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, suggesting 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. 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. 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 intend to survey vegetation in additional plots in which biomass harvest/burn treatments are applied, and using these additional data, will determine whether 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. 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 early June 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) that comprised $\geq$ 10% of the total number of individual plants 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. The amount of these species in each field will be estimated using distance sampling (Buckland et al. 2004).

We also examined seasonal and temporary wetlands in mid-April that had vegetation removed, primarily cattails, during biomass harvest the previous fall. 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 2-6). 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.

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.

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 is 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 2 field seasons, there appears to be little difference in vegetation characteristics between bioharvested and burned subplots. We will monitor all subplots again in 2010 to look for any vegetative differences among subplots that may occur with time. Further, 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. 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.

ACKNOWLEDGEMENTS

LITERATURE CITED

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

Buckland, S. T., D. R. Anderson, K. P. Burnham, J. L. Laake, D. L. Borchers, and L. Thomas. 2004. Introduction to distance sampling: estimating abundance of biological populations. Oxford University Press 416pp.

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

Perlack, R. D., L. L. Wright, A. F. Turhollow, R. L. Graham, B. J. Stokes, D. C. Erbach. 2005. Biomass as feedstock for bioenergy and Bioproducts industry. The technical feasibility

of a billion ton annual supply. US Department of Energy, Document Number: DOE/GO-102995-21 35.

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

Sample, D. W., and M. J. Mossman. 1997. Managing Habitat for Grassland Birds. A Guide for Wisconsin. Department of Natural Resources, Madison, Wisconsin, 154pp.

Steel RGD, Torrie JH, and Dickey D. 1997. Principles and procedures of statistics: a Biometrical Approach, 3rd edition. McGraw-Hill, Boston, MA, USA.

Stewart, R. E., and H. A. Kantrud. 1971. Classification of natural ponds and lakes in the glaciated prairie region. U.S. Fish and Wildlife Service Resource Publication 92, 57p.

Zicus, M. C., D. P. Rave, A. Das, M. R. Riggs, and M. L. Buitenwerf. 2006. Influence of land use on mallard nest-structure occupancy. Journal of Wildlife Management, 70(5): 1325-1333.
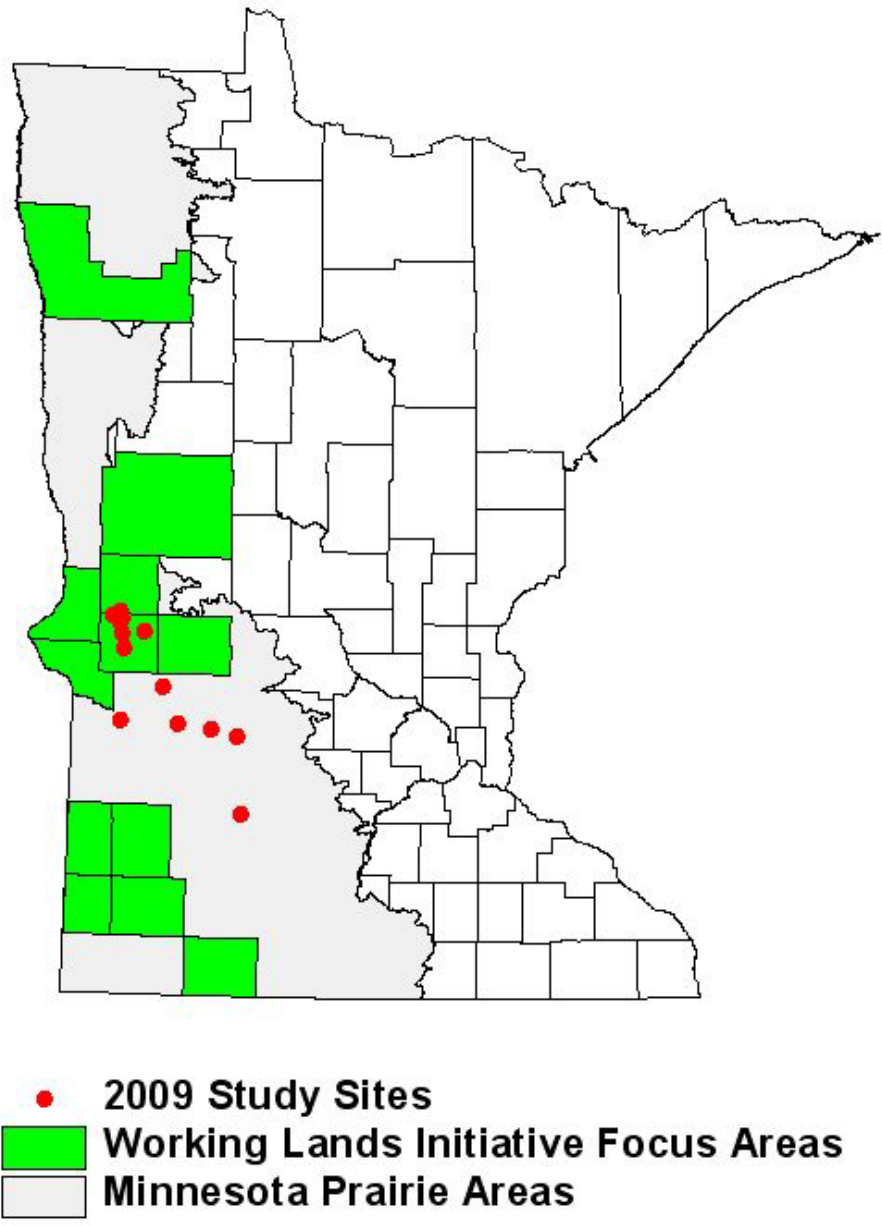
Figure 1. Minnesota counties showing prairie areas and Working Lands Initiative focus areas, 2009.
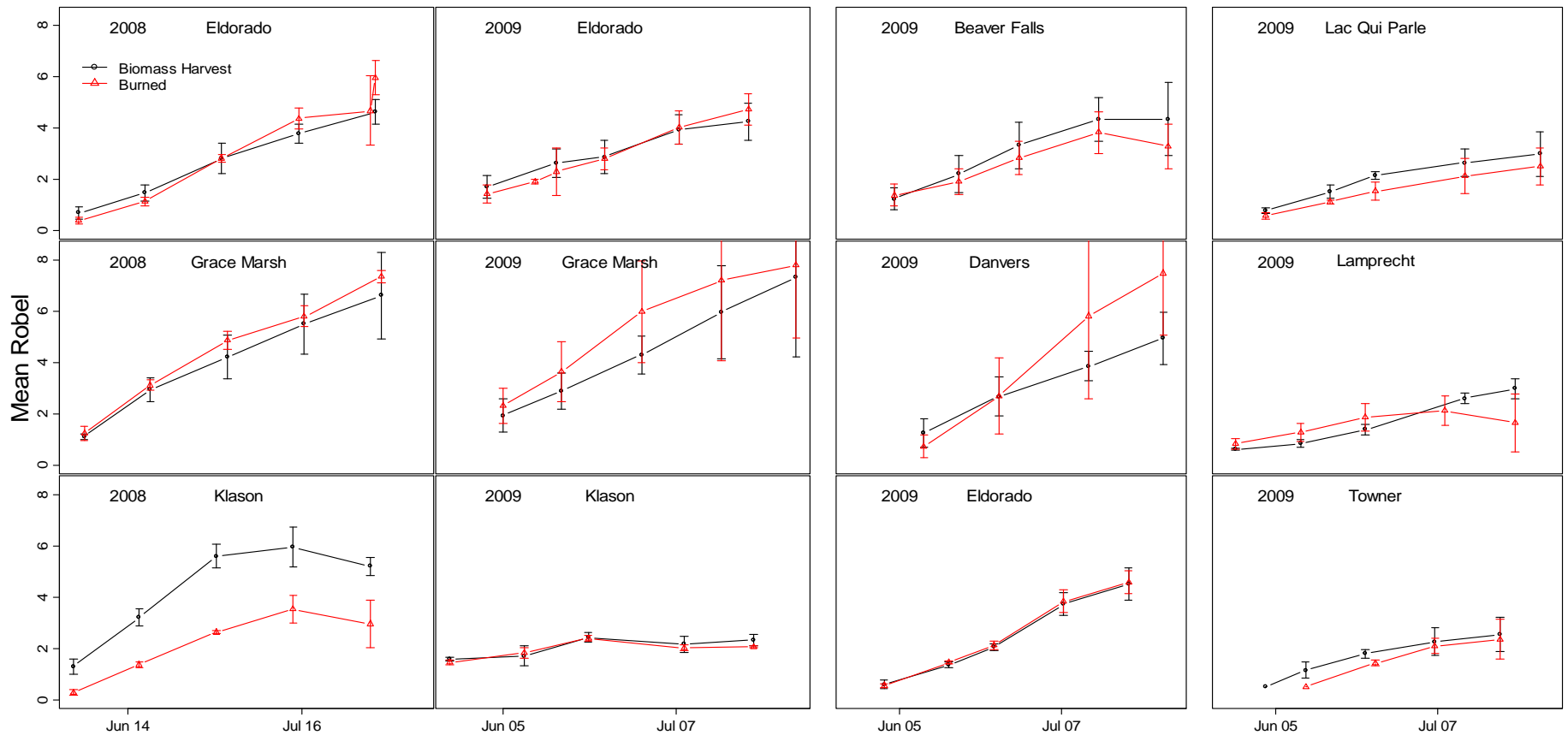
Figure 2. Comparison of mean Robel measurements (dm) and 95% confidence intervals between 2 treatment subplots (a fall biomass harvest and a prescribed burn the following spring) within the same restored native grass field on 3 State Wildlife Management Areas in west-central Minnesota, in both summer 2008 and summer 2009 (leftmost two columns), and on 5 State Wildlife Management Areas and 1 Federal Waterfowl Production area in west-central Minnesota, in only summer 2009 (rightmost two columns).
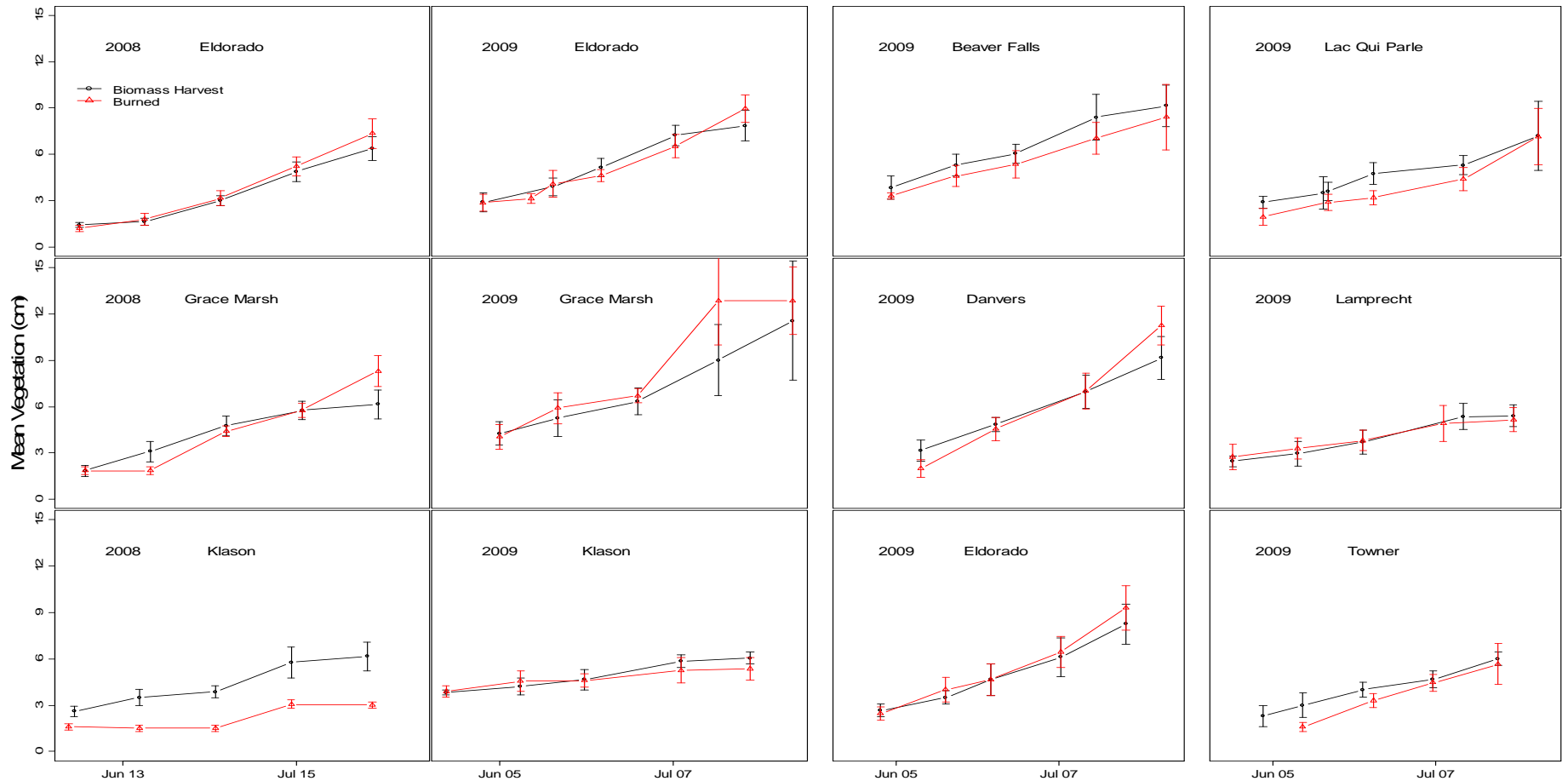
Figure 3. Comparison of mean vegetation height (dm) and 95% confidence intervals between 2 treatment subplots (a fall biomass harvest and a prescribed burn the following spring) within the same restored native grass field on 3 State Wildlife Management Areas in west-central Minnesota, in both summer 2008 and summer 2009 (leftmost two columns), and on 5 State Wildlife Management Areas and 1 Federal Waterfowl Production area in west-central Minnesota, in only summer 2009 (rightmost two columns).

