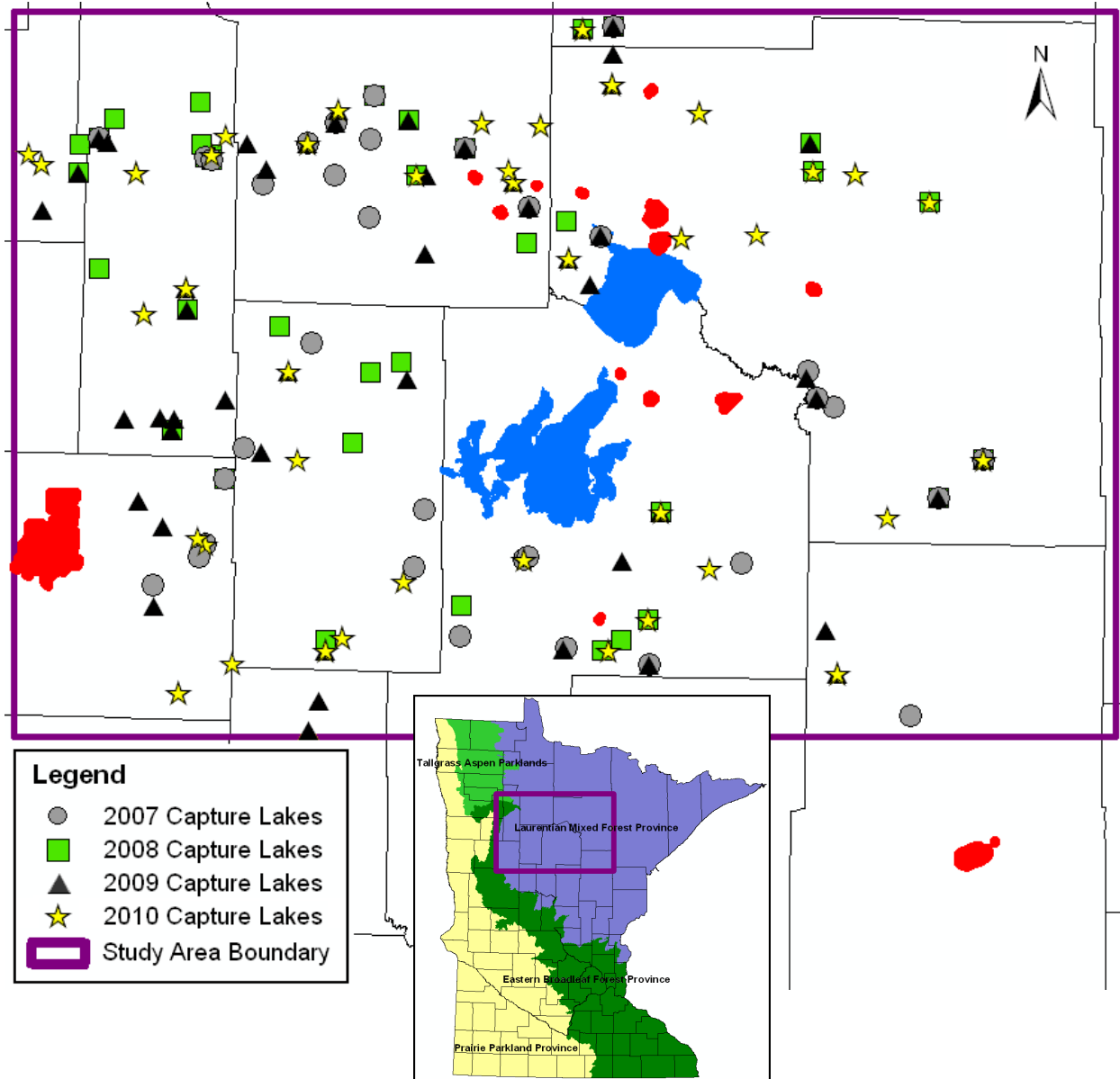| Day use	Night use	24-hour use	Not used
Donkey	Pigeon River	Mud Goose	Tamarac NWR - Chippewa
Drumbeater	Rice Pond	Round	
Fiske Blue Rocks		Tamarac NWR – Little Flat	
Gimmer			
Hatties and Jim			
Hole-in-Bog			
Lower Pigeon			
Preston Lakes			
Tamarac NWR – Flat			
Rice Lake NWR			

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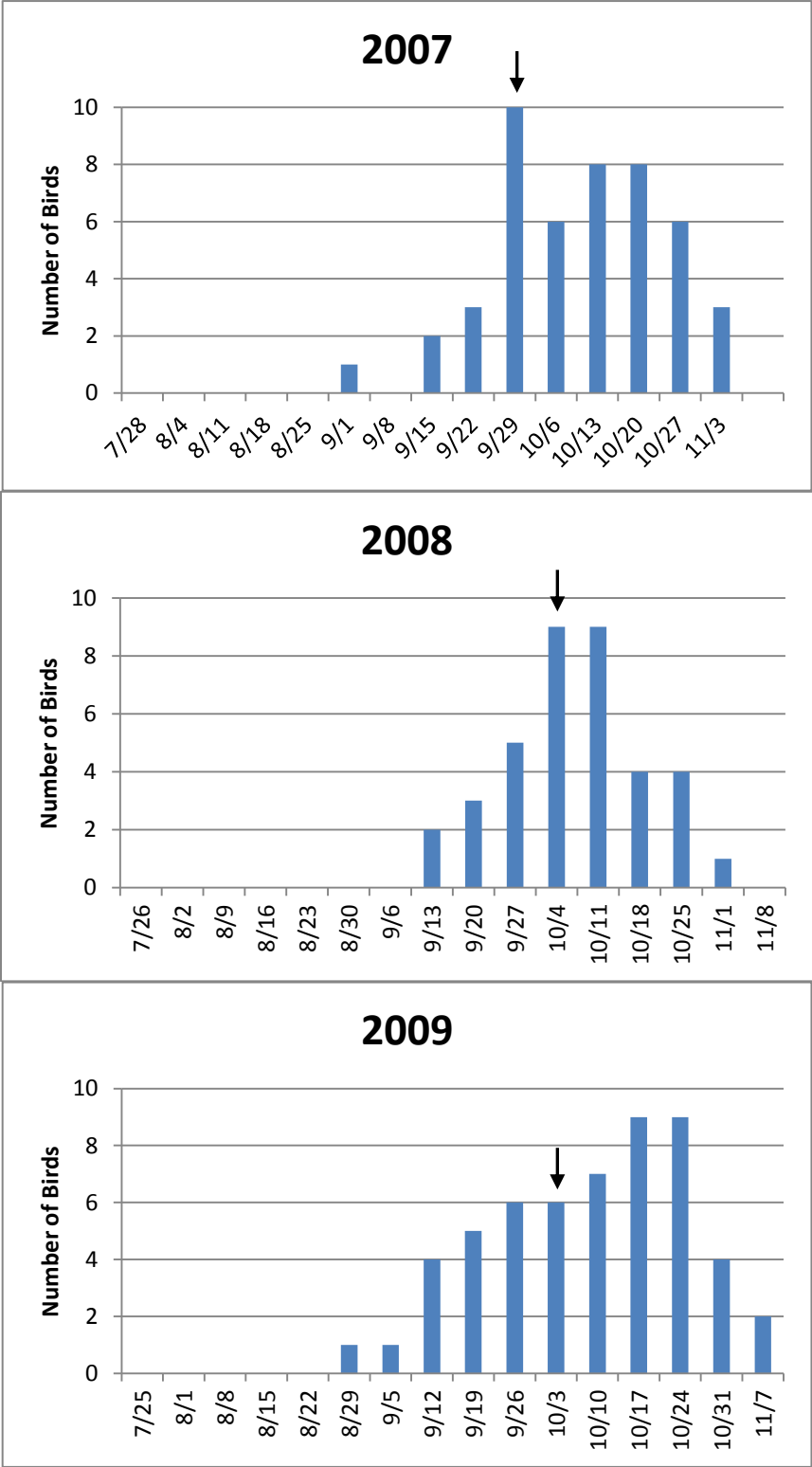
Figure 2. Weekly use of refuges by post-fledging ring-necked ducks before and during hunting season in Minnesota, 2007–2010. Weeks are from Saturday through Friday with the Saturday date shown. Arrows indicate the week waterfowl hunting opened.
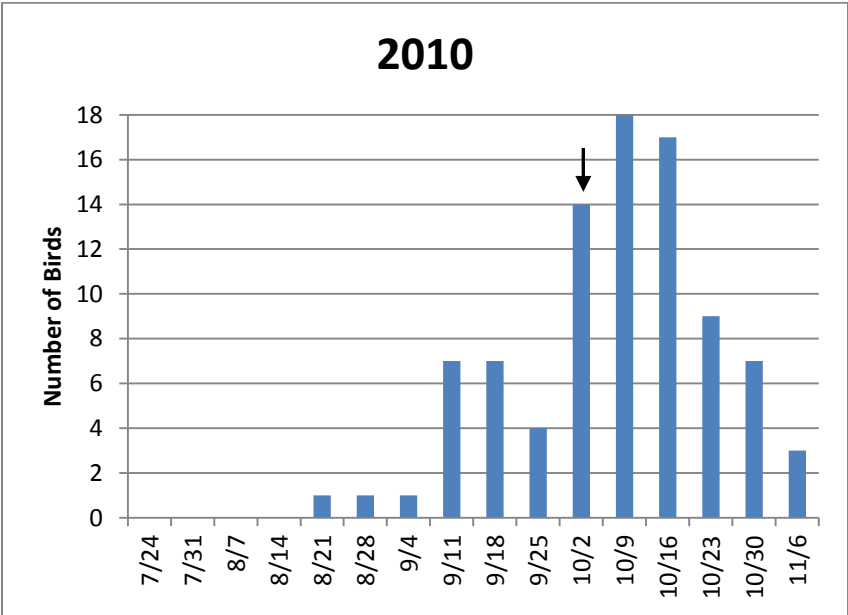
Figure 2. Continued
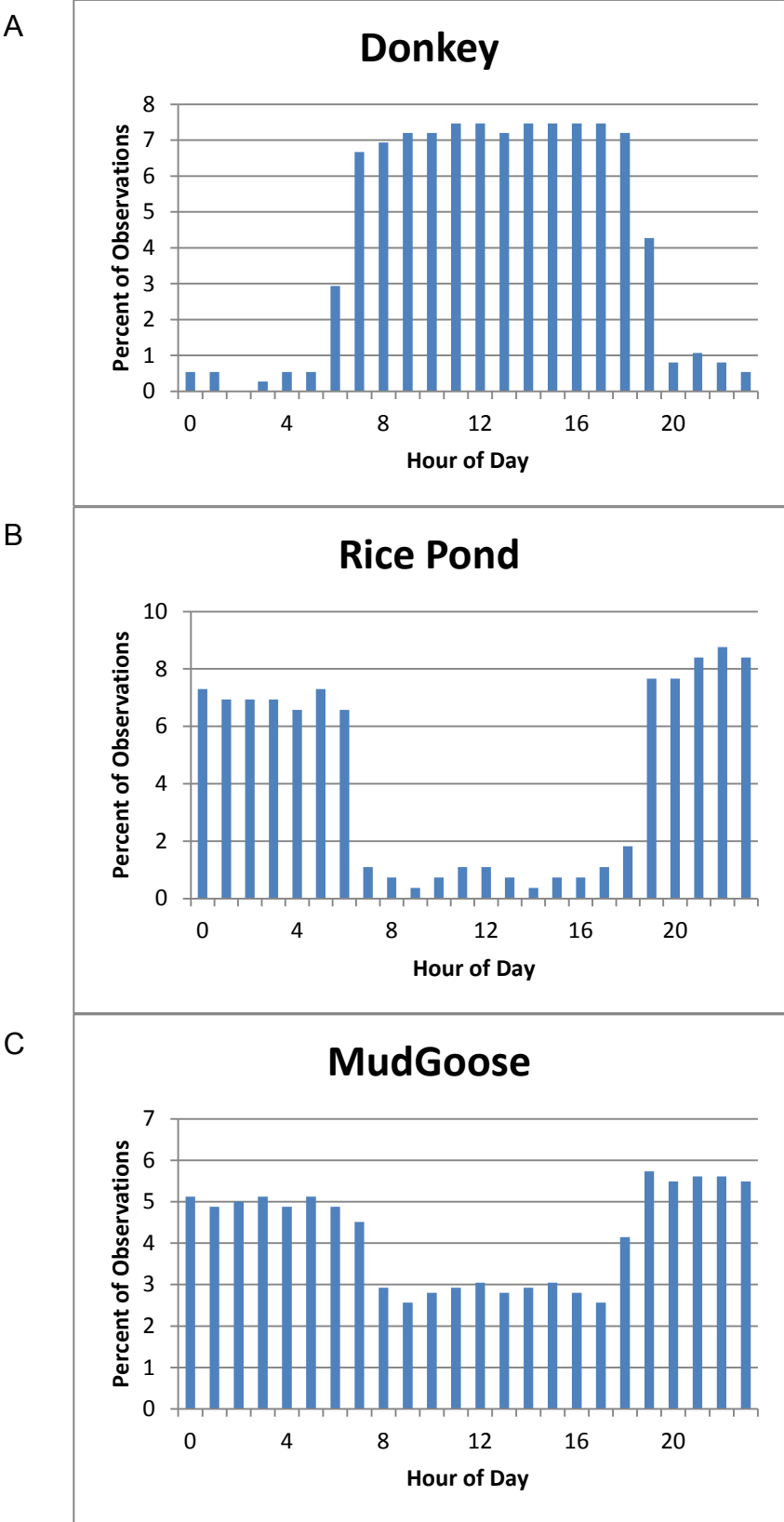
A

Donkey

B

Rice Pond

C

MudGoose

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 $\pm$ 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.