Figure 4. Comparison of mean litter depth (dm) and 95% confidence intervals between 2 treatment subplots (a fall biomass harvest and a prescribed burn the following spring) within the same restored native grass field on 3 State Wildlife Management Areas in west-central Minnesota, in both summer 2008 and summer 2009 (leftmost two columns), and on 5 State Wildlife Management Areas and 1 Federal Waterfowl Production area in west-central Minnesota, in only summer 2009 (rightmost two columns).
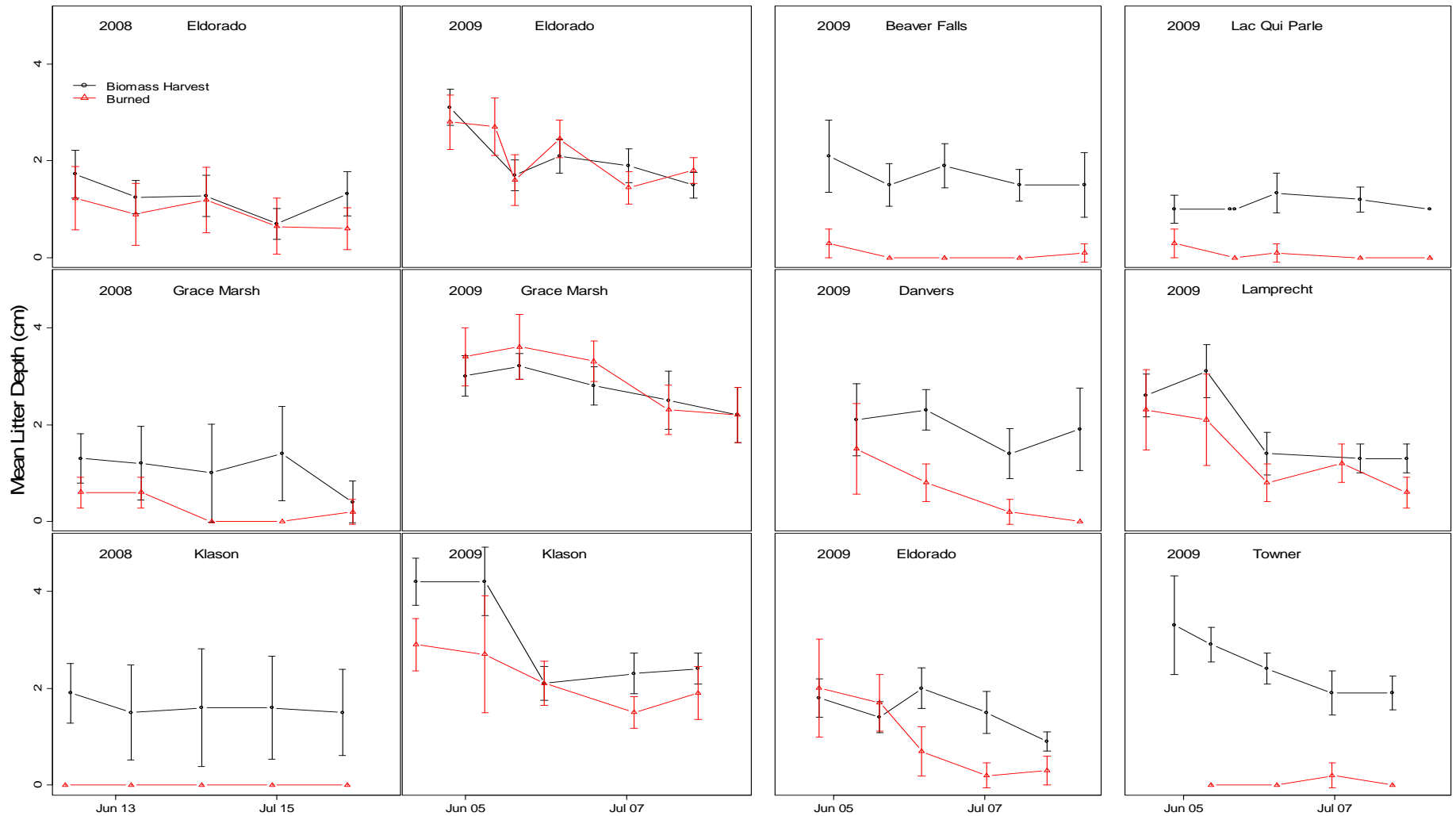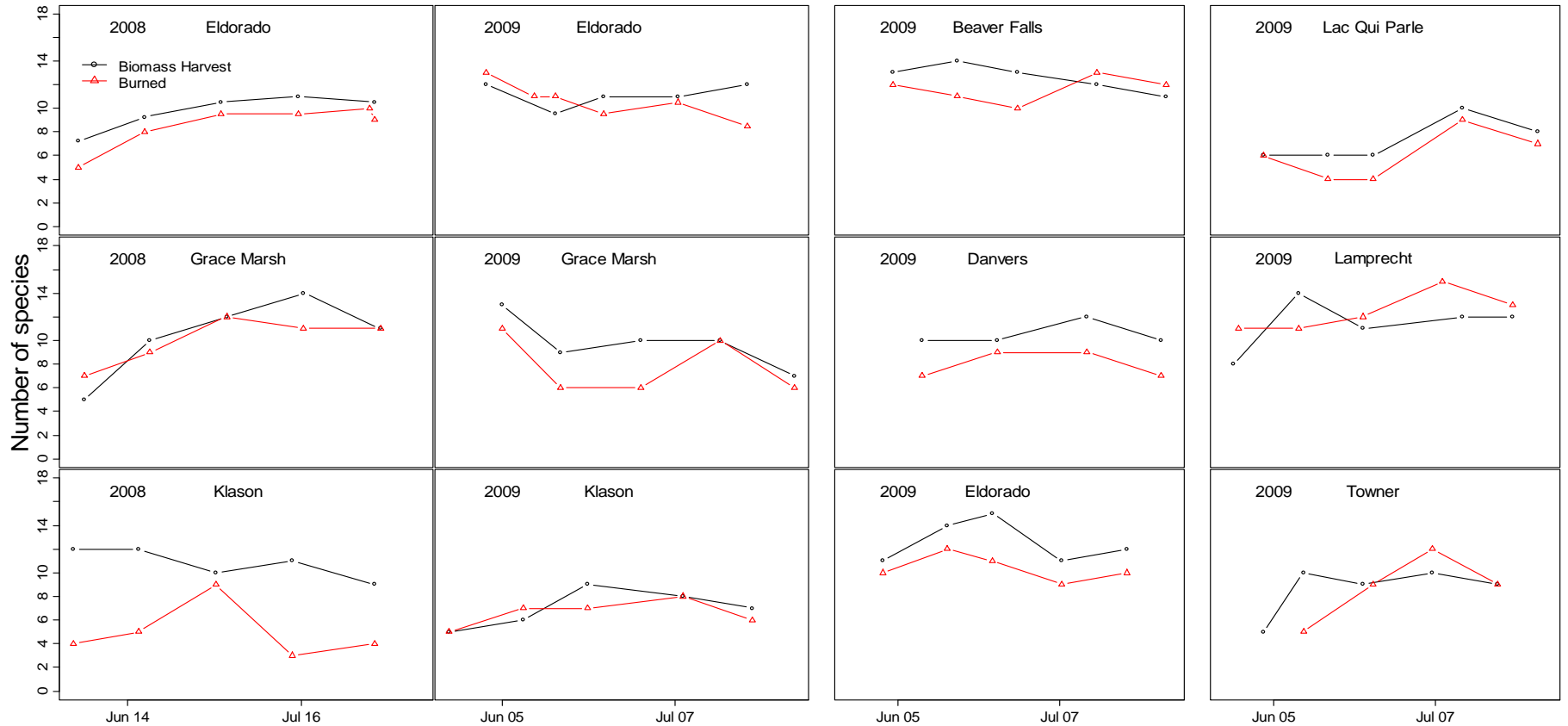
Figure 5. Comparison of mean number of plant species per transect between 2 treatment subplots (a fall biomass harvest and a prescribed burn the following spring) within the same restored native grass field on 3 State Wildlife Management Areas in west-central Minnesota, in both summer 2008 and summer 2009 (leftmost two columns), and on 5 State Wildlife Management Areas and 1 Federal Waterfowl Production area in west-central Minnesota, in only summer 2009 (rightmost two columns).