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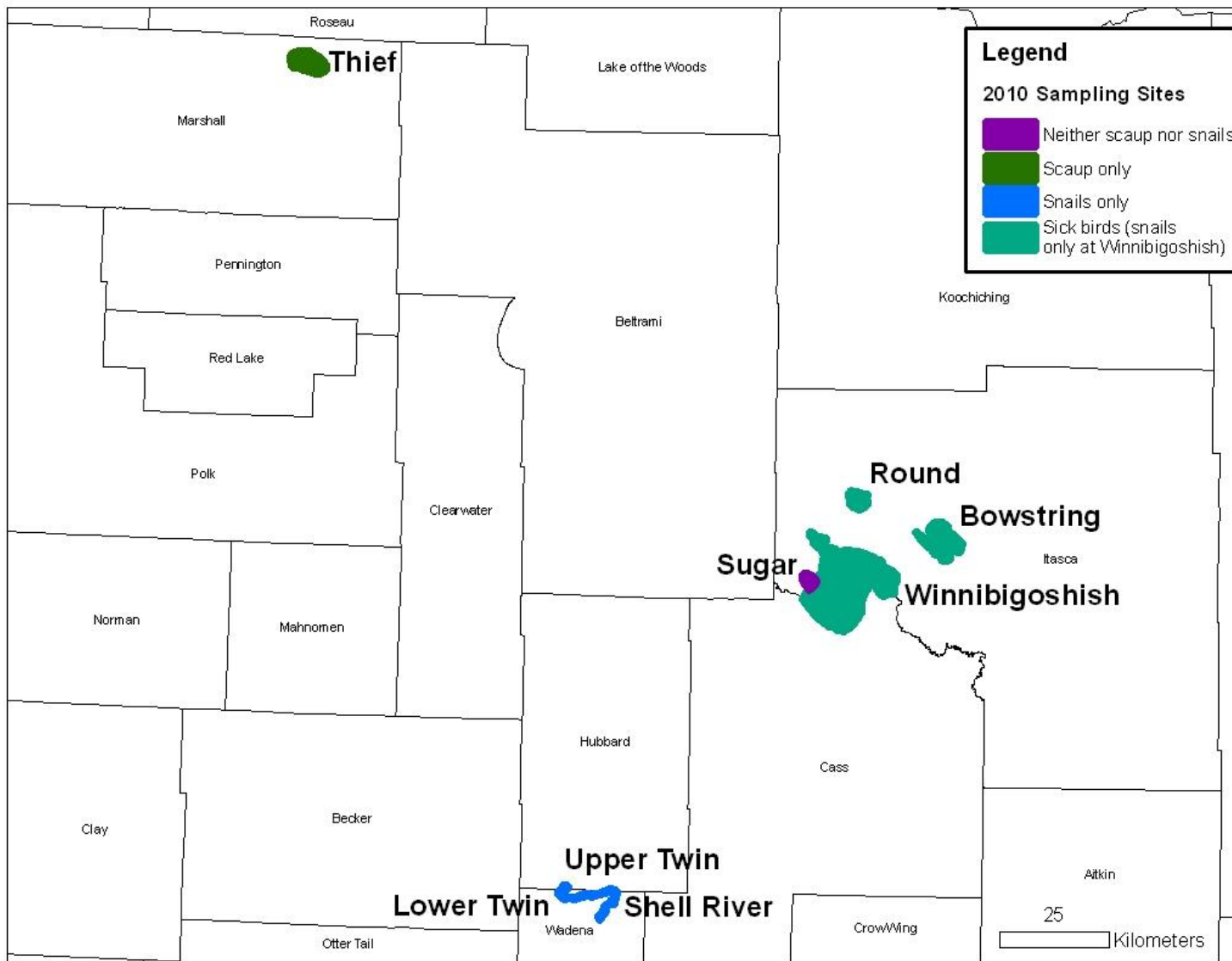
Figure 1. Lakes sampled for faucet snails and scaup presence in northern Minnesota in fall 2010.

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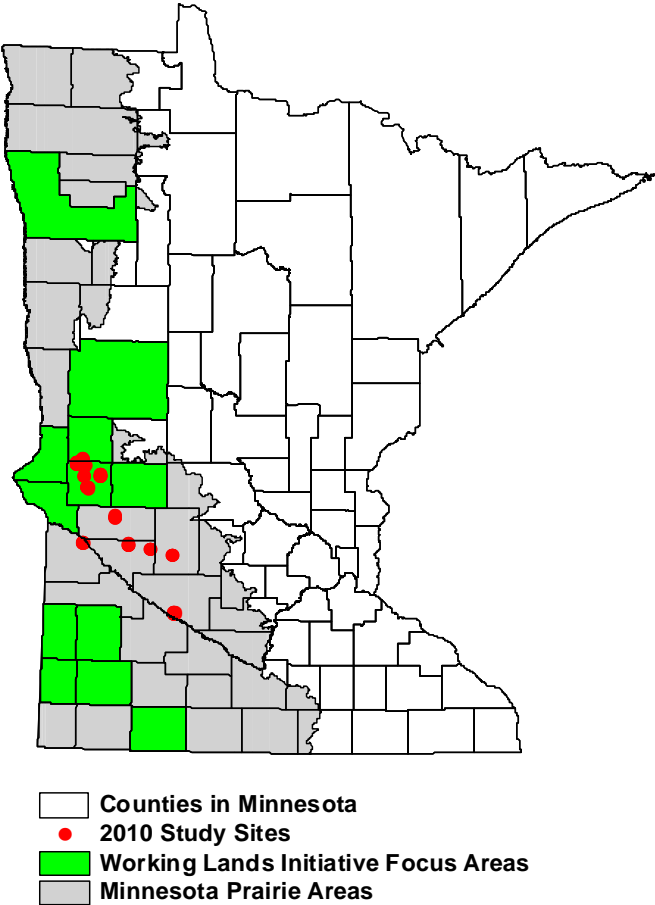
Figure 1.   Minnesota counties showing study sites, Minnesota's prairie areas, and Working Lands Initiative focus areas, 2010.