Figure 6. Comparison of the percent of native plant species per transect between 2 treatment subplots (a fall biomass harvest and a prescribed burn the following spring) within the same restored native grass field on 3 State Wildlife Management Areas in west-central Minnesota, in both summer 2008 and summer 2009 (leftmost two columns), and on 5 State Wildlife Management Areas and 1 Federal Waterfowl Production area in west-central Minnesota, in only summer 2009 (rightmost two columns).
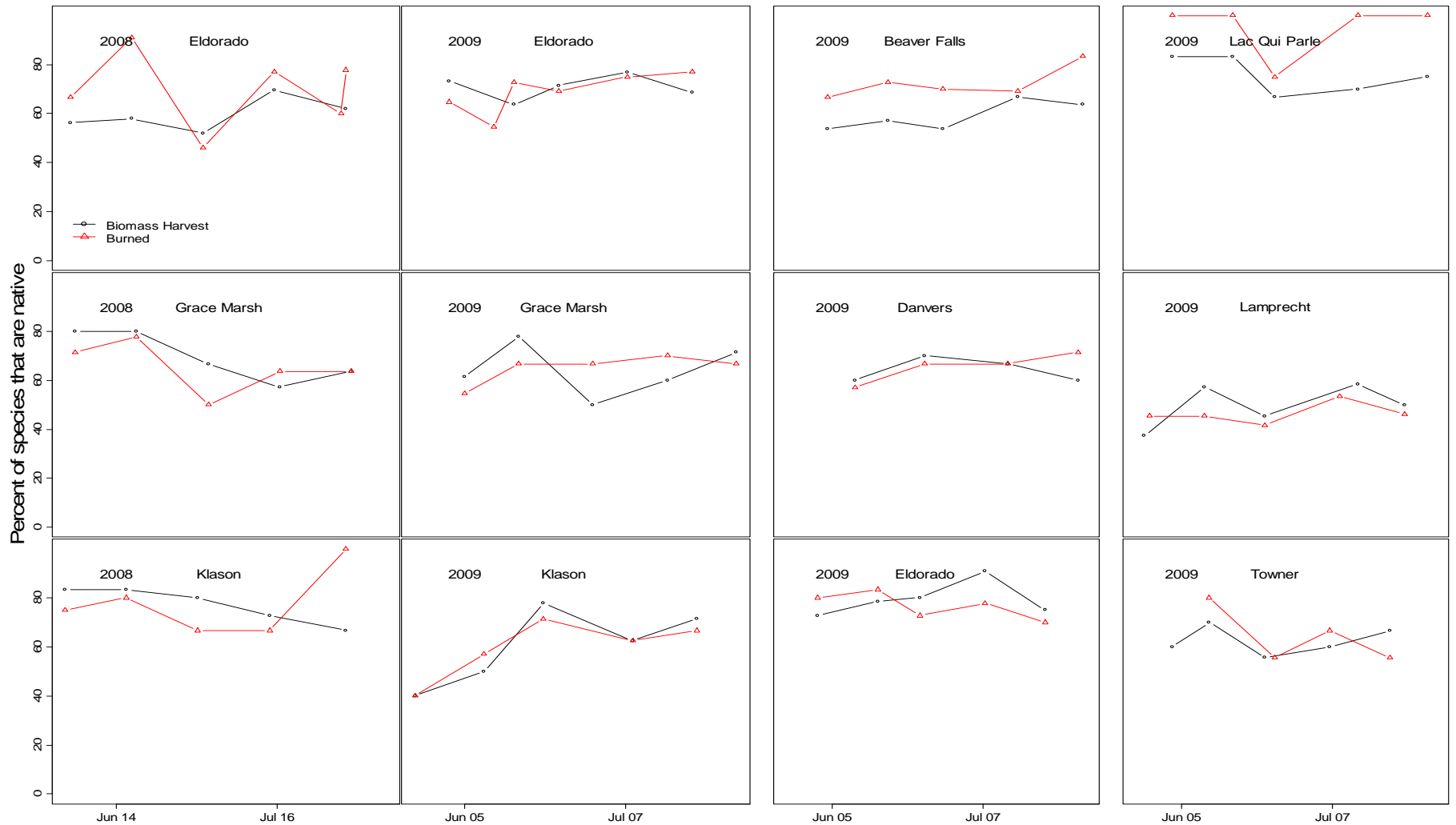
MOVEMENTS, SURVIVAL, AND REFUGE USE BY RING-NECKED DUCKS AFTER FLEDGING IN MINNESOTA

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

SUMMARY OF FINDINGS

The Minnesota Department of Natural Resources (MNDNR) is conducting a study that examines use and survival benefits of waterfowl refuges to locally produced ring-necked ducks (*Aythya collaris*). During 2007 – 2009, we captured and implanted 176 flightless ring-necked ducks with radiotransmitters. Ducklings were tracked weekly by aircraft and from telemetry receiving stations located on 14 waterfowl refuges. The distance between weekly locations averaged ~8 km in all years. Young ring-necked ducks used state and federal waterfowl refuges, but this use was not evenly distributed among refuges; 3 refuges received the majority of use and 1 refuge has yet to be used by marked birds. Refuge use was also higher during hunting season than prior to the season opening. Additional data collection in 2010 will be aimed at increasing sample sizes to address survival benefits of refuge use to young birds.

INTRODUCTION

Sizable populations of resident breeding ducks were recognized as a cornerstone to improving fall duck use in the MNDNR Fall Use Plan. Although breeding ring-necked duck populations have been increasing continentally, they may be declining 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.

Factors influencing resident populations of ring-necked ducks are poorly understood. Further, the Fall Use Plan 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 is unknown.

The intent of this 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 documented, and this study provides information for an important Minnesota species. Understanding movements and refuge use in the fall may provide valuable insights into the distribution of refuges required to meet management objectives for ring-necked ducks in Minnesota.

OBJECTIVES

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

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[2] U.S. Fish and Wildlife Service, Rice Lake National Wildlife Refuge, McGregor, MN

STUDY AREA

The study area lies primarily in the Laurentian mixed forest province of Minnesota (Figure 1). This area is characterized by mixed coniferous and hardwood forest interspersed with lakes, many of which are dominated by wild rice (*Zizania palustris*). The study area is ~200 x 135 km in size and encompasses 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). Lakes we monitored with remote receiving stations in this study were not open to public hunting, thus providing "refuge" for ducks during the fall migration.

METHODS

Night-lighting techniques were employed to capture flightless ring-necked ducks during July and August in 2007 – 2009. Duckling age (Gollop and Marshall 1954) and sex were determined at capture. We implanted radiotransmitters dorsally and subcutaneously on class 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, US Geological Survey (USGS), Alaska Science Center, personal communication) 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, we discontinued marking with nasal saddles in 2008 and 2009.

By early September, radiotelemetry stations were established at each refuge as a means of quantifying refuge use. Receivers were programmed to scan each of the established frequencies periodically each hour, 24 hours/day. Data were downloaded weekly from data-loggers from mid-September through early November. Reference radiotransmitters 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 include the hunters' names and the dates and locations of harvest.

RESULTS

Capture and Tracking

We captured 52 ducklings with night-lighting techniques 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. Capture locations were distributed throughout the study area, but more ducklings were captured on the western half of the study area in all years (31 in 2007, 32 in 2008, and 46 in 2009 in western counties compared to 21, 24, and 22 in each respective year in eastern counties, Table 2 and Figure 2).

The number of locations per bird varied from 1 to 14 (mean = 7.46, SE = 0.24) for the 176 marked birds. On average, 67% in 2007, 82% in 2008, and 82% in 2009 of birds were located weekly during aerial surveys beginning when the first bird was marked and continuing through early November. Success locating birds from aerial flights, however, was higher before hunting season (87% in 2007, 95% in 2008, 95% in 2009) than the week hunting opened in all years (66% in 2007, 83% in 2008, 83% in 2009). Success locating birds also declined as birds began moving more in preparation for migration.

Average weekly movements tended to increase as the fall progressed until mid to late October when birds started leaving the study area. For the tracking period, average weekly movements were 8.5 $\pm$ 1.9 km (mean $\pm$ SE) in 2007, 8.3 $\pm$ 2.1 km in 2008 and 7.2 $\pm$ 1.8 km in 2009, but average weekly movements prior to the start of hunting, after birds started moving (6.4 $\pm$ 1.1 km in 2007, 6.8 $\pm$ 1.6 km in 2008, 7.1 $\pm$ 1.7 km in 2009) were shorter than after hunting season opened (14.5 $\pm$ 3.0 km in 2007, 16.6 $\pm$ 3.5 km in 2008, 14.4 $\pm$ 2.4 km in 2009) in all years. All but 3 birds left their natal lake before hunting opened over the 3 year period.

Mortalities

In 2007, 15 radiomarked birds (n = 52) *were* known to have died by the end of the monitoring period (8 Nov); 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 Sept). 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*) or other raptors (3). Six transmitters retrieved from open water in 2007 were thought to have dehisced; thus the fate of these birds was unknown. 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).

In 2008, 25 radiomarked birds (n = 56) were known to have died by the end of the monitoring period (Nov 18); 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 (Oct 4 and 5). Radioed birds were either depredated or scavenged by mink, raccoon (*Procyon lotor*) and other mammals (5), raptors (1), and 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. Five radios were thought to have dehisced in 2008; and 2 of the birds which lost their radios were subsequently harvested (1 in 2008 in Oklahoma and 1 in 2009 in Cuba). Four additional birds were harvested after the monitoring period ended; 3 were harvested during the 2008 hunting season (2 in Louisiana, and 1 in South Carolina), and 1 was harvested during 2009 (Minnesota).

In 2009, 29 radiomarked birds (n = 68) were known to have died by the end of the monitoring period (Nov 9); 6 birds were shot by hunters (all in Minnesota), 12 were depredated, 10 died of unknown causes, and 1 may have died as a result of surgery. Two of the 6 harvested birds were shot during the youth opener (Sept 19) and 1 was shot during the first 2 days of the waterfowl hunting season (Oct 3 and 4). Radioed birds were either depredated or scavenged by mink, river otter (*Lontra canadensis*) and other mammals (9), raptors (1), and unknown sources (2). Four transmitters were thought to have dehisced in 2009, and the fate of these individuals was unknown. At the time of production of this document, 5 additional birds were harvested after the monitoring period ended during the 2009 hunting season (1 each in Alabama, Florida, Illinois, Missouri, and Texas).

Possible losses to predation prior to hunting season (7 in 2007, 12 in 2008, 21 in 2009) were similar or slightly higher than those during hunting (3 in 2007, 5 in 2008, 1 in 2009). Depredation earlier in the study period was expected to be higher, because during the first few weeks after marking, many ducklings are incapable of flight and more susceptible to predation. During hunting, some of the birds that appeared to have been depredated may have been wounded by hunters and later scavenged by predators. We x-rayed 2 birds that were recovered during the hunting season and found definitive shot pellets in 1 bird.

Refuge Use

Overall, 44 (25%) birds were documented at refuges based on aerial surveys and tower detections (17 in 2007, 11 in 2008, and 16 in 2009, Table 1). Although refuges were used before hunting season, use by radiomarked birds increased markedly with the onset of hunting (Figure 3). Numbers of ducks using refuges prior to hunting was less than during the hunting season. However, many birds were not capable of flight the first few weeks after capture.

Most 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 Mud Goose (15), Drumbeater (12), and Tamarac NWR (10, Table 1). Rice Lake NWR has never been used by radiomarked ducklings, but this refuge was outside the capture area, and we expected use of this refuge by radiomarked birds to be less than for refuges located within the capture area. Most birds visited only 1 refuge (29 of 44 birds); however, a number of birds used more than 1 refuge during the fall period (Table 3).

Although use of individual refuges varied each year, a number of refuges were used every year: Mud Goose, Drumbeater, Tamarac NWR, and Rice Pond. In 2007, refuge use was documented for 17 radiomarked birds from both aerial and tower data. Six refuges were used by marked birds, but the most heavily used refuges based on number of birds located there were Mud Goose (6), Tamarac NWR (6), Fiske and Blue Rock (4), and Drumbeater (3). Several state refuges also received no documented use by radiomarked birds in 2007 (Table 1). A similar pattern was observed in 2008 with 11 radiomarked birds using 8 refuges. The most heavily used refuge was Mud Goose (6 birds; Table 1). In 2009, refuge use was documented for 16 radiomarked birds at 11 refuges during the fall migration. The most heavily used refuge in 2009 was Drumbeater (7 birds). The tower data are challenging to interpret and the number of birds detected by towers is subject to revision as we continue to analyze the data.

From the tower data, we also determined diurnal versus nocturnal use. Refuges could also be classified as day use (7:00 am and 6:00 pm), night use (7:00 pm to 6:00 am), and 24-hour use, based on the majority of observations occurring at various times during a 24 hour period (Table 4, Figure 4).

DISCUSSION

One more field season is anticipated. Methods in 2010 will be similar to those of 2008 and 2009. More formal analyses will be conducted at the conclusion of the study. Results and discussion of these analyses will be included in future Summaries of Wildlife Research Findings.

ACKNOWLEDGMENTS

LITERATURE CITED

Gollop, J. B. and W. H. Marshall. 1954. A guide to aging duck broods in the field. Unpublished report. Mississippi Flyway Council.

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

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

Table 1. National Wildlife Refuges and Minnesota State Refuges included in the study area, approximate location of the refuges, peak numbers of ring-necked ducks during fall migration, number of recording telemetry stations established on each refuge, and the use of each refuge by radiomarked post-fledging ring-necked ducks during 2007 – 2009. Note that the tower data are challenging to interpret and the number of birds detected by towers is subject to revision as we continue to analyze the data.

Refuge	Location	~Peak numbers	Stations	No. radiomarked birds using Refuge		
				2007	2008	2009
National Wildlife Refuge						
Rice Lake	5 mi SSW of McGregor	120,000	4	0	0	0
Tamarac	16 mi NE Detroit Lakes	50,000	3	6	1	3
State Waterfowl Refuge/State Game Refuge						
Donkey Lake	6 mi SW Longville	350	1	1	0	1
Drumbeater Lake	2 mi N of Federal Dam	280,000	1	3	2	7
Fiske and Blue Rock Lakes	8 mi SE Northhome	40,000	1	4	0	0
Gimmer Lake	10 mi SE Blackduck	3,500	1	0	3	0
Hatties and Jim Lakes	13 mi SE Blackduck	0	1	0	0	1
Hole-in-Bog Lake	2 mi SW Bena	4,000	1	0	0	4
Mud Goose	4 mi SSW of Ballclub	4,000	1	6	6	3
Lower Pigeon Lake	4 mi S Squaw Lake	700	1	0	1	3
Pigeon River Flowage	6 mi S Squaw Lake	700	1	0	1	3
Preston Lakes	22 mi ENE of Bemidji	1,800	1	0	2	2
Round Lake	8 mi N Deer River	11,000	1	0	0	2
Rice Pond	9 mi E of Turtle River	32	1	2	2	2

Table 2. Ring-necked duckling captures per county in Minnesota during 2007 – 2009.

County	Captures		
	2007	2008	2009
Aitkin	1	0	2
Becker	6	1	4
Beltrami	17	7	17
Cass	9	10	7
Clearwater	5	15	13
Hubbard	3	7	7
Itasca	9	10	11
Koochiching	2	4	2
Polk	0	2	3
Wadena	0	0	2

Table 3. Number of ring-necked ducklings that used 1 or more refuges, Minnesota 2007 – 2009.

No. refuges visited	No. birds
1	29
2	8
3	4
4	2
5	0
6	1

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.