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.
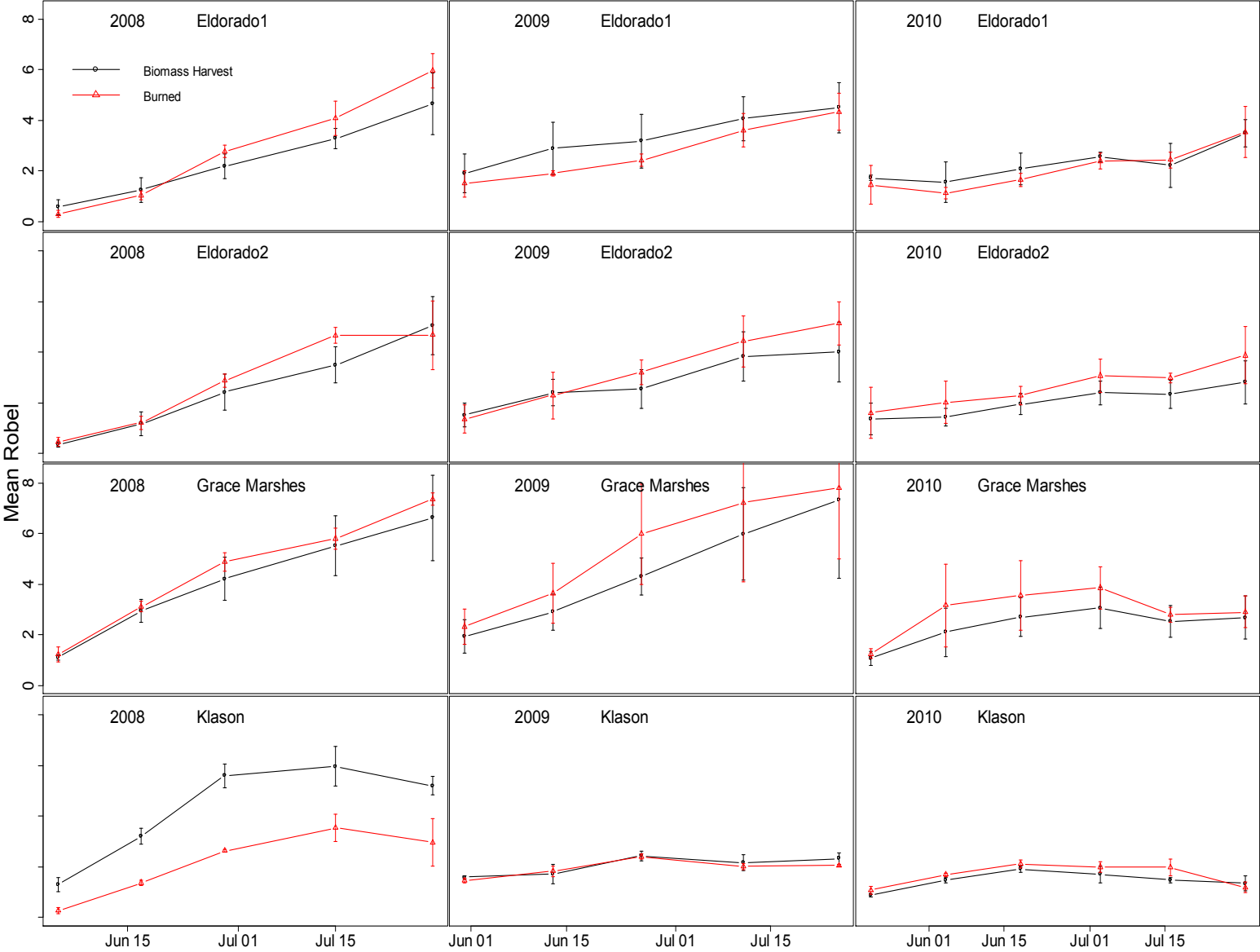
Figure 2b. Comparison of mean Robel measurements (dm) and 95% confidence intervals between 2 treatment subplots (a fall 2008 biomass harvest and a prescribed burn in spring 2009) within the same restored native grass field on 5 State Wildlife Management Areas and 1 Federal Waterfowl Production area in 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.
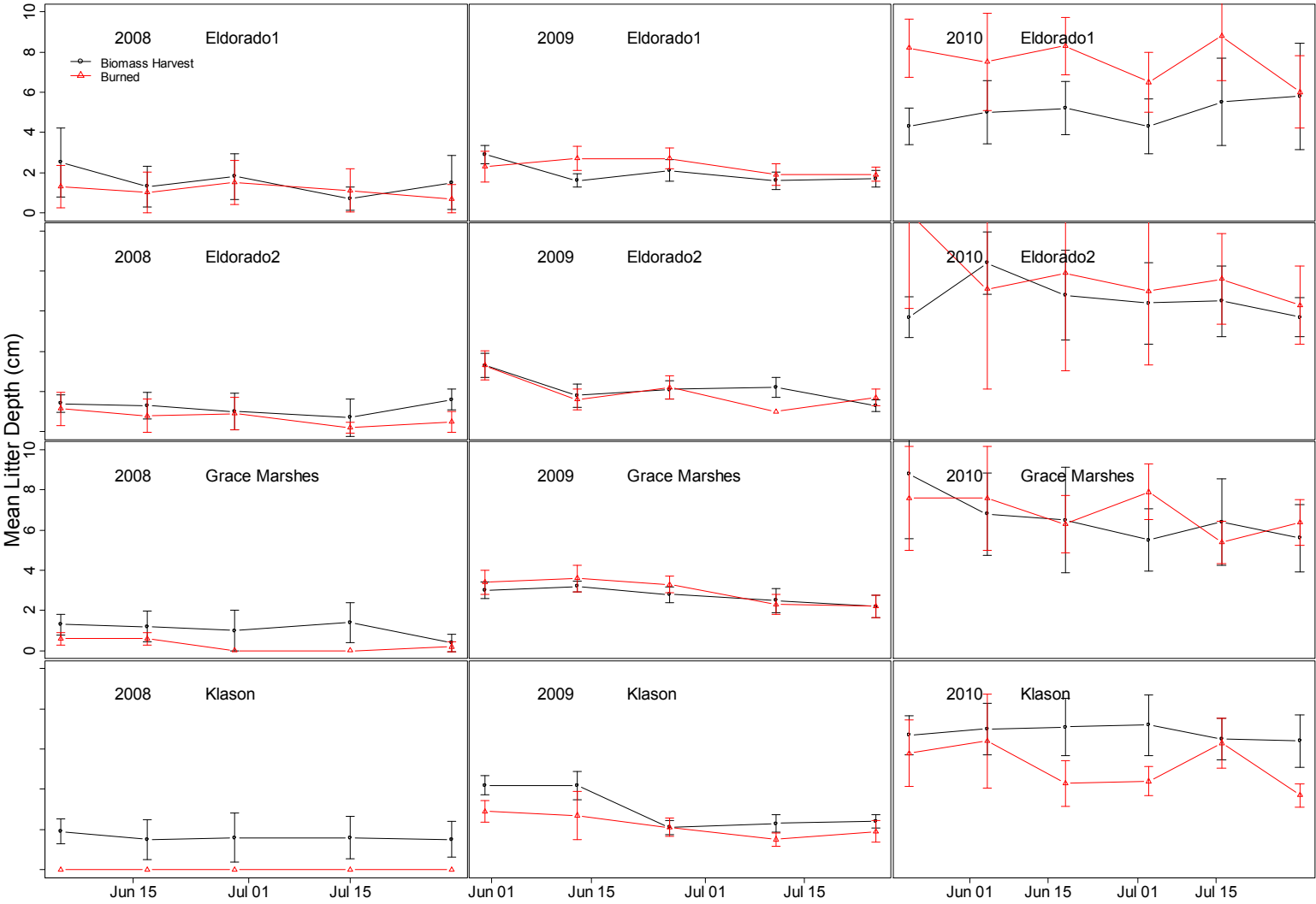
Figure 4a. Comparison of mean litter depth (dm) and 95% confidence intervals between 2 treatment subplots (a fall 2007 biomass harvest and a prescribed burn in spring 2008) within the same restored native grass field on 3 State Wildlife Management Areas (2 fields on one area) in west-central Minnesota, in summers 2008 – 2010.
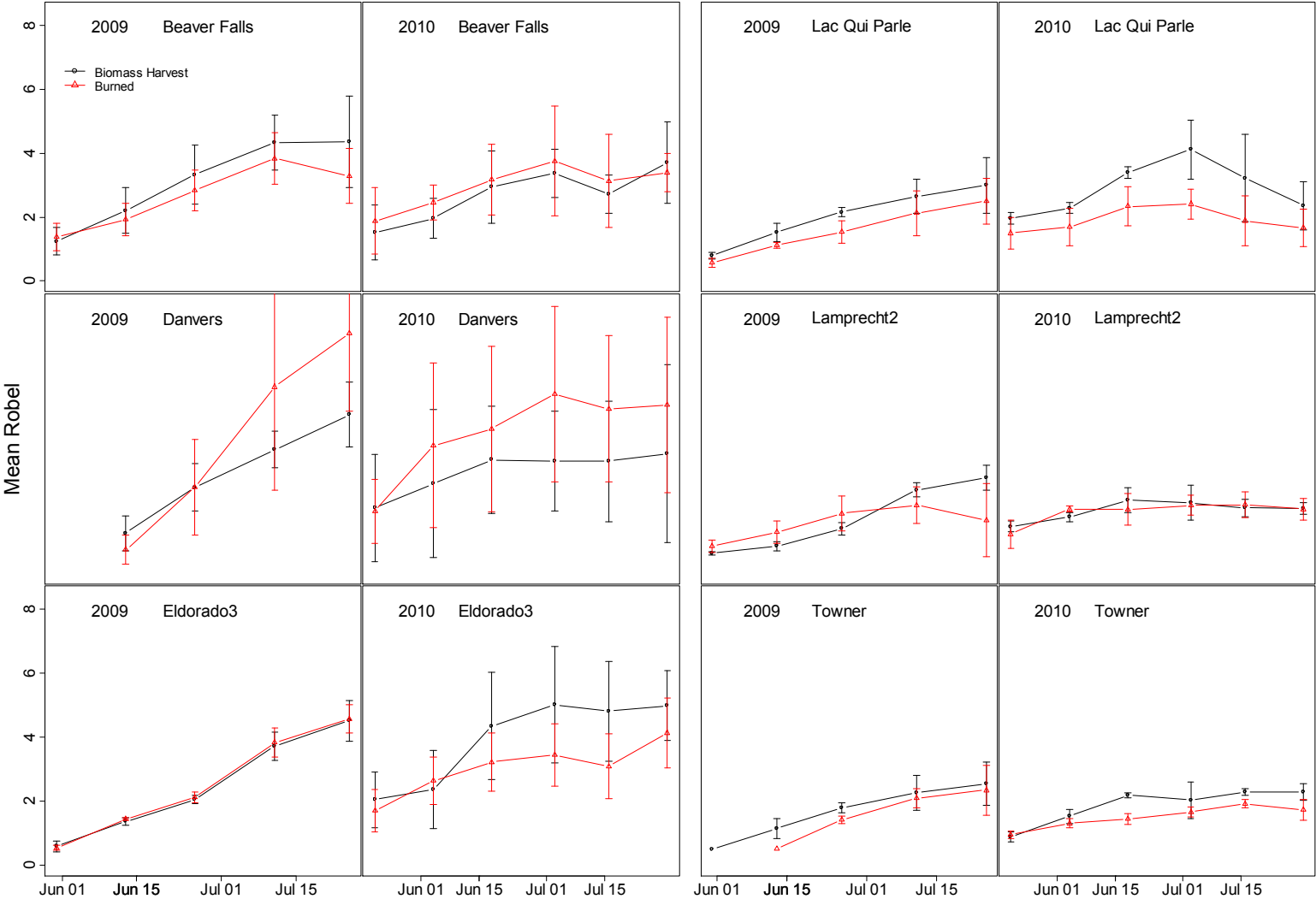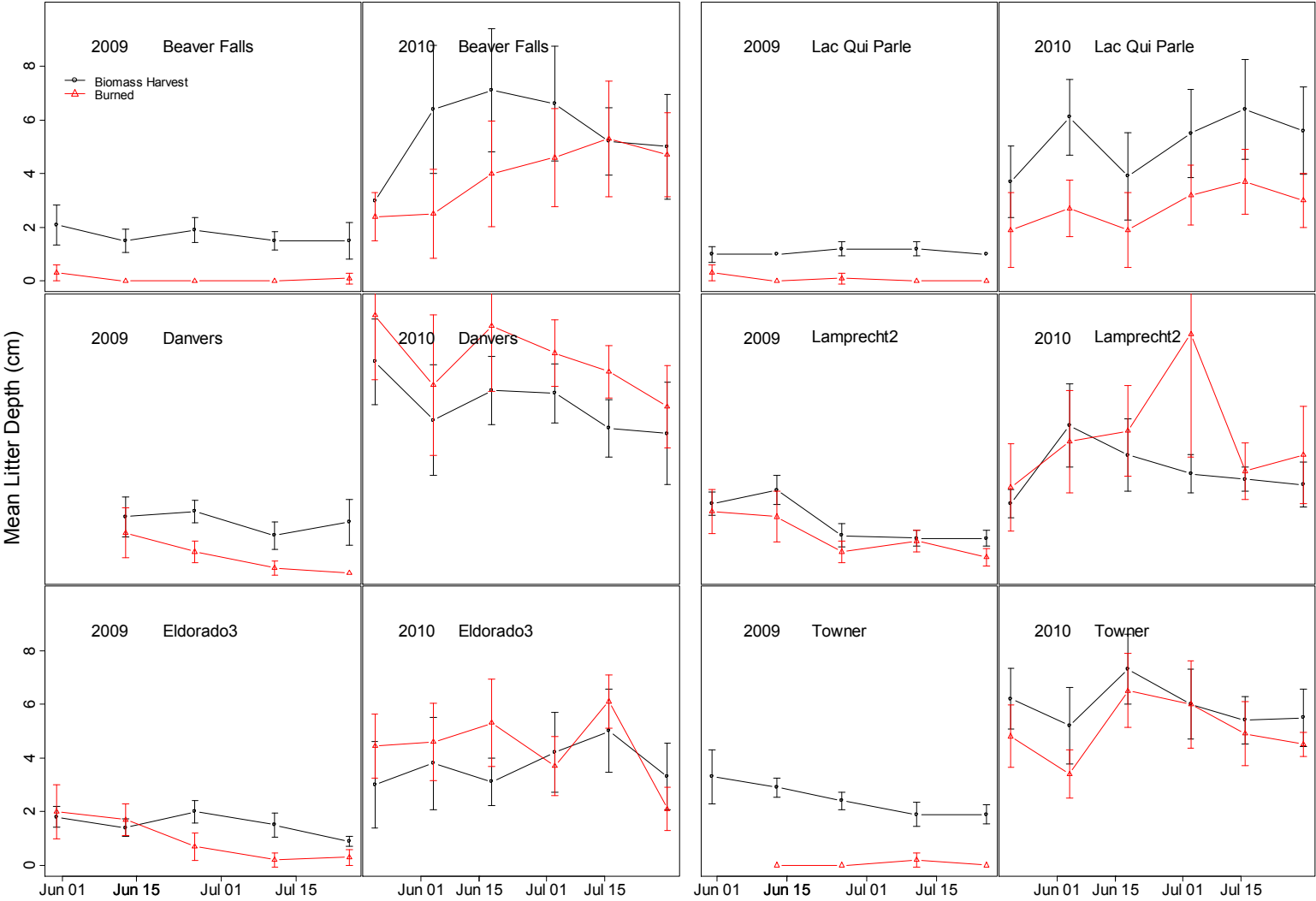