Day use	Night use	24-h2our use	Not used
Donkey	Pigeon River	Mud Goose	Rice Lake NWR
Drumbeater	Rice Pond	Round	Tamarac NWR - Chippewa
Fiske Blue Rocks		Tamarac NWR – Little Flat	
Gimmer			
Hatties and Jim			
Hole-in-Bog			
Lower Pigeon			
Preston Lakes			
Tamarac NWR – Flat			
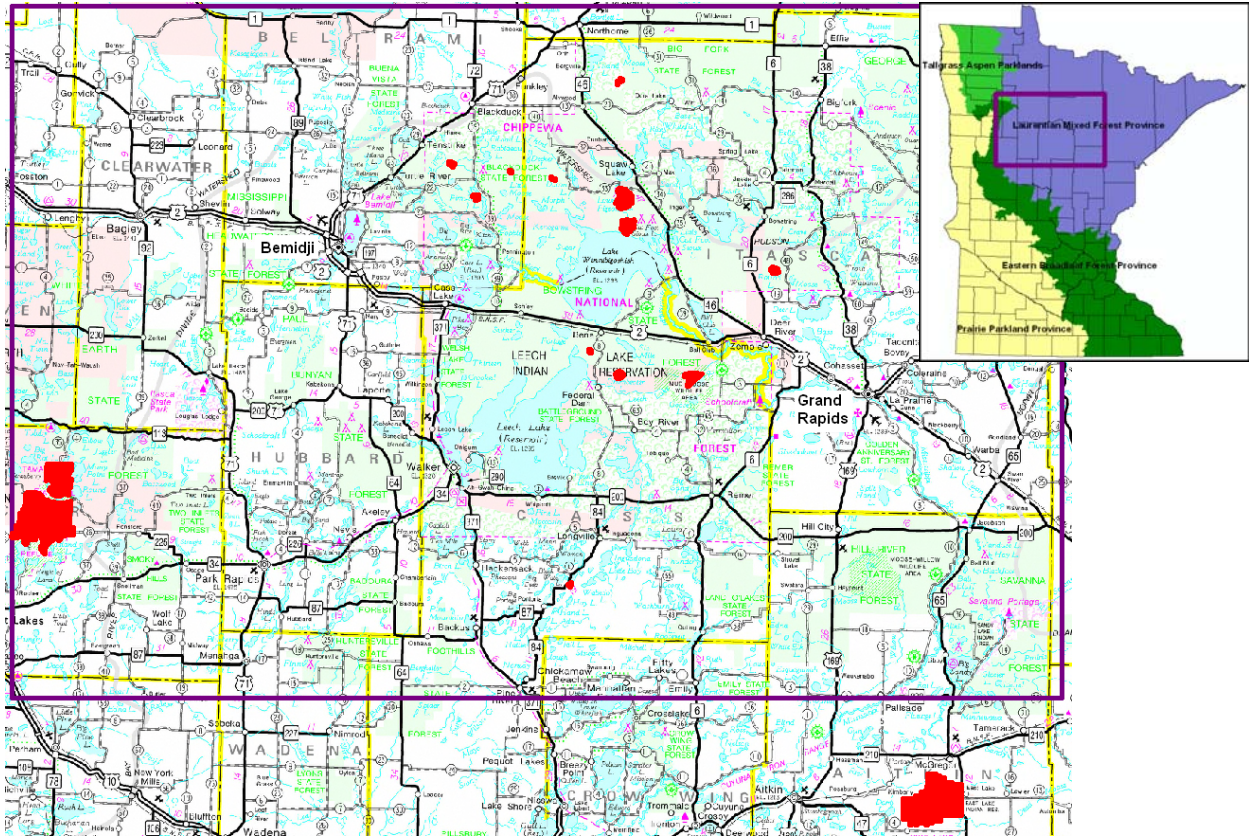
Figure 1.   Ring-necked duck study area in Minnesota during 2007 – 2009 with 12 state waterfowl/game refuges and 2 National Wildlife Refuges depicted in red.
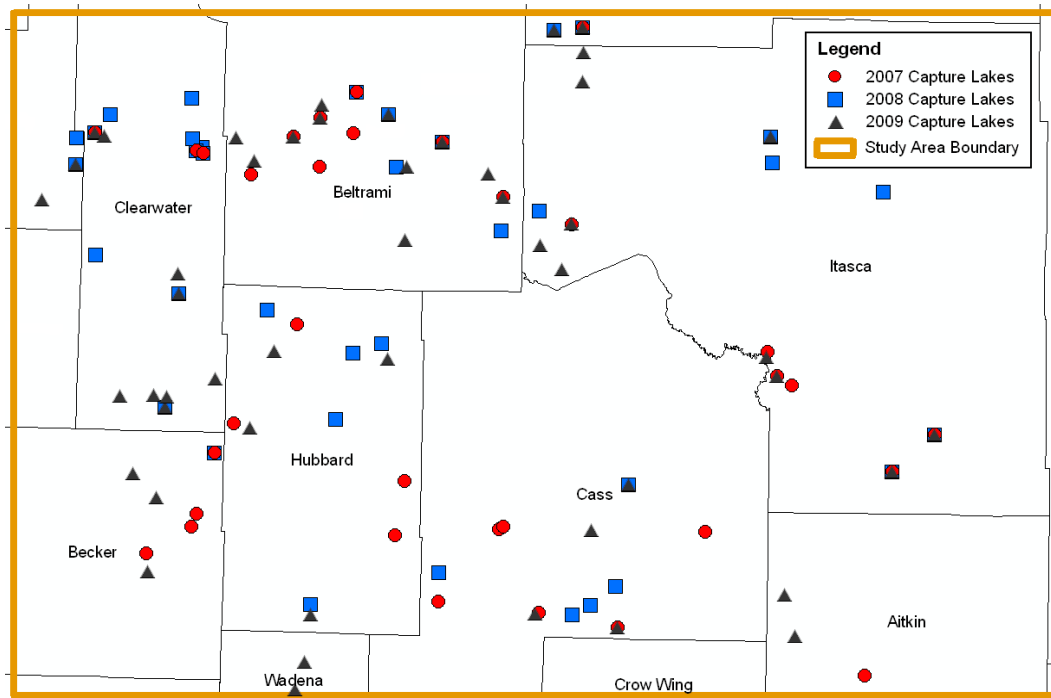
Figure 2. Capture locations for ring-necked duck ducklings in Minnesota during 2007 – 2009.
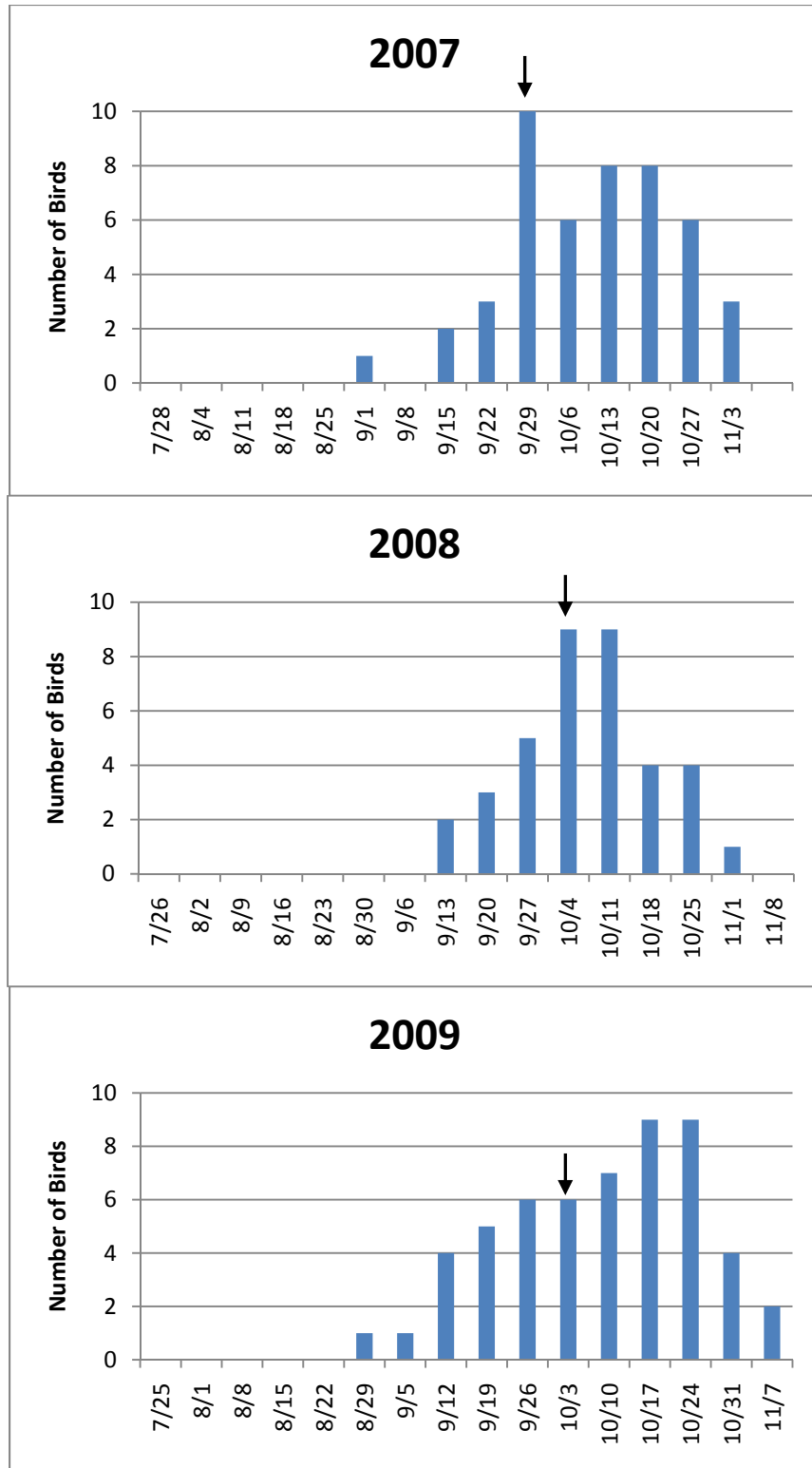
Figure 3. Weekly use of refuges by post-fledging ring-necked ducks before and during hunting season in 2007 – 2009 in Minnesota. Weeks are from Saturday through Friday with the Saturday date shown. Arrows indicate the week waterfowl hunting opened.
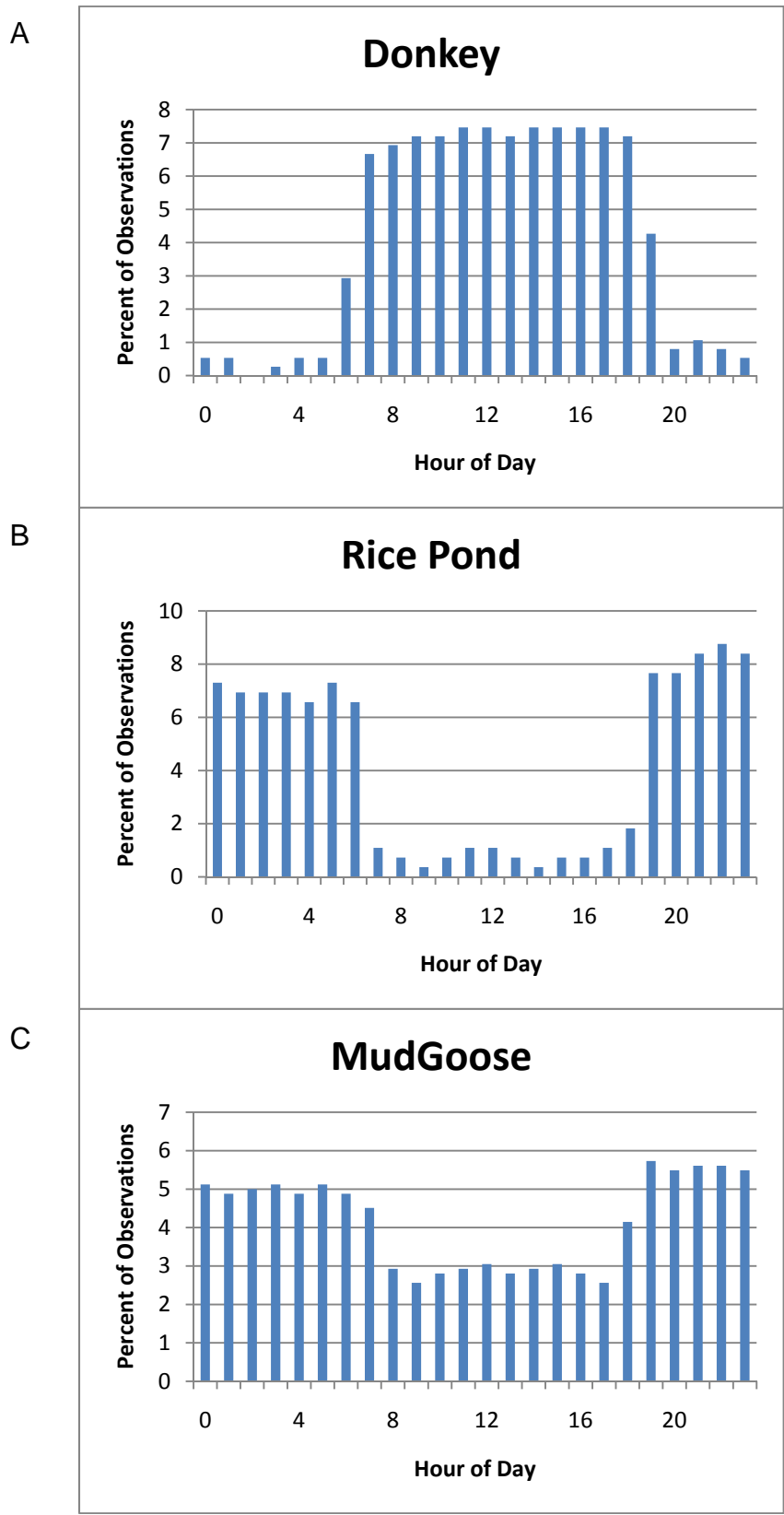
Figure 4. Example refuge use data to show the difference among day use (A), night use (B), and 24-hour use (C) refuges in Minnesota during 2007 - 2009.