Figure 4b. Comparison of mean litter depth (dm) and 95% confidence intervals between 2 treatment subplots (a fall 2008 biomass harvest and a prescribed burn in spring 2009) within the same restored native grass field on 5 State Wildlife Management Areas and 1 Federal Waterfowl Production area in west-central Minnesota, in summers 2008 – 2010.
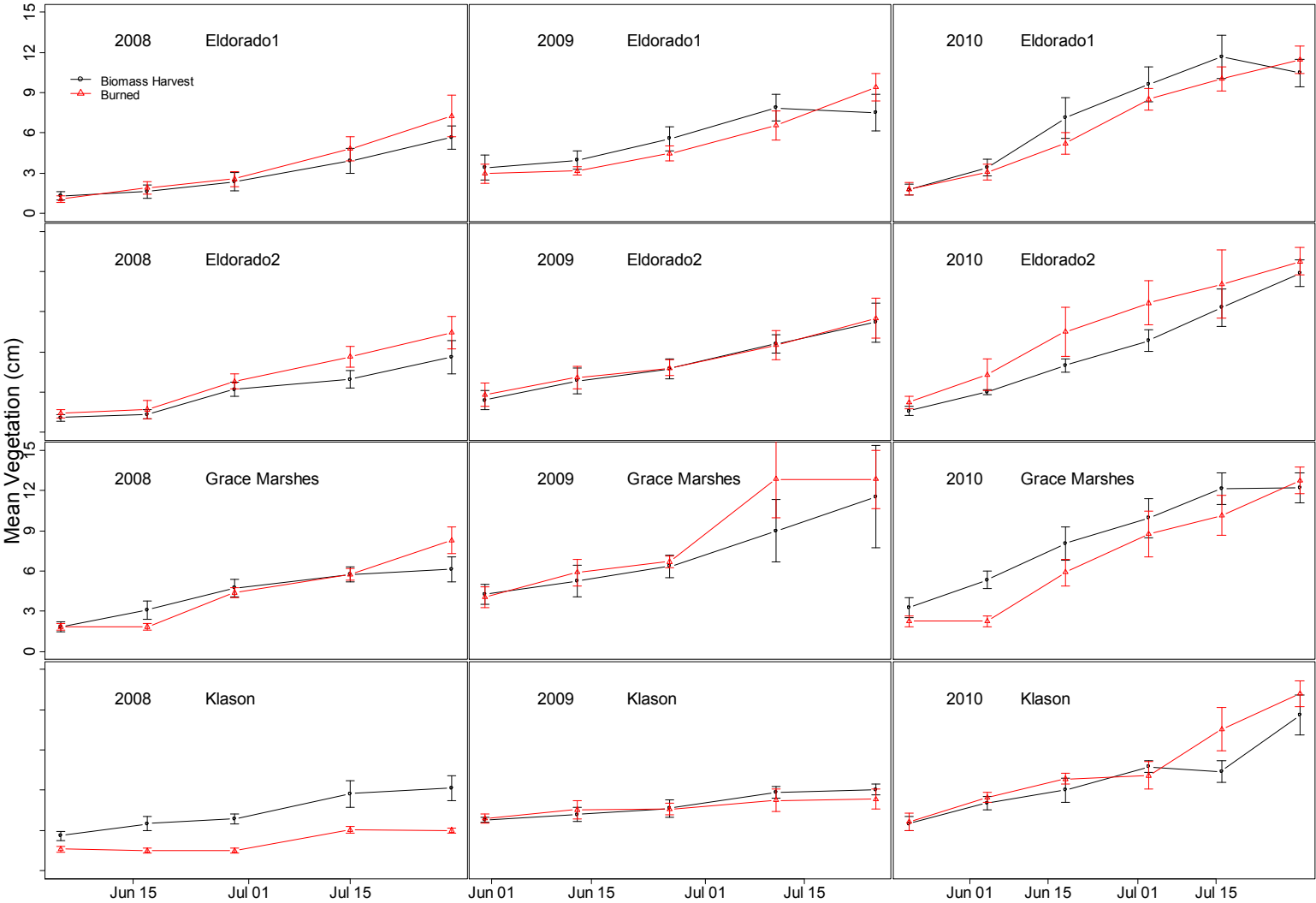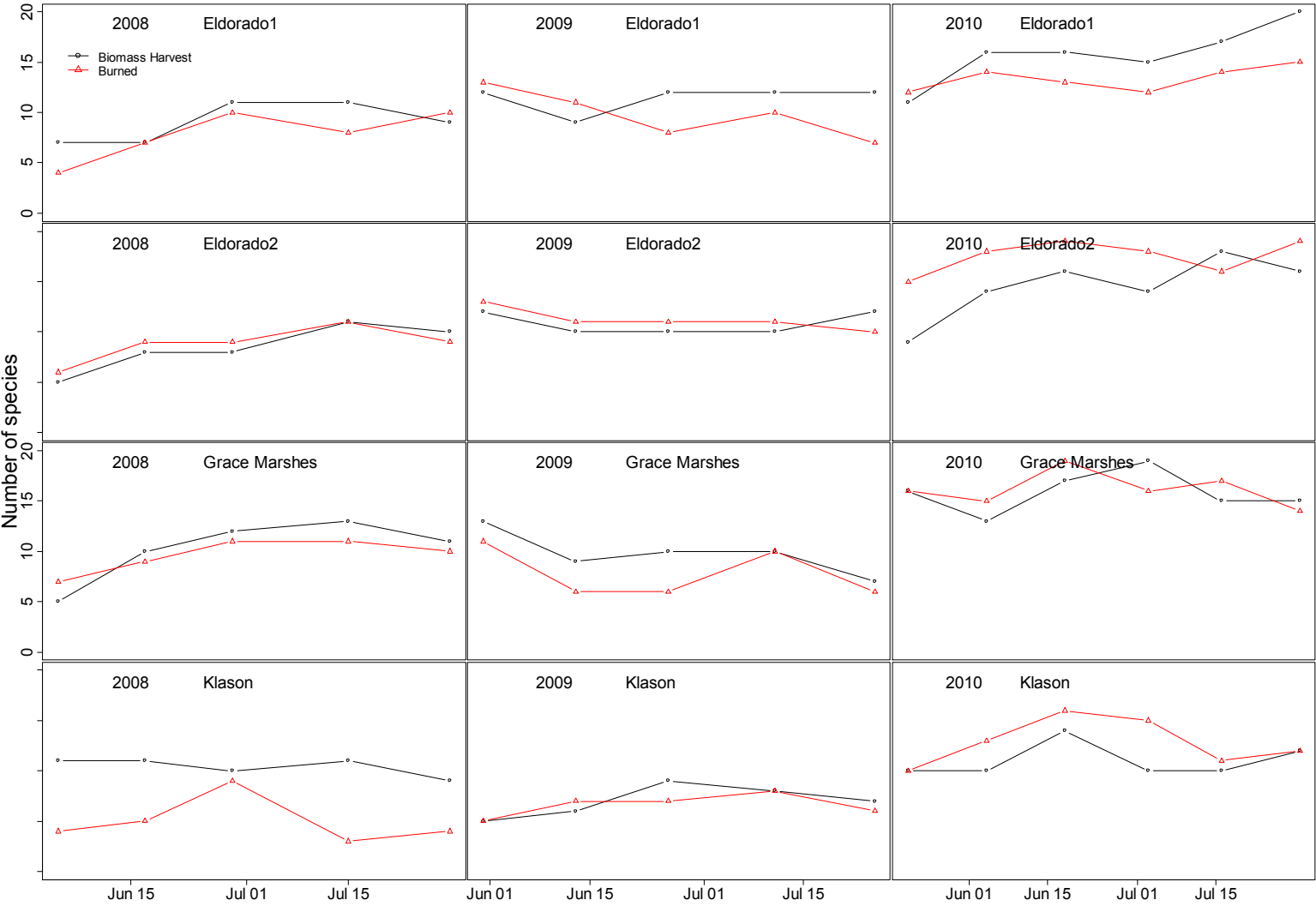
Figure 5a. Comparison of mean number of plant species per transect between 2 treatment subplots (a fall 2007 biomass harvest and a prescribed burn in spring 2008) within the same restored native grass field on 3 State Wildlife Management Areas (2 fields on one area) in west-central Minnesota, in summers 2008 – 2010.
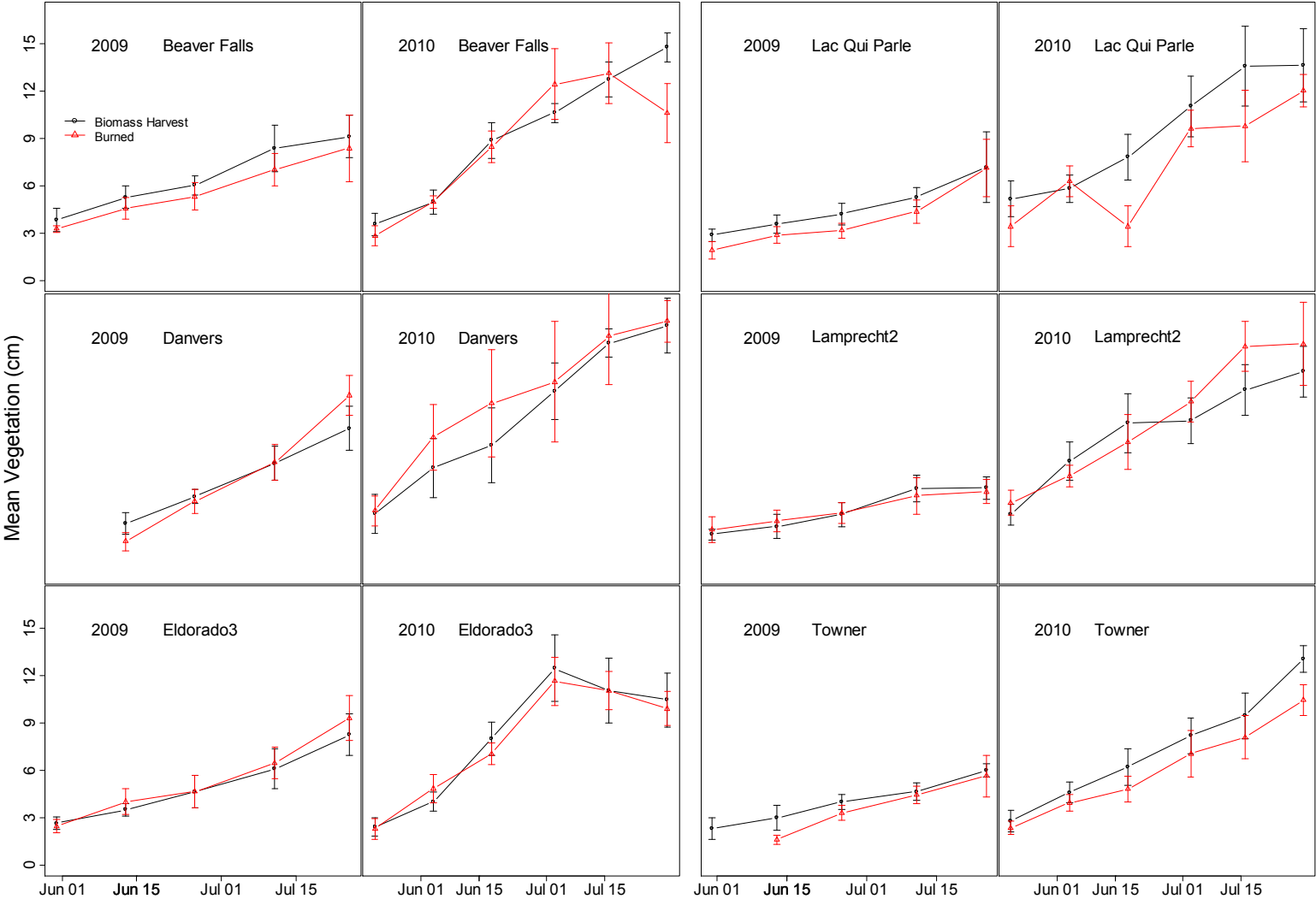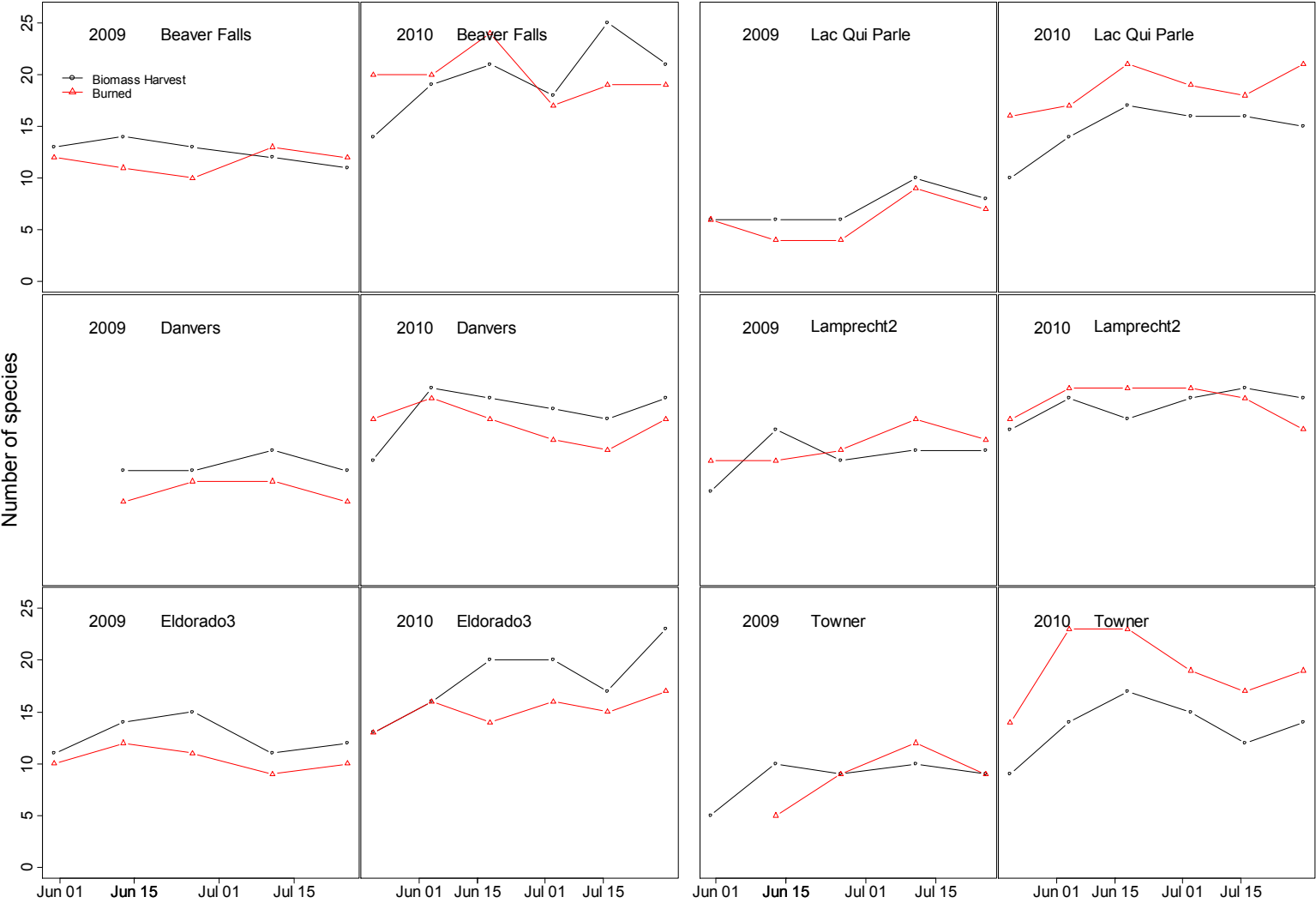
Figure 5b. Comparison of mean number of plant species per transect between 2 treatment subplots (a fall 2008 biomass harvest and a prescribed burn in spring 2009) within the same restored native grass field on 5 State Wildlife Management Areas and 1 Federal Waterfowl Production area in west-central Minnesota, 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

[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, $f*_{age * time * region}$. 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

$$\hat{f}* = \hat{H}\hat{\lambda},$$ (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. 2009b) to estimate $\hat{H}$. We used the delta method (Seber 1982) to estimate the variance ($\hat{var}$) of $\hat{H}$ as:

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

with $\hat{SE}\,(\hat{H}) = \sqrt{\hat{var}\,(\hat{H})}$ and the bounds of the 95% CIs associated with this parameter estimated as $\hat{H} \pm 1.96 * \hat{SE}\,(\hat{H})$. Our application of this method requires the assumption of spatiotemporally constant band reporting rates in Minnesota during the study period. Further, values of $\hat{var}\,(\hat{H})$ 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. 2009b).

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 $S_{age + \#\ Days1 + \#\ Days1^2}\ f_{age + \#\ Days1 + \#\ Days1^2}$ 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 ($f^*_{age + time,\ time * region,\ age + region}$ and $f^*_{age + time,\ time * region,\ age * region}$). 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 $f *_{age + \# Days1 + \# Days1^2}$ 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:

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

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. 2009b) 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. 2009a) 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. 2009a). 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 2009a,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. 2009b) 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. 2009b).

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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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 age + ecological province f age + ecological province	10	59,033.27	47.02	0	61,548.74
S age * ecological province f age * ecological province	16	59,035.51	49.27	0	61,538.56
S age f age * hunting zone	14	59,036.28	50.04	0	61,543.54
S age * region * time f age * region * time	176	59,037.18	50.77	0	61,204.38
S age f age + hunting zone + time	14	59,037.11	50.87	0	61,544.40
S age f age * hunting zone, L-time	44	59,037.27	51.02	0	61,481.87
S age f age + ecological province + time	12	59,040.62	54.37	0	61,552.23
S age f age * ecological province	10	59,040.78	54.54	0	61,556.58
S age f age + region + time	16	59,041.46	55.21	0	61,544.76
S age f age + hunting zone	9	59,043.52	57.27	0	61,561.52
S age * ecological province * time f age * ecological province * time	88	59,055.11	68.86	0	61,408.31
S age f age + region	11	59,060.77	74.53	0	61,575.34
S age f age + ecological province	7	59,071.58	85.33	0	61,594.96
S age f age * time	14	59,099.73	113.49	0	61,609.71
S age f age + time	9	59,102.17	115.93	0	61,622.70
S age + time f age + time	13	59,104.87	118.63	0	61,617.16
S age * time f 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 hunting zone f hunting zone	12	59,122.03	136.62	0	61,638.02
S ecological province f 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 age * ecological province, L-time f 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
$S_{\text{age} + \#\text{Days1} + \#\text{Days1}^2} \ f_{\text{age} + \#\text{Days1} + \#\text{Days1}^2}$	8	59,048.69	0	1.00	61,569.01
$S_{\text{age} + \#\text{Days1}} \ f_{\text{age} + \#\text{Days1}}$	6	59,088.99	40.30	0	61,615.21
$S_{\text{age} + \text{Total }\#\text{ Days} + \text{Total }\#\text{ Days}^2} \ f_{\text{age} + \text{Total }\#\text{ Days} + \text{Total }\#\text{ Days}^2}$	8	59,089.21	40.52	0	61,611.27
$S_{\text{age} * \#\text{Days1}} \ f_{\text{age} * \#\text{Days1}}$	8	59,091.11	42.42	0	61,613.25
$S_{\text{age} + \text{Total }\#\text{ Days}} \ f_{\text{age} + \text{Total }\#\text{ Days}}$	6	59,094.79	46.10	0	61,621.26
$S_{\text{age} * \text{Bag1}} \ f_{\text{age} * \text{Bag1}}$	8	59,096.15	47.46	0	61,618.50
$S_{\text{age} + \#\text{Days2} + \#\text{Days2}^2} \ f_{\text{age} + \#\text{Days2} + \#\text{Days2}^2}$	8	59,097.58	48.88	0	61,619.99
$S_{\text{age} * \text{Total }\#\text{ Days}} \ f_{\text{age} * \text{Total }\#\text{ Days}}$	8	59,097.84	49.15	0	61,620.27
$S_{\text{age} + \text{Bag2}} \ f_{\text{age} + \text{Bag2}}$	6	59,097.89	49.19	0	61,624.49
$S_{\text{age} + \#\text{Days2}} \ f_{\text{age} + \#\text{Days2}}$	6	59,098.66	49.97	0	61,625.30
$S_{\text{age} + \text{Julian2} + \text{Julian2}^2} \ f_{\text{age} + \text{Julian2} + \text{Julian2}^2}$	8	59,098.87	50.18	0	61,621.34
$S_{\text{age} + \#\text{DaysSplit1,2} + \#\text{DaysSplit1,2}^2} \ f_{\text{age} + \#\text{DaysSplit1,2} + \#\text{DaysSplit1,2}^2}$	8	59,100.36	51.67	0	61,622.89
$S_{\text{age} + \#\text{DaysSplit1,2}} \ f_{\text{age} + \#\text{DaysSplit1,2}}$	6	59,101.50	52.81	0	61,628.26
$S_{\text{age} * \text{Bag2}} \ f_{\text{age} * \text{Bag2}}$	8	59,101.87	53.18	0	61,624.47
$S_{\text{age} + \text{Bag2} + \text{Bag2}^2} \ f_{\text{age} + \text{Bag2} + \text{Bag2}^2}$	8	59,101.89	53.20	0	61,624.49
$S_{\text{age} * \#\text{Days2}} \ f_{\text{age} * \#\text{Days2}}$	8	59,102.22	53.53	0	61,624.84
$S_{\text{age} * \text{Bag3}} \ f_{\text{age} * \text{Bag3}}$	8	59,102.65	53.95	0	61,625.28
$S_{\text{age} + \text{Bag1}} \ f_{\text{age} + \text{Bag1}}$	6	59,104.65	55.96	0	61,631.55
$S_{\text{age} * \#\text{DaysSplit1,2}} \ f_{\text{age} * \#\text{DaysSplit1,2}}$	8	59,105.14	56.45	0	61,627.88
$S_{\text{age} + \text{Bag3}} \ f_{\text{age} + \text{Bag3}}$	6	59,106.71	58.02	0	61,633.70
$S_{\text{age} + \#\text{Days3}} \ f_{\text{age} + \#\text{Days3}}$	6	59,106.88	58.18	0	61,633.87
$S_{\text{age} * \#\text{Days3}} \ f_{\text{age} * \#\text{Days3}}$	8	59,108.22	59.53	0	61,631.10
$S_{\text{age} + \text{Bag3} + \text{Bag3}^2} \ f_{\text{age} + \text{Bag3} + \text{Bag3}^2}$	8	59,108.30	59.61	0	61,631.18
$S_{\text{age} + \text{Bag1} + \text{Bag1}^2} \ f_{\text{age} + \text{Bag1} + \text{Bag1}^2}$	8	59,108.66	59.96	0	61,631.55
$S_{\text{age} + \text{Julian1}} \ f_{\text{age} + \text{Julian1}}$	6	59,108.79	60.10	0	61,635.86
$S_{\text{age} + \#\text{Days3} + \#\text{Days3}^2} \ f_{\text{age} + \#\text{Days3} + \#\text{Days3}^2}$	8	59,110.88	62.19	0	61,633.87
$S_{\text{age} * \text{Julian1}} \ f_{\text{age} * \text{Julian1}}$	8	59,111.30	62.61	0	61,634.30
$S_{\text{age} + \text{Julian1} + \text{Julian1}^2} \ f_{\text{age} + \text{Julian1} + \text{Julian1}^2}$	8	59,112.79	64.10	0	61,635.87
$S_{\text{age} + \text{Julian2}} \ f_{\text{age} + \text{Julian2}}$	6	59,112.84	64.15	0	61,640.09
$S_{\text{age} * \text{Julian2}} \ f_{\text{age} * \text{Julian2}}$	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.

| Length of early goose hunting season (days) | f * | | f | | S | |
| | Age class | | Age class | | Age class | |
	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^*_{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^*_{age}	2	28,504.19	128.93	0	28,500.19
f^*_{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.

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

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

Region [a]	Age class[b]	2002–2003	2003–2004	2004–2005	2005–2006	2006–2007	2007–2008
				Time			
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 banding [b]	n [c]	Region of recovery [b]								
		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 X̄		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 X̄		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 X̄		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 X̄		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.