REGIONAL COMPARISONS OF RELATIONSHIPS AMONG LANDSCAPE SETTING, AMBIENT NUTRIENTS, LAND USE, FISH COMMUNITIES, AND ECOLOGICAL CHARACTERISTICS OF SHALLOW LAKES - PRELIMINARY EFFORTS – 2009

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SUMMARY

Minnesota's shallow lakes provide numerous valuable ecosystem services and habitat for native species along with direct human benefits including clean water, recreational opportunity, and carbon sequestration. Unfortunately, water and habitat quality of Minnesota's shallow lakes have deteriorated dramatically during the past century. Conversion from native upland covers, widespread wetland drainage, and surface-water consolidation to facilitate agricultural and urban/residential development have been implicated as major causes for these changes. We are studying approximately 136 shallow lakes in 5 ecological regions of Minnesota to: (1) identify major factors leading to deterioration, (2) evaluate results of specific lake restoration approaches, including cost-effectiveness of various combinations of lake management strategies; and (3) assess the impacts of increased surface water connectivity on fish invasions and resulting habitat quality. Our efforts include extensive sampling of shallow lakes to identify direct and indirect causes of deterioration, evaluation of approximately eight lakes currently undergoing rehabilitation, and economic analyses to determine which restoration strategies are likely to produce the greatest improvements in water quality and other lake characteristics per unit cost. Ultimately, our results will allow municipalities, state, county, and local governments, and private organizations to identify cost-effective approaches for maintaining and restoring ecological integrity of shallow lakes throughout Minnesota. Special attention will be directed towards development of regionally-specific recommendations for sustainable lake management.

BACKGROUND

Minnesota has approximately 4,000 lakes characterized by mean depth $\leq$ 15 ft and surface area > 40 acres (Nicole Hansel-Welch, personal comm.) 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 40 of these lakes > 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

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quality, seasonal duck abundance and habitat use, and hunter satisfaction, 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. At the same time, restoration 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. In general, managers receive little technical guidance useful 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, biological interactions, especially wetland fish communities, also exert major structuring influences on communities and characteristics of prairie wetlands and 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 manipulations 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 one of two 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. 2009a).

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 often 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 (Tonn and Magnuson 1982; Rahel 1984). 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. 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 environmental characteristics of shallow lakes themselves.

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 (2005-06) 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. In revision). 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 prep.), and fish variables were always included in best models for predicting amphibian site occupancy and abundance in shallow lakes (Herwig et al. In Prep.). 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. 2009a). 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 region. We are also comparing relative influences of within-site and landscape-scale characteristics on shallow lake invertebrate communities. Contributions from Sean Vaughn (Division of Waters, MNDNR) and Robert Wright (Section of Wildlife, MNDNR) provided new spatial analysis tools (delineating lake watershed boundaries, spatial analysis, etc.) that were not only critical for the recently-completed study, but will have direct application to questions and hypotheses posed in this current effort.

Major goals of our previous study were 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 influence ecological characteristics of shallow lakes. An overarching finding of the prior work was that regional differences typically 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 observations from MNDNR shallow lakes program staff indicating that baseline characteristics of shallow lakes differ dramatically across regions of the state (Nicole Hansel-Welch, pers. comm.). 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, and 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 and/or decreased macrophytes will often induce regime shifts in prairie lakes, and these changes probably portend shifts to turbid-water states. 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- vs. turbid-water states.

RESEARCH APPROACHES

Extensive Lakes

We are currently gathering data from, and characterizing watershed features of, 128 shallow lakes (hereafter Extensive lakes) from 6 regions of Minnesota. Lakes will be sampled once each July in 2009-11 to assess general ecological features and determine whether basins exhibit characteristics of clear- or turbid-water regimes. Lake watershed characteristics associated with each study lake will also be determined by creating and applying numerous lake watershed variables via GIS technology and interpretation of aerial photography. Resulting data will be used to develop models to identify combinations of variables that explain most 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.

Intensive Lakes

During 2010-11 we will also evaluate responses of 8 shallow lakes (hereafter Intensive lakes) currently undergoing lake restoration treatments such as draw downs or fish community manipulation. Ecological characteristics of Intensive lakes will be sampled monthly from May-August each year, including all components measured in the 128 Extensive sites. Identical landscape-level analyses will be conducted on these areas to determine upland cover and surface-water connectivity in lake watersheds using 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.

Connectivity Emphasis

Ecological health of shallow lakes is a reflection of their upstream and downstream watersheds and the hydrologic connectivity within those flow networks. Increased surface water connectivity due to drainage, ditching, road construction, and other anthropogenic activities is known to increase the transfer of organisms, especially undesirable fishes, among shallow lakes in Minnesota. Such connectivity probably also provides major pathways for the spread of invasive species, which threaten native communities.

We will identify, delineate and digitize unmapped natural and human-induced water conveyance features that constitute present-day surface water connectivity. Using data from the Extensive (128) and Intensive lakes (8), we propose to document water quality, biodiversity, habitat characteristics, and measure lake responses to various surface water connectivity scenarios. This will allow the development of models useful for assessing probable results from increased surface water connectivity within the watersheds. We believe this will provide useful

data and guidance for natural resource managers who frequently evaluate requests for landscape modifications that increase surface-water connectivity, runoff and channelized flow.

Economic Analysis

An economic analysis will be conducted using the empirical data from all study lakes in identifying water quality improvements (such as cost per unit of algae reduced [µg/L chlorophyll *a*]) resulting from various application of various management options being utilized or considered within Minnesota. We plan to quantify the costs of applying various combinations of upland vegetation restoration (conversion of agriculture to grass) and in-lake habitat enhancements (fish removal, installation of barriers, etc.) 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 regionally 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-run costs (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.

Working Hypotheses

Our overall, general working hypothesis is that 6 fundamental "drivers" are ultimately responsible for most of the variation in ecosystem characteristics of Minnesota 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 (fish, plant and invertebrate communities, water quality, carbon cycling, etc.), along with lake responses to rehabilitation efforts, requires understanding the influence of these drivers, as well as synergistic combinations of influences arising from two 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 are also a function of our main drivers such that within-lake interactions will contribute to spatial and temporal patterns that can be predicted from these influences.

We believe it is also especially important to test further hypotheses regarding stability regimes in shallow lakes. Previous work (Hanson and Butler 1994) suggests that shallow lakes in MN 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. 2009a). 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

will provide guidelines useful for maximizing future lake restoration and management decisions, including suggestions as to how more cost-effective approaches vary across the state. Finally, all resulting data will be used to assess extent to which surface connectivity among surface waters influences ecological characteristics of shallow study lakes.

METHODS

Study Areas

A key goal of our study is to increase understanding of spatial patterns of shallow lake characteristics across Minnesota. Shallow lakes here occur across a wide range of lake watershed characteristics (agriculture and urban land uses, native cover types, etc.), phosphorus concentrations, and water transparency gradients. We used an aquatic ecoregion approach for characterizing shallow lake features (sensu Heiskary et al. 1987). We used classifications based on Omerik's (1987) Level III ecoregion delineations denoting areas of general similarity in the type, quality, and quantity of environmental resources. Under this approach we established a study area (or "study landscape"), each containing a cluster of study sites, within each of the 5 ecoregions that collectively encompass the vast majority of lakes and wetlands in Minnesota: Northern Minnesota Wetlands (NMW), Northern Lakes and Forests (NLF), Northern Glaciated Plains (NGP), Western Corn Belt Plains (WCP), and North Central Hardwood Forests (CHF). As previously mentioned, there are large gradients in in-lake phosphorus (P) and nitrogen concentrations across these ecoregions. For example, in a survey of 1,062 lakes, Heiskary et al. (1987) found median P concentrations of 23 ppb (NLF), 50 ppb (CHF), 121 ppb (WCP), and 177 ppb (NGP). No information was available for NMW, but we expect lower P concentrations here, perhaps intermediate between NLF and CHF. Cover types also vary widely, ranging from heavily forested, with some marshlands (NLF) to nearly level marsh, containing both boreal vegetation and expansive swamps (NMW), to principally cropland agriculture (WCP & NGP), to a mosaic of cover types, including forests, wetlands and lakes, cropland agriculture, pasture, grasses, and urban development (CHF) (Omerik 1987).

Our study focuses on 6 landscape areas distributed across 5 aquatic ecoregions within Minnesota as follows: (1) the NMW study area (hereafter "Red Lake") is located within the boundaries of the Red Lake Indian Reservation in far northern Clearwater and west-central Beltrami counties, (2) a NLF study area (hereafter "Itasca") is positioned within and around Itasca State Park in south-eastern Clearwater County, (3) a second NLF study area with sites located in western portions of the Chippewa National Forest in far western Itasca County (hereafter "Chippewa"), (4) the NGP study landscape (hereafter "Elbow Lake") located in the southern portions of Grant County, extending into the northern and western margins of Stevens and Douglas counties, respectively (we have a long time series here, dating back the mid 1990's), (5) the WCP study area (hereafter "Windom") centered around Windom, MN, and thus roughly split between Cottonwood and Jackson counties, (6) the CHF study landscape (hereafter "Twin Cities") located in the Hennepin-Carver county metro area (Figure 1).

Our study landscapes are also positioned in several different major river watersheds. In some cases, study areas fall within two or more drainages. For example, Red Lake is entirely within the Red River drainage, Itasca is entirely within the Upper Mississippi River drainage, but Twin Cities is within both the Upper Mississippi River and Minnesota River drainages. Similarly, Windom is within the Minnesota River and Lower Mississippi River drainages, and Elbow Lake is within the Red River and Minnesota River drainages.

Individual Study Sites

Within each study landscape, we are studying up to 24 shallow lakes, measuring fish assemblages, wetland characteristics, and surrounding landscape attributes. Study lakes were distributed across both public and private ownerships, and all lakes are of semipermanent or

permanent (type IV or V) duration of flooding (Shaw and Fredine 1956; Stewart and Kantrud 1971). Within these broad classifications, shallow lakes span a range of values of surface area, depth, and adjacent upland cover types.

General Data Collection Approaches

Development of land use and lake watershed variables using GIS and air photo interpretation

GIS data layers will be used to derive metrics that characterize features of the landscape associated with each study site, including proportions of the dominant cover types at the watershed-scale as well as upstream and downstream hydrological connectivity. Lake watershed boundaries will be delineated for each site using the delineation methods of Sean Vaughn (2009). Existing land cover layers (perhaps MN GAP or land cover layers developed by the University of Minnesota's Remote Sensing and Geospatial Laboratory: http://land.umn.edu) will then be overlaid and summarized for the individual lake watersheds. Data summaries will be developed as needed and will primarily include connectivity attributes and watershed characteristics (e.g. surface area of different cover types, inter-lake surface connection distances, watershed:lake area ratios).

Landscape/watershed connectivity analyses may include but are not limited to the following: (1) presence of upstream/downstream connections to surface waters capable of supporting fish populations; (2) modeled upstream/downstream connections of surface water from digital elevation models (DEM) to surface waters capable of supporting fish populations; (3) distances to represent "as the fish swims" to surface waters capable of supporting fish populations (horizontal and vertical dimensions); and (4) rank variable for type and degree of connectivity to other surface waters (also a potential proxy for geomorphic setting).

Fish assemblages

Fish species composition and relative abundance (biomass per unit effort) will be determined using a combination of gears deployed overnight. All fish sampling will be done during July and August each year. Three mini-fyke nets (6.5 mm bar mesh with 4 hoops, 1 throat, 7.62 m lead, and a 0.69 X 0.99 m rectangular frame opening into the trap) will be set overnight in the littoral zone of each lake. One experimental gill net (61.0 m multifilament net with 19, 25, 32, 38, and 51-mm bar meshes) will be set along the deepest depth contour available in lakes less than 2 m deep or along a 2 m contour in lakes with sufficient depth. The protocol outlined above has been shown to be effective in sampling fish assemblages in shallow lakes in Minnesota (Herwig et al. 2010) as well as small lakes from other regions (Tonn and Magnuson 1982; Rahel 1984; Jackson and Harvey 1989; Robinson and Tonn 1989). This should enable us to capture both small- and large-bodied fish, and species from all of the major trophic guilds (e.g., plankivores, benthivores, piscivores) potentially present in the study wetlands. All fish sampled will be sorted by species, rated (counts per unit weight), and weighed in bulk. Fish data will likely be quantified as the summed total biomass of each species collected in all four nets. Voucher specimens will be collected and returned to the laboratory for identification when field identification cannot be made.

Aquatic invertebrates

Zooplankton will be sampled once per year in July concurrent with fish sampling by collecting two replicate vertical column samples (Swanson 1978) at 5 locations in each wetland. Estimates will be made of density and taxon richness. Relative abundance of macroinvertebrates will be sampled concurrent with other sampling in July using sweep net samples (Murkin et al. 1983) at 0.75m depth at 5 randomly selected locations in each lake. Abundance and taxon richness of macroinvertebrates will be measured.

Nutrients, specific conductance, light attenuation, and phytoplankton

Surface (dip) water samples will be taken from the center of each lake once during July concurrent with other sampling. Samples will be frozen and transported to the University of St. Thomas for analysis of chlorophyll *a*, total nitrogen, total phosphorus and total dissolved phosphorus. Turbidity will be measured in the field with a portable nephelometer. Phytoplankton biomass will be estimated from chlorophyll *a* (Strickland and Parsons 1972). Collection of samples for chlorophyll *a* simultaneously with measurement of turbidity will allow assessment of the contribution of phytoplankton to turbidity, and ultimately to light attenuation.

Submerged macrophytes

Abundance of submerged macrophytes and *Chara* spp. will be assessed using modified techniques of Jessen and Lound (1962), and Deppe and Lathrop (1992). In each lake, submerged macrophytes will be sampled at 15 stations located equidistant along four transects running the width of each basin in July or August of each year. Two throws of a weighted plant rake will be made at each station, and dragged along 3 m of lake bottom. Plants collected on the first throw will be weighed (all taxa combined) and frequency of occurrence (1 = sampled on one throw, or 2 = sampled on both throws) will be recorded for each plant species sampled. Plant data will be summarized as mass and frequency of occurrence (all taxa combined) summed across the total number of throws used for each metric.

Earthworms

We will study earthworm effects on shallow lakes in only one region due to lack of facilities and personnel for examining this phenomenon elsewhere. Earthworms will be collected from uplands within 50 m of all study lakes in our Itasca core area. Near each lake, 10 35 cm x 35 cm areas will be cleared of surface duff and flooded with a saturated solution of mustard (after methods of Laurence and Bowers 2002). Extracted worms will be collected, preserved in 75% ethanol, and identified according to an ecological classification system of Hale et al. (2005). Data will be used to develop a relative abundance estimate for earthworms in catchment areas immediately adjacent to study lakes.

We will correlate earthworm abundance and ecological classifications with the nutrient concentrations, chlorophyll *a*, and other water quality characteristics in adjacent study basins. Earthworm collections will be restricted to lakes within our Itasca core area due to relatively uniform forest composition in this ecological region (enabling earthworm effects to be assessed independent of other factors) and because related measurements require laboratory facilities available at the University of Minnesota field station in Itasca State Park. It is also important to note that students (using non-project funds provided to J. Cotner) will be collecting ancillary data on forest characteristics and soils in this region.

Intensive sampling

In consultation with Minnesota Ducks Unlimited staff, we recently identified 8 case study lakes to evaluate effectiveness of restoration strategies typically used by state, federal, and private organizations working on shallow lake management. We plan to assess effectiveness of various combinations of lake rehabilitation approaches including installation of fish barriers, water level draw downs, rotenone, and perhaps other measures commonly used by lake managers in Minnesota. Study sites include lakes that have been restored by drawdown and re-flooding during the past 2-4 years.

We will measure the effectiveness of the various management activities by assessing changes in ecosystem features following the specific manipulation to the lake. Variables to be

assessed include water clarity, nutrient levels in the water column, and abundance and species composition of phytoplankton, submerged macrophytes, zooplankton, macroinvertebrates, and fish. The relative improvement in each of these variables will be assessed using data from our larger Extensive study for lakes in the same ecoregion. For example, it is difficult to quantify the degree of "successfulness" following lake-drawdown when duration of improvements and responses of submerged plants cannot be predicted. However, the Extensive (128) lake study will help to quantify this change if lakes shift from turbid-to clear-water states following drawdown. Interpreting lake response in the context of natural regional variability should also facilitate assessment of success across ecoregions where lake features naturally vary.

Data Analysis

We anticipate applying a suite of analysis strategies to evaluate the various hypotheses outlined above. This is necessary because no single approach we are aware of allows for identification and measurement of multiple complex linkages discussed above. Our approach will include gradient analysis (ter Braak 1995; ter Braak and Smilauer 1998; McCune and Grace 2002), classification and regression tree techniques (Breiman et al. 1984, De'ath and Fabricus 2000), variance partitioning (Borchard et al. 1992; ter Braak and Wiertz 1994), mixed effect linear models (Littell et al. 2006), piecewise regression (Toms and Lesperance 2003), information-theoretic model selection techniques (Burnham and Anderson 1998; Anderson et al. 2000), and traditional parametric approaches (SLR, ANOVA) (Zar 1999). Collectively, our analyses are intended to provide evidence whether ecological features of study lakes differ in predictable ways (thus whether lakes can be grouped) and, if so, whether fish communities, landscape and lake watershed features, cover types, ambient nutrients, lake basin morphology, and climate and other regional patterns account for observed differences among groups. Analyses will likely include situations where data are pooled from all landscapes to ensure a considerable range of values in both predictor and response variables, and situations where analyses will be developed for each study landscape separately, especially if separate modeling improves predictive ability, or if region-specific prediction and models are required.

Synthesis and Expected Research Products

We will use data from 8 Intensive and 128 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.

PROGRESS TO DATE

During July 2009, we gathered data from 128 Extensive lakes in our 6 study regions (Figure 1). In each lake, we measured water transparency and lake depth, and collected surface-dip samples for water-column concentrations of phosphorus and nitrogen in major pools, and phytoplankton biomass (chlorophyll *a*). At the same time, we gathered samples of aquatic macroinvertebrates and zooplankton, and conducted surveys to estimate relative abundance of submerged aquatic plants. We also assessed presence and composition of fish communities in each lake. Sediment samples were gathered from selected lakes within boundaries of Itasca State Park, from lakes within our Chippewa core, and from Alexandria, Twin Cities, and Windom areas. Samples for determination of major nutrient and chlorophyll *a* concentrations are being processed using facilities available at the University of St. Thomas (St. Paul, MN). Similarly, samples of zooplankton (column samples) and macroinvertebrates are being enumerated in the lab during January - May 2010. Watershed delineations have been

mostly completed for the 128 Extensive lakes and we are in final stages of selecting 8 sites for intensive aspects described above. Presently, we are awaiting a final decision on a proposal submitted to Legislative Citizens Commission on Minnesota Resources (funding has been approved). During the past year, Dr. Kyle Zimmer (our study collaborator) received National Science Foundation support (funding started in July 2009) for studies of carbon burial in a subset of our study lakes and we expect to partner on some aspects of data gathering to facilitate transfer of study results and interpretation between projects (Zimmer et al. 2009b). Also, during 2010-2011, we plan more detailed measurements of groundwater contributions on a subset of lake sites to better assess extent to which geomorphic setting influences ecological characteristics of these sites (details in Bischof et al. 2010).

Although results of formal analyses are not yet available, several preliminary observations are noteworthy. Climate/precipitation gradients were strong in Minnesota in 2009 with a number of Twin Cities lakes in severe drawdown during July. In contrast, we observed normal to high water levels in Windom, and normal water levels in our other study areas. These patterns should have important influences on shallow lake characteristics, and will likely vary yearly. Contrary to our expectations, preliminary data indicate that turbid regimes (sensu Zimmer et al. 2009a) are possible for lakes within the NLF ecoregion in north-central Minnesota, at least one site was turbid in both the Itasca and Chippewa study areas. Shallow lakes in our Windom study area were not all turbid.

Trends in fish communities across study regions were also surprising. For example, our Itasca study area, especially lakes within Itasca State Park, had the highest prevalence of fishless sites of the 6 landscapes we studied. The Chippewa study area had the highest richness of planktivorous and piscivorous fish, and benthivorous fish richness was higher than Itasca, but similar to other study areas. The higher number of planktivorous and piscivorous fish species probably reflects the more widespread distribution of certain minnows (northern redbelly dace, central mudminnow, golden shiner), yellow perch, and northern pike within the Chippewa. Average biomass of planktivores and piscivores was also higher in the Chippewa than other study areas (Figure 2). Although carp are widely distributed and are often associated with poor water quality in shallow lakes, some lakes in the Windom area did not have carp. Fish communities and fish densities changed sharply in several of Elbow Lake sites compared to 2006 (when we last worked on these sites), and many of these lakes switched from turbid to clear regimes, presumably due to lower fish densities, perhaps following winterkill.