Region [a]	n [b]	Proportion directly-recovered in-state by time period		
		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.
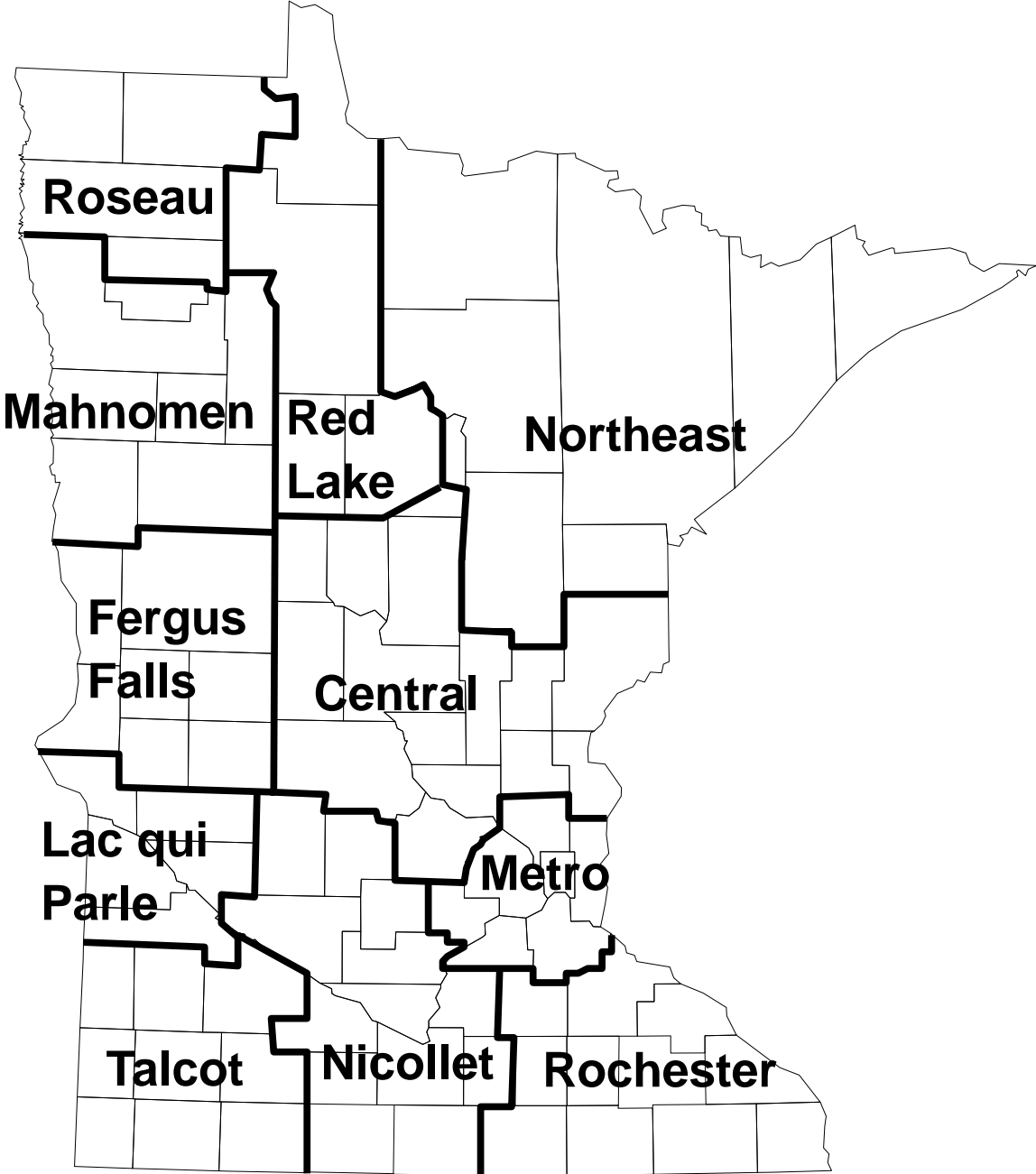
Figure 1. Goose Management Blocks of Minnesota in which large Canada geese were leg-banded during 2002-2007.
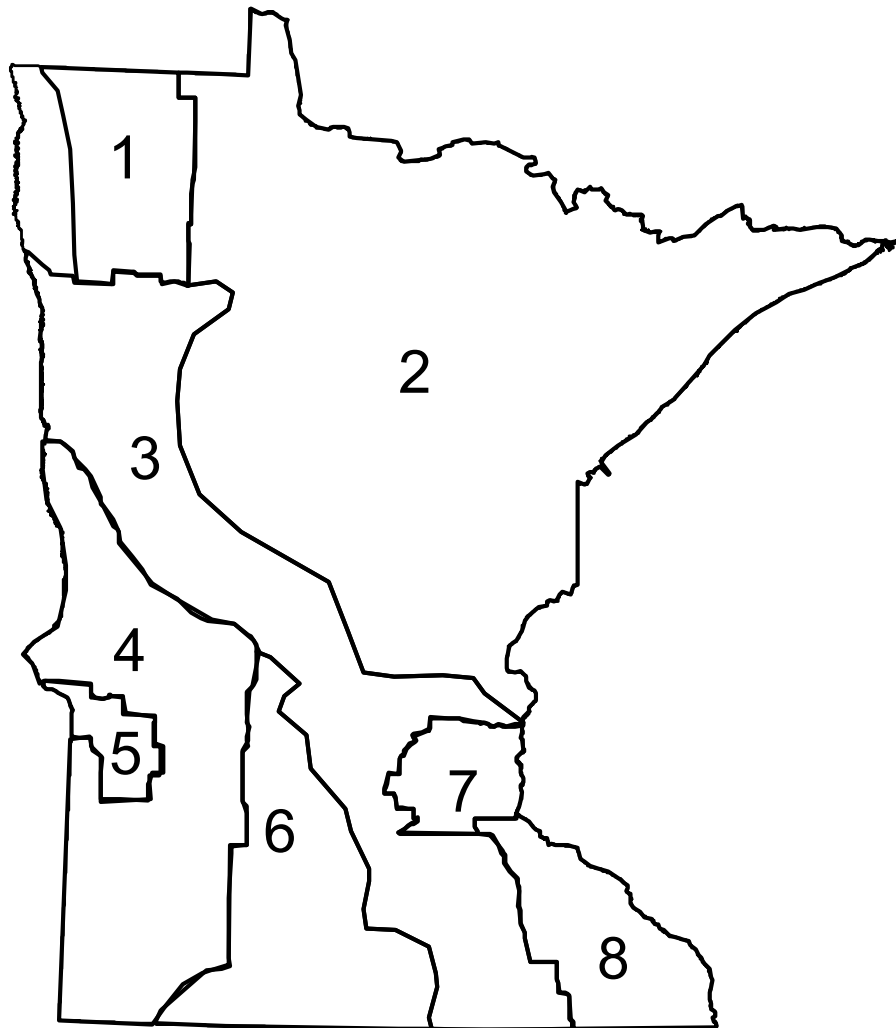
Figure 2. The regions (combinations of ecological provinces and hunting zones) of Minnesota in which large Canada geese were leg-banded during 2002–2007. The regions are: 1 = Northwest Zone–Aspen Parkland, 2 = Rest-of-State Zone–Forest, 3 = Rest-of-State Zone–Transition, 4 = West Zone–Prairie, 5 = Westcentral Zone–Prairie, 6 = Rest-of-State Zone–Prairie, 7 = Metro Zone–Transition, 8 = Southeast Zone–Transition.
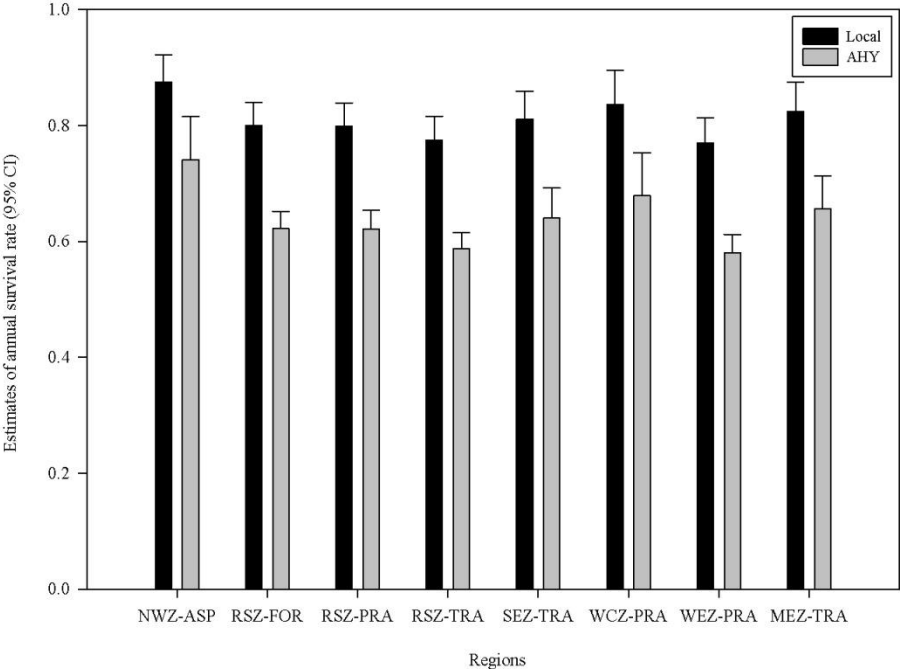
Figure 3. The age-specific annual survival rate estimates of large Canada geese captured in 8 regions (NWZ-ASP [Northwest Zone–Aspen Parkland], RSZ-FOR [Rest-of-State Zone–Forest], RSZ-PRA [Rest-of-State Zone–Prairie], RSZ-TRA [Rest-of-State Zone–Transition], SEZ-TRA [Southeast Zone–Transition], WCZ-PRA [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.
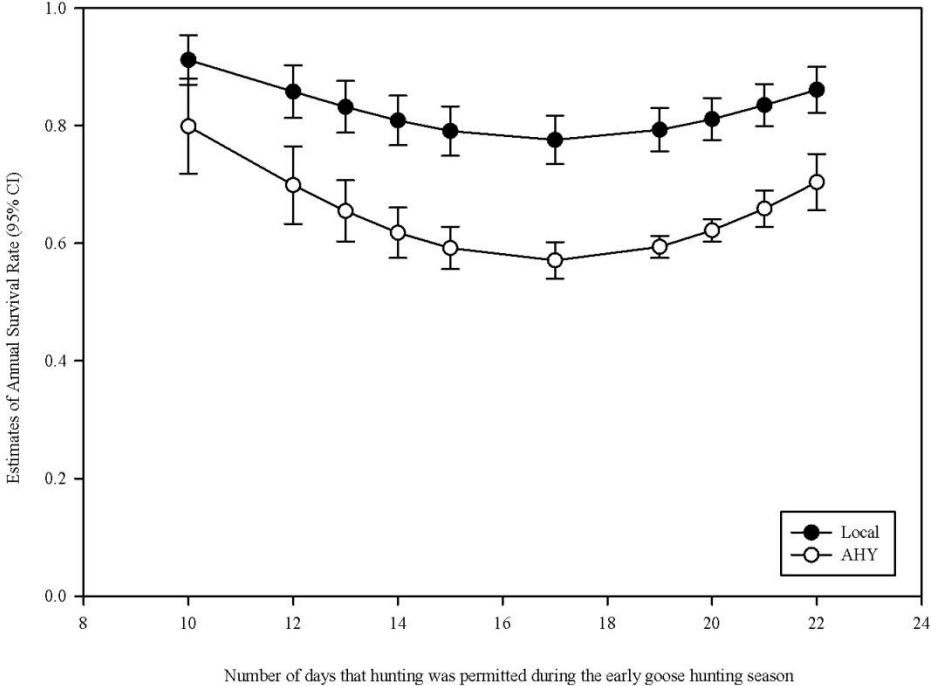
Figure 4. The influence of the number of days that hunting was permitted during the early goose hunting season on age-specific annual survival rates of large Canada geese captured in Minnesota during 2002–2007.
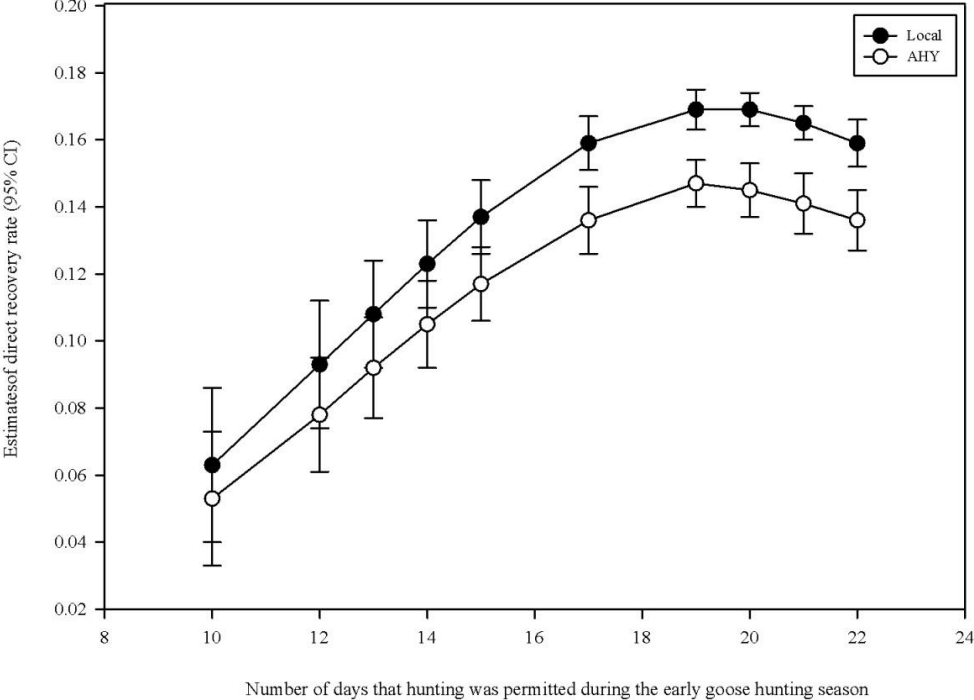
Figure 5.   The influence of the number of days that hunting was permitted during the early goose hunting season on the direct recovery rates of large Canada geese captured in Minnesota during 2002–2007.
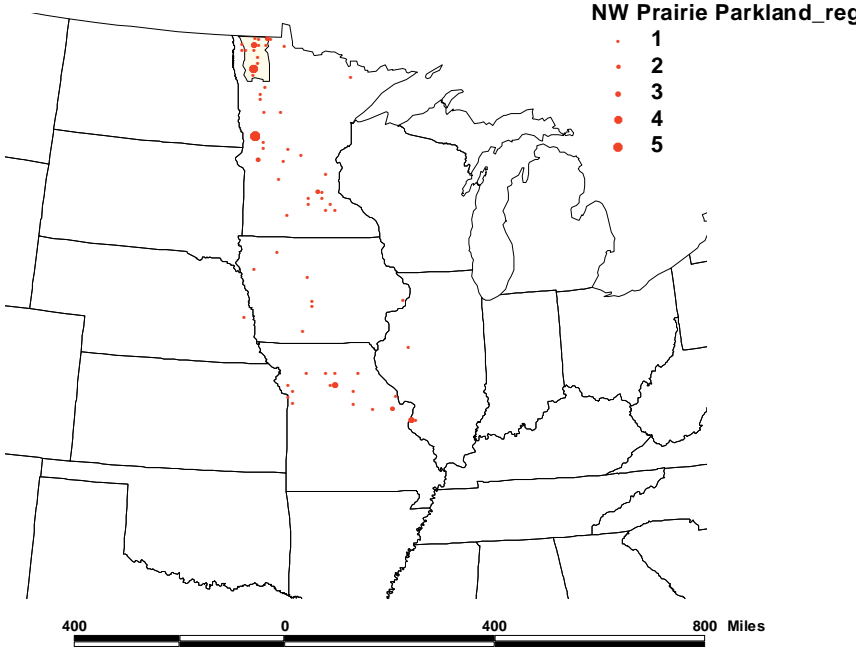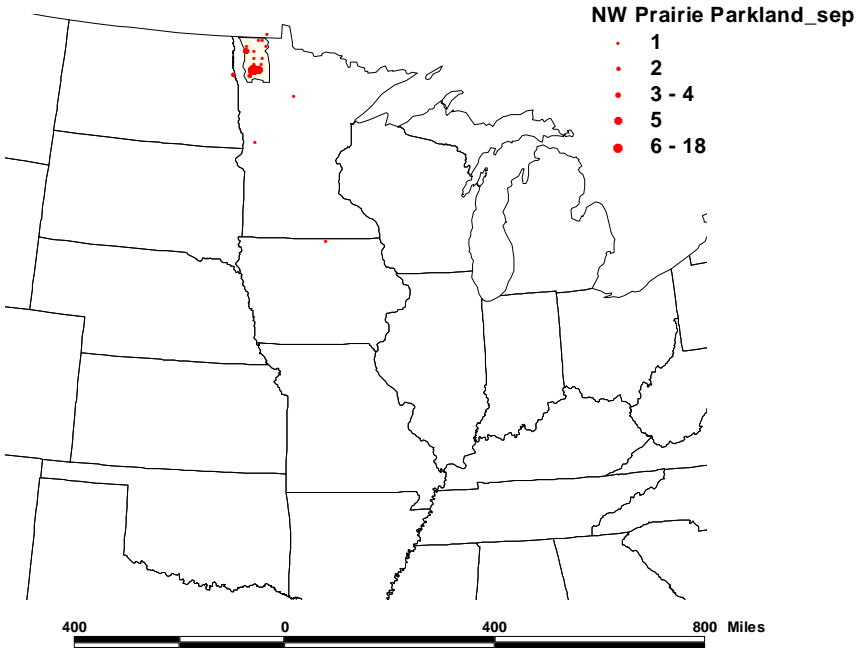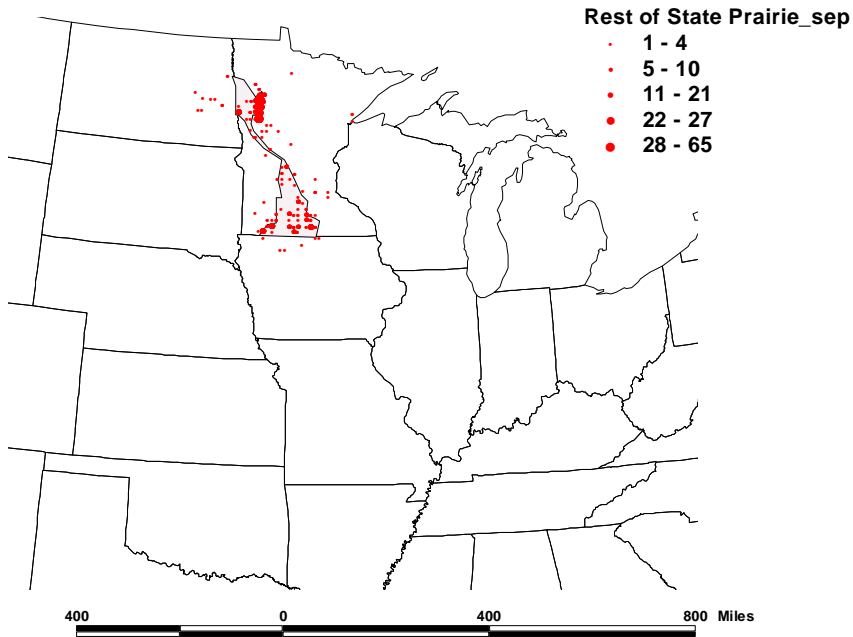
Figure 6. Distribution of direct recoveries of large Canada geese banded in the Northwest Zone –Aspen Parkland region and recovered during the 1–22 September (top) and 23 September–21 February (bottom) portions of the 2002–2003 to 2007–2008 hunting seasons. Figure excludes one regular season recovery in Manitoba.
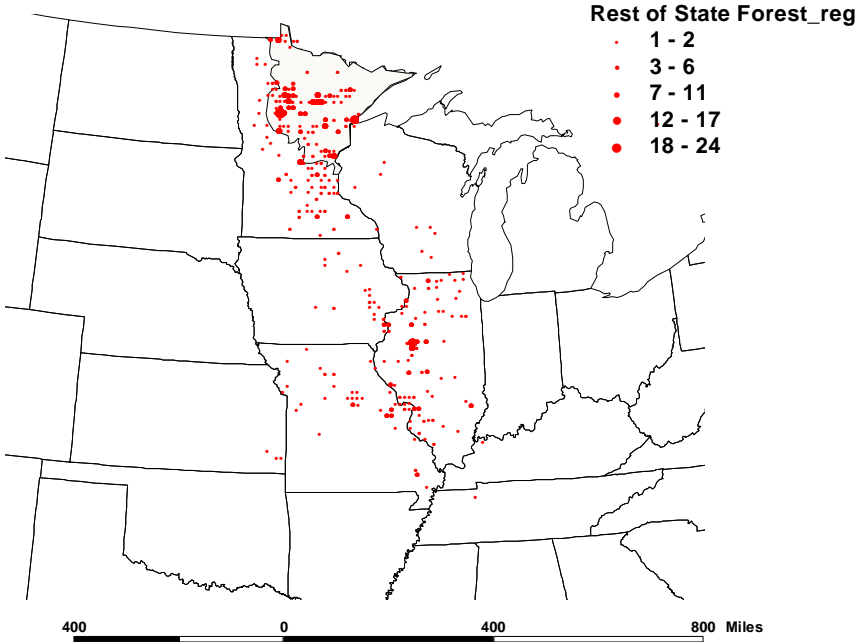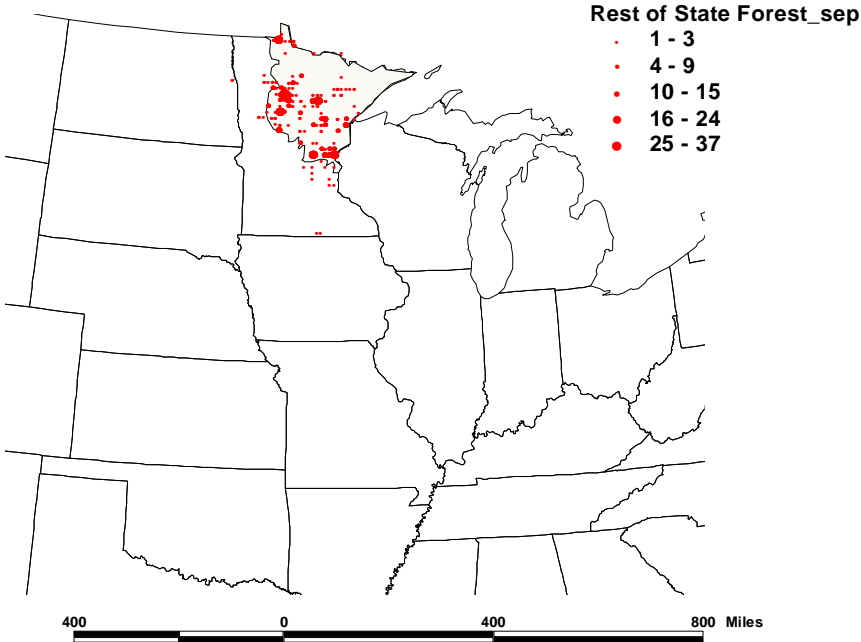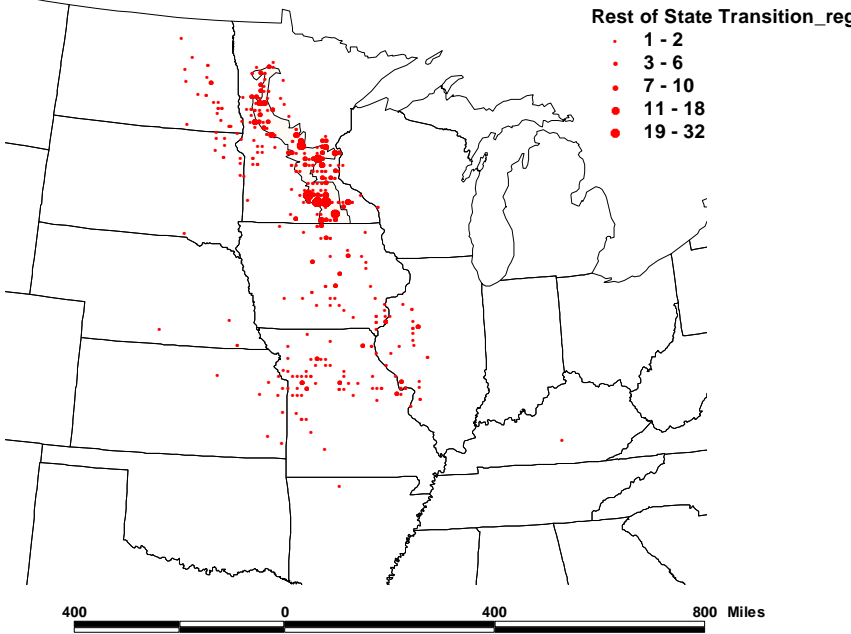
Figure 7. Distribution of direct recoveries of large Canada geese banded in the Rest-of-State Zone–Forest region and recovered during the 1–22 September (top) and 23 September–21 February (bottom) portions of the 2002–2003 to 2007–2008 hunting seasons. Figure excludes 1 early season recovery in New Jersey, 1 regular season recovery in Manitoba, and 1 regular season recovery in Ontario.
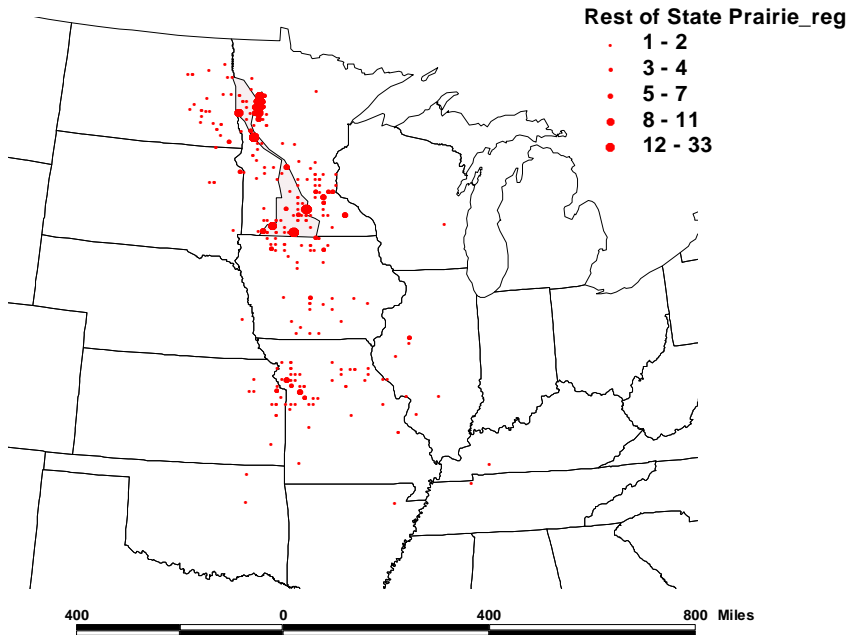
Figure 8. Distribution of direct recoveries of large Canada geese banded in the Rest-of-State Zone–Prairie region and recovered during the 1–22 September (top) and 23 September–21 February (bottom) portions of the 2002–2003 to 2007–2008 hunting seasons. Figure excludes 1 early season recovery in Montana and 1 early season recovery in Virginia.
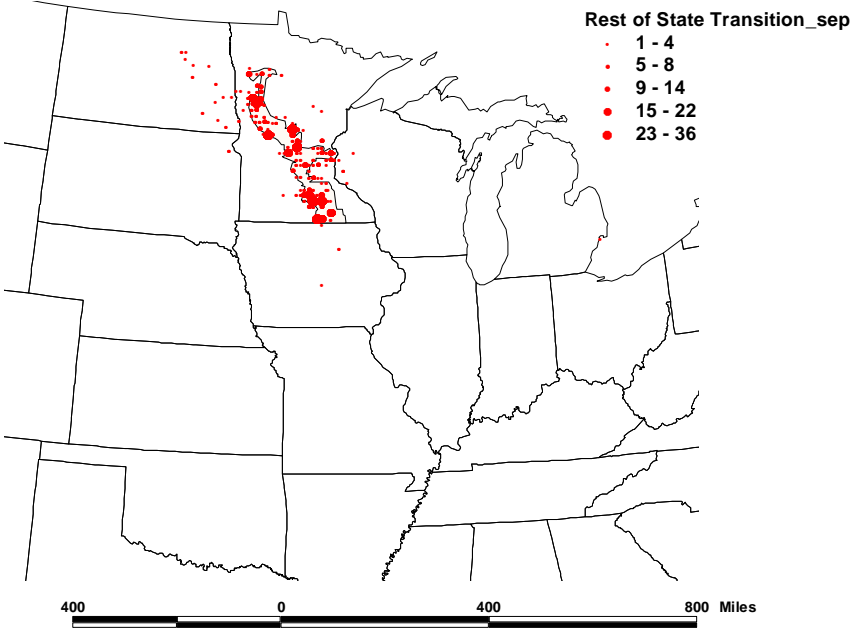
Figure 9. Distribution of direct recoveries of large Canada geese banded in the Rest-of-State Zone–Transition region and recovered during the 1–22 September (top) and 23 September–21 February (bottom) portions of the 2002–2003 to 2007–2008 hunting seasons.
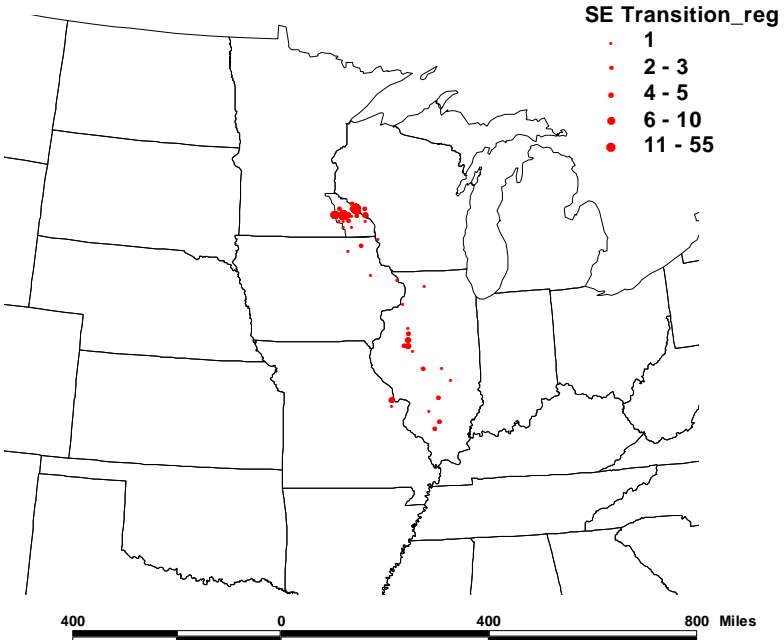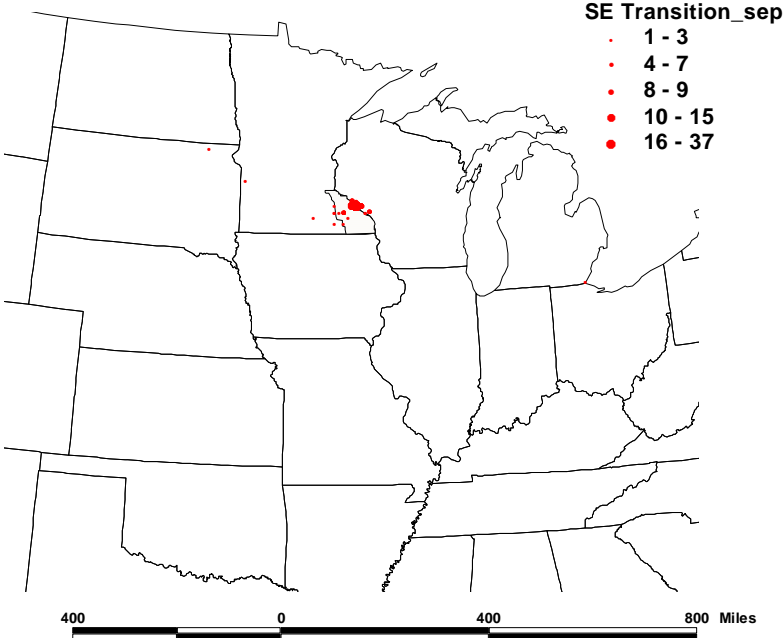
Figure 10. Distribution of direct recoveries of large Canada geese banded in the Southeast Zone–Transition region and recovered during the 1–22 September (top) and 23 September–21 February (bottom) portions of the 2002–2003 to 2007–2008 hunting seasons.
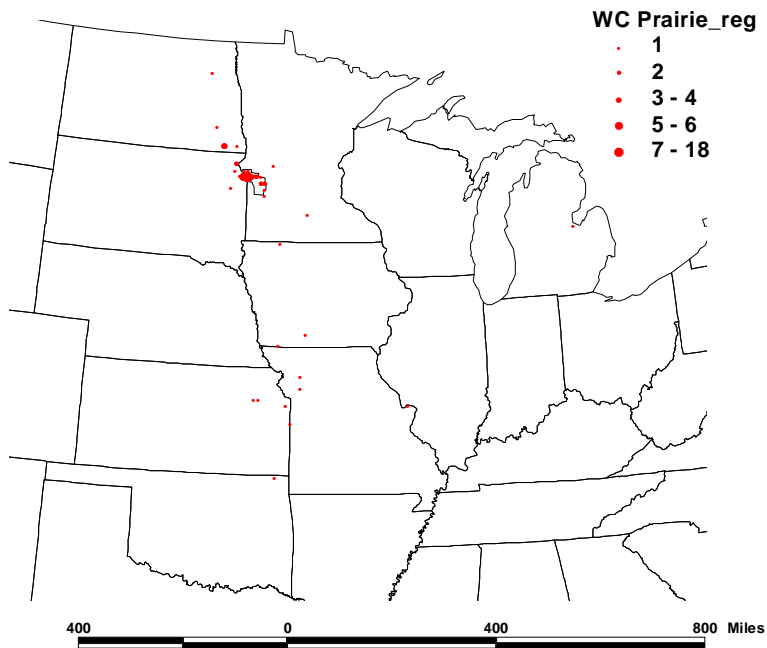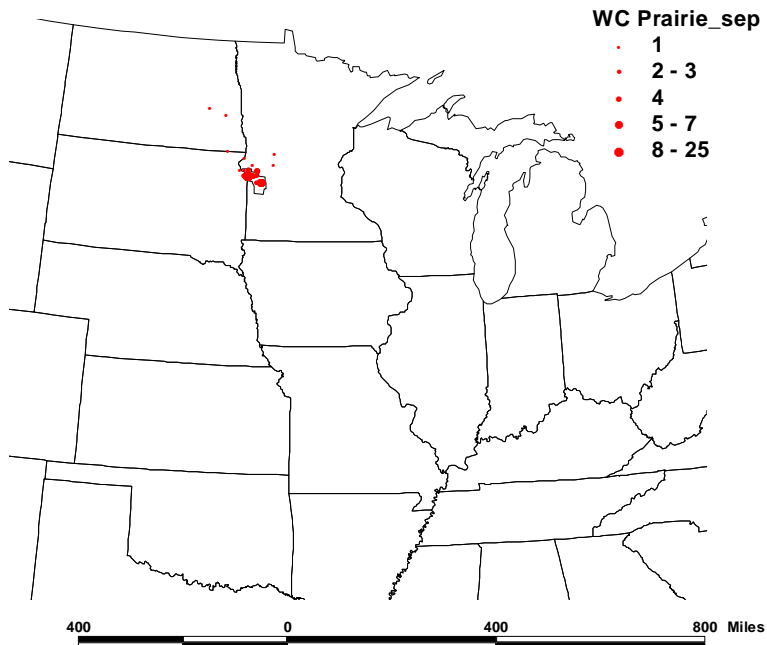
Figure 11. Distribution of direct recoveries of large Canada geese banded in the West-central Zone–Prairie region and recovered during the 1–22 September (top) and 23 September–21 February (bottom) portions of the 2002–2003 to 2007–2008 hunting seasons.
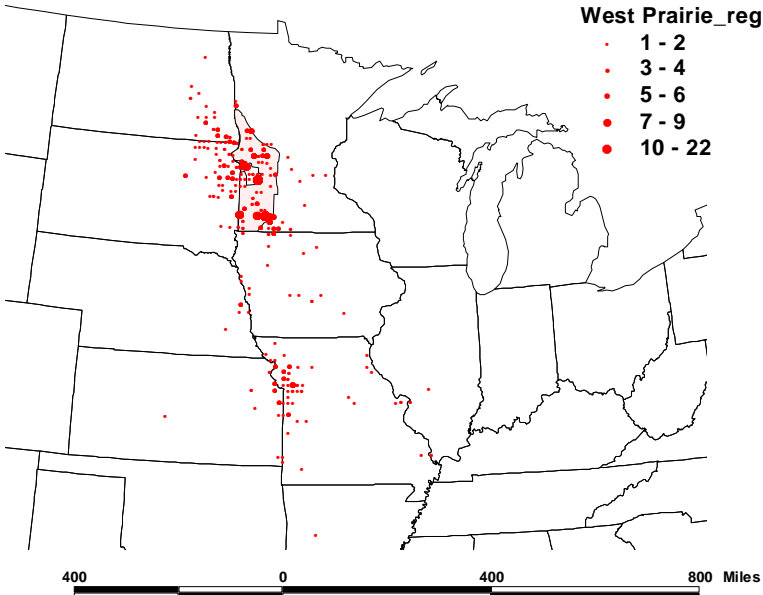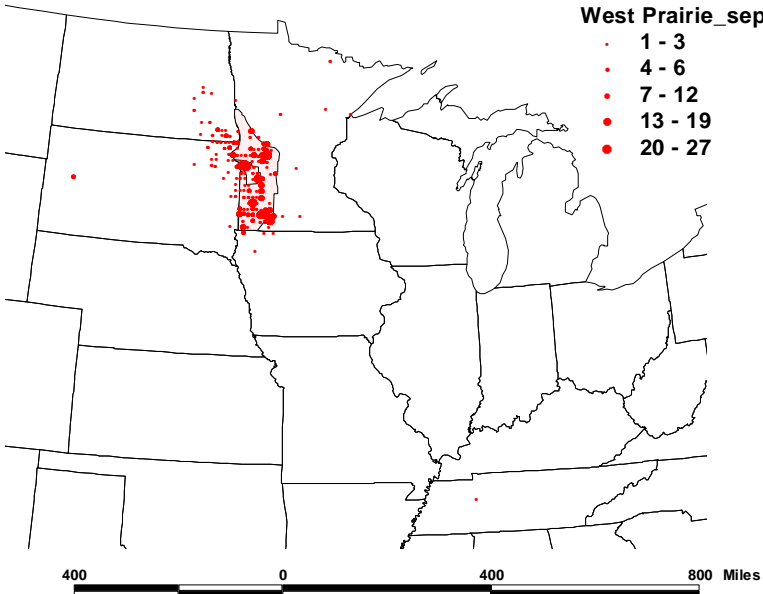
Figure 12. Distribution of direct recoveries of large Canada geese banded in the West Zone–Prairie region and recovered during the 1–22 September (top) and 23 September–21 February (bottom) portions of the 2002–2003 to 2007–2008 hunting seasons. Figure excludes 1 early season recovery in Utah, 1 regular season recovery in Manitoba, 1 regular season recovery in Mississippi, 1 regular season recovery in Nevada, and 1 regular season recovery in New Brunswick.
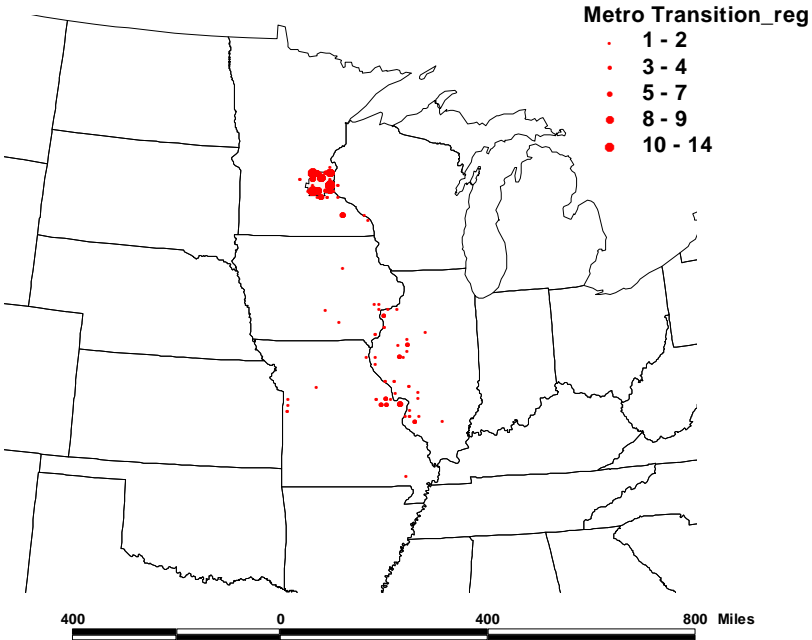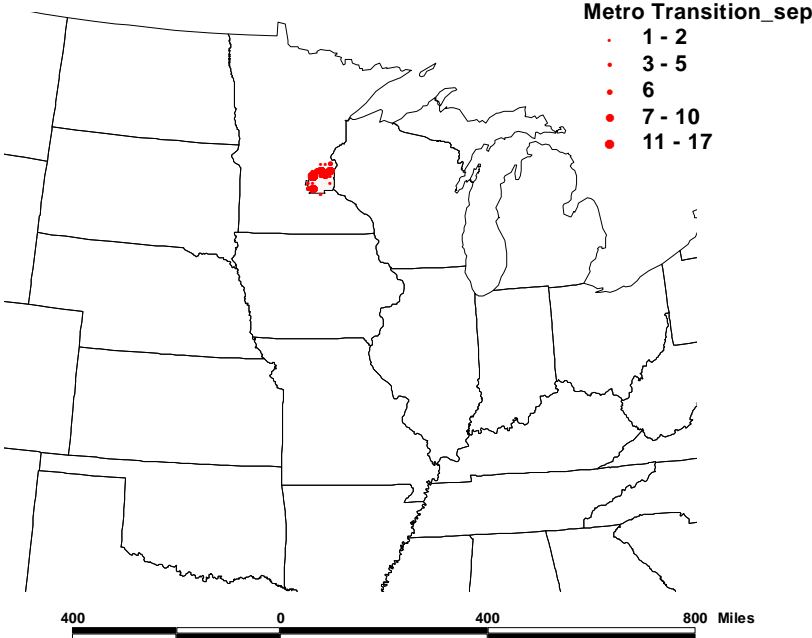
Figure 13. Distribution of direct recoveries of large Canada geese banded in the Metro Zone–Transition region and recovered during the 1–22 Sep (top) and 23 Sep–21 Feb (bottom) portions of the 2003–2004 to 2007–2008 hunting seasons. Figure excludes 1 early season recovery in Manitoba.

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

Hunting Zone	Year	Dates of hunting seasons			Number of days of large Canada goose hunting				Daily bag limit		
		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 ≤ 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 1 record 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.

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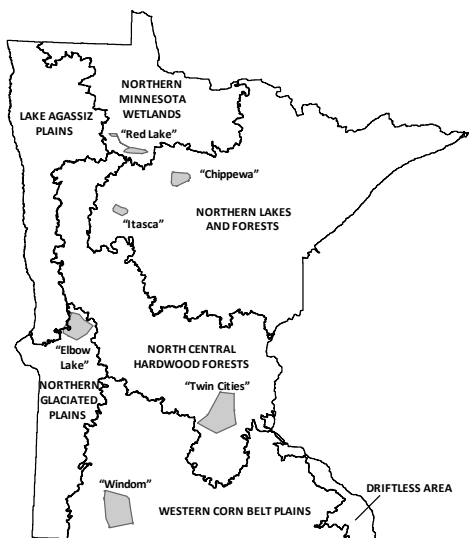
Figure 1.   Map showing locations of proposed study areas (shaded gray) in relationship to Minnesota's aquatic ecoregions (thick black lines).
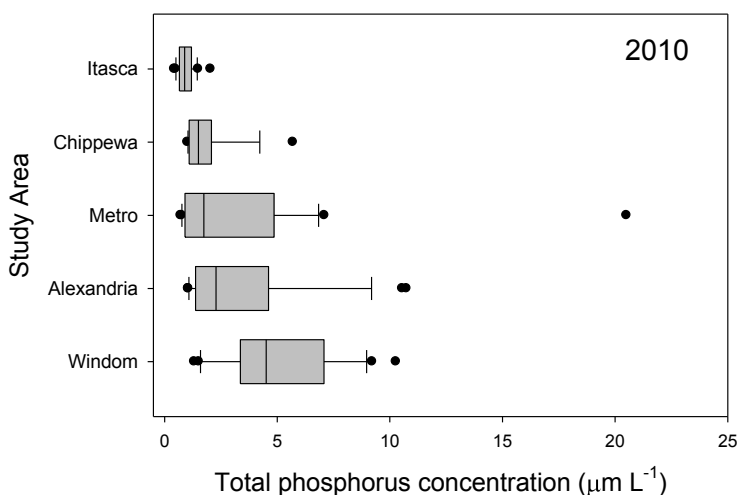

Figure 2.   Box plots showing mean abundance of total phosphorus (TP) for 127 shallow lakes sampled within 5 study areas during 2010. Vertical lines within boxes depict median TP values for each study area; boxes depict 25th and 75th percentiles. Whiskers show 10th and 90th percentiles, with dots indicating more extreme values.
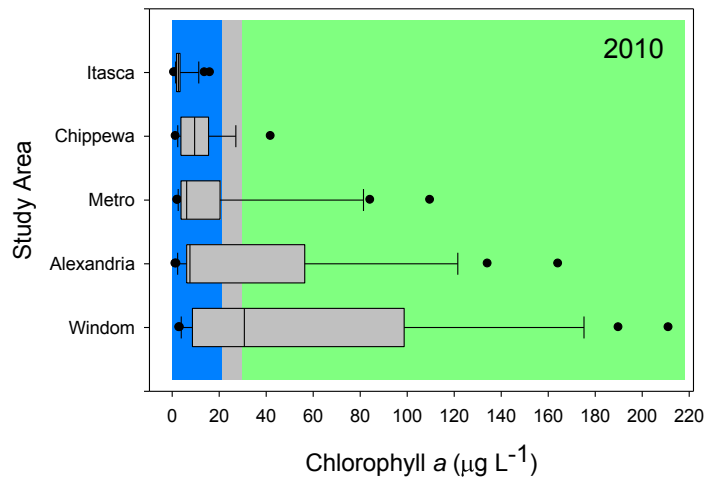
Figure 3. Box plots showing mean abundance of phytoplankton (chlorophyll *a* concentration) for 127 shallow lakes sampled within 5 study areas during 2010. Background colors depict expected chlorophyll *a* regions for clear- (blue), unstable (grey), and turbid-regimes (green) based on threshold values of Zimmer et al. (2009).
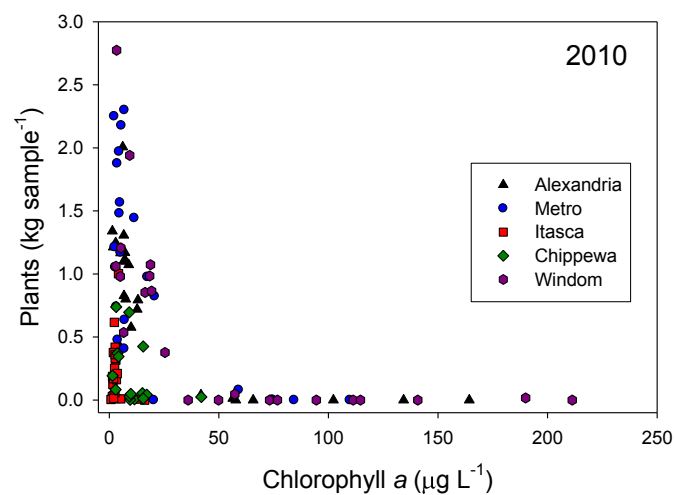

Figure 4. Abundance of phytoplankton (chlorophyll *a* concentration) and submerged macrophyte biomass for 127 shallow lakes in 5 study areas during 2010. Plant mass indicates average wet weight of plants collected on rake casts in each lake. Colors depict study area as indicated in legend.
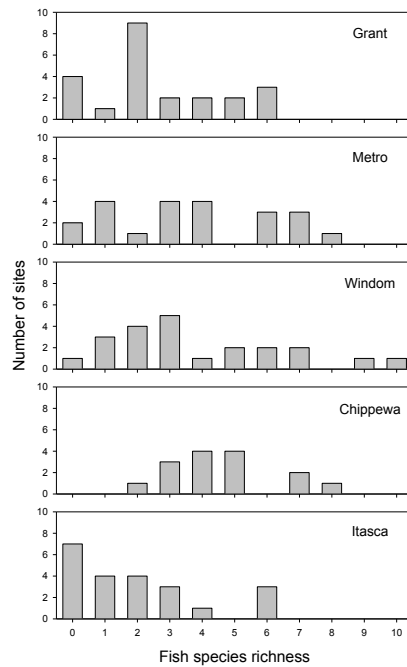
Figure 5.   Fish species richness for 127 shallow lakes in 5 study areas during 2010. Height of bars on *x*-axis depicts number of lakes in which corresponding number of fish species were collected.
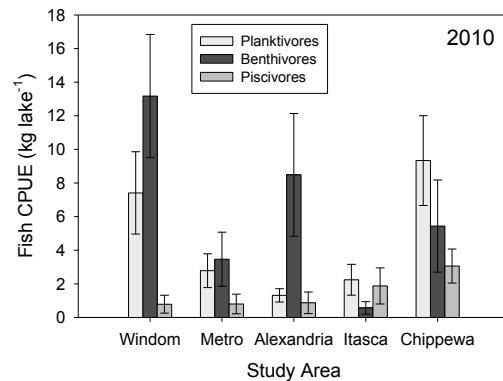

Figure 6.   Summary of fish relative abundance for 127 shallow lakes in 5 study areas during 2010. Height of bars on *x*-axis depicts average weight (mass) for each of 3 major fish feeding guilds common in these lakes (planktivores, benthivores, piscivores).