During 2010, we expect to continue comprehensive food web, sediment, and water chemistry sampling of 128 shallow lakes (Extensive sites). At the same time, we plan to begin sampling the 8 case study lakes (Intensive sites) and to start developing datasets that will allow us to assess shallow lake responses to various lake rehabilitation efforts. We also plan to begin developing conceptual models to assess cost-effectiveness of lake rehabilitation activities We expect that final review of watershed delineations for our shallow lake study sites, and development of land cover summaries will be completed within the next year.

LITERATURE CITED

Anderson, D.R., Burnham, K.P., and Thompson, W.L. 2000. Null hypothesis testing: problems, prevalence, and an alternative. Journal of Wildlife Management 64: 912-923.

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

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

Bischoff, S., M. A. Hanson, B. R. Herwig, and T. J. Kroeger. 2010. Identifying proxy variables for assessing influences of lake geomorphic setting: A research prospectus. Submitted in April 2010 to Section of Wildlife, Minnesota DNR, St. Paul, MN.

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

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

Breiman, L., Friedman, J.H., Olshen, R.A., and Stone, C.J. 1984. Classification and Regression Trees. Chapman & Hall (Wadsworth, Inc.), New York, NY.

Burnham, K.P., and Anderson, D.R. 1998. Model selection and inference: a practical information-theoretic approach. Springer-Verlag, New York, NY.

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

De'ath, G., and Fabricius, K.E. 2000. Classification and regression tree: a powerful yet simple technique for ecological data analysis. Ecology 81(11): 3178-3192.

Deppe, E.R., and Lathrop, R.C. 1992. A comparison of two rake sampling techniques from sampling aquatic macrophytes, Wisconsin Bureau of Research Findings 32, Madison.

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

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

Friederichs, S., K.D. Zimmer, B.R. Herwig, and M.A. Hanson. In Review. Top-down and bottom-up influences on shallow lake food webs. Submitted to Oikos.

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

Hale, C. M., L. E. Frelich and P.B. Reich. 2005. Exotic European Earthworm Invasion Dynamics in Northern Hardwood Forests of Minnesota, USA. Ecological Applications 15: 848-860.

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

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

Heiskary, S.A., Wilson, C.B., and Larsen, D.P. 1987. Analysis of regional patterns in lake water quality: using ecoregions for lake management in Minnesota. Lake and Reservoir Management 3: 337-344.

Herwig, B.R., Hanson, M.A., Reed, J.R., Parsons, B.G., Potthoff, A.J., Ward, M.C., Zimmer, K.D., Butler, M.G., Willis, D.W., and Snook, V.A. 2004. Walleye stocking as a tool to

suppress fathead minnows and improve habitat quality in semipermanent and permanent wetlands in the prairie pothole region of Minnesota. Minnesota Department of Natural Resources, Division of Fisheries and Wildlife, Special Publication 159, St. Paul, MN.

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

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

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

Jackson, D.A., and Harvey, H.H. 1989. Biogeographic associations in fish assemblages: local vs. regional processes. Ecology 70: 1472-1484.

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

Jessen, R., and Lound, R. 1962. An evaluation of a survey technique for submerged aquatic plants, Minnesota Department of Conservation Game Investigational Report 6, St. Paul.

Johnson, D.H. 1999. The insignificance of significance testing. Journal of Wildlife Management 63: 763-772.

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

Laurence, A. P. and M. A. Bowers. 2002. A test of the 'hot' mustard extraction method of sampling earthworms. Soil Biology and Biochemisty, 34: 549-552.

Littell, R.C., Milliken, G.A., Stroup, W.W., and Wolfinger, R.D. 1996. SAS Systems for Mixed Models. SAS Institute Inc., Cary, North Carolina, USA.

McCune, B., and Grace, J.B. 2002. Analysis of Ecological Communities. MJM Software Design, Gleneden Beach, OR.

Murkin, H.R., Abbott, P.G., and Kadlec, J.A. 1983. A comparison of activity traps and sweep nets for sampling nektonic invertebrates in wetlands. Freshwater Invertebrate Biology 2: 99-106.

Omernik, J.M. 1987. Ecoregions of the conterminous United States. Map (scale 1:7,500,000). Annals of the Association of American Geographers 77: 118-125.

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

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

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

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

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

Robinson, C.L.K., and Tonn, W.M. 1989. Influence of environmental factors and piscivory in structuring fish assemblages of small Alberta lakes. Canadian Journal of Fisheries and Aquatic Sciences 46: 81-89.

Scheffer, M. 1998. Ecology of Shallow Lakes. Champman and Hall, London, UK.

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

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

Shaw, S.P., and Fredine, C.G. 1956. Wetlands of the United States, U.S. Fish Wildlife Service Circular 39, Washington, D.C.

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

Stewart, R.E., and Kantrud, H.A. 1971. Classification of natural ponds and lakes in the glaciated prairie region, Bureau of Sport Fisheries and Wildlife Resources Publication No. 92, Washington, D. C.

Strickland, J., and Parsons, T.R. 1972. A practical handbook of seawater analysis. Bull. Fish. Res. Bd. Can. 167.

Swanson, G.H. 1978. A plankton sampling device for shallow wetlands. Journal of Wildlife Management 42: 670-672.

ter Braak, C.J. 1995. Ordination. *In* Data Analysis in Community and Landscape Ecology. *Edited by* R.H. Jongman, C.J. ter Braak and O.F. van Tongeren. Cambridge University Press, Cambridge, U.K.

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

ter Braak, C.J., and Wiertz, J. 1994. On the statistical analysis of vegetation change: a wetland affected by water extraction and soil acidification. Journal of Vegetation Science 5: 361-372.

Toms, J.D., and Lesperance, M.L. 2003. Piecewise regression: a tool for identifying ecological thresholds. Ecology 84: 2034-2041.

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

Vaughn, S.R., 2009. DNR Watershed Delineation: *Project History, Methodology, Terminology & Data Attribution*. Minnesota Department of Natural Resources - Waters.

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

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

Zar, J.H. 1999. Biostatistical Analyses. Fourth Edition. Prentice Hall, Upper Saddle River, New Jersey, USA.

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

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

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

Zimmer, K.D., Hanson, M.A., and Butler, M.G. 2003. Relationships among nutrients, phytoplankton, macrophytes, and fish in prairie wetlands. Canadian Journal of Fisheries and Aquatic Sciences 60: 721-730.

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

Zimmer, K.D., J. Cotner, M. Edlund, and K. Theissen. 2009b. Organic carbon burial in shallow lakes. Submitted to NSF Biological Sciences Program, Division of Environmental Biology, Ecosystem Science, Washington, DC. (Awarded 1 July 2009; $1,227,272)
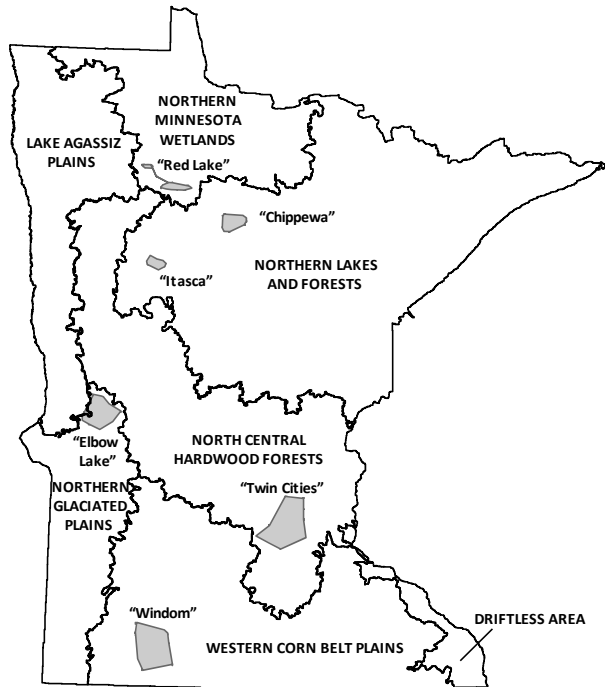
Figure 1. Map showing locations of study landscapes (shaded gray) in relationship to Minnesota's aquatic ecoregions (thick black lines).
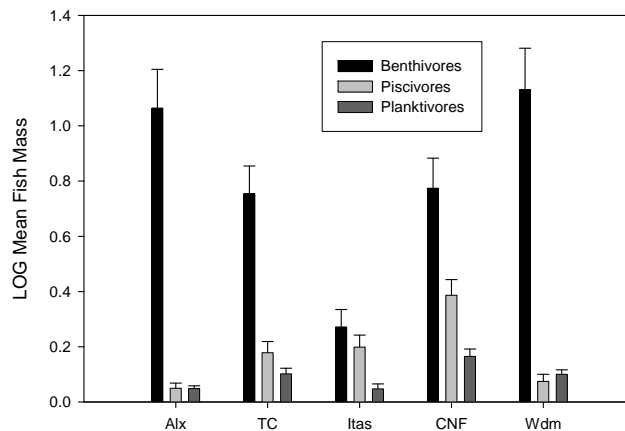

Figure 2.   Log$_{10}$- transformed mean fish mass of benthivores, piscivores, and planktivores collected from shallow lakes in 5 study regions in Minnesota during 2009. X-axis labels depict each of 5 study areas, Alx = Alexandria, TC = Twin Cities/Metro, Itas = Itasca State Park and surrounding, CNF = Chippewa National forest, and Wdm = Windom. Vertical bars indicate 1 SE.

NESTING ECOLOGY OF RING-NECKED DUCKS IN NORTHERN MINNESOTA

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SUMMARY OF FINDINGS

We have completed 2 years of field work on this research project. Thus far, we have searched 75 wetlands, located 38 nests, marked 22 hens, and followed 15 broods. We have searched lakes with (16%) and without (84%) boat accesses, 48-1583 m from roads, near both dirt (48%) and paved roads (52%), with houses (49%) and without houses (51%). Nest success is within the range of previous reports for north-central Minnesota. Hen and brood survival estimates require additional data collection to enable interpretation.

INTRODUCTION

The ring-necked duck (*Aythya collaris*) is a characteristic and important species for the Laurentian Mixed Forest province of Minnesota (MN DNR 2006). Recent surveys of 14 lakes important for ring-necked duck breeding 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 to enable a more informed interpretation of these local survey data and to understand how vital rates affect population growth of ring-necked ducks in the forest. These data are pertinent given the increasing development and recreational use in the forest (MN DNR 2006) and predictions that the spruce-fir forest will shift north of Minnesota as a result of global climate change (Iverson and Prasad 2001).

OBJECTIVES

1. To obtain baseline information on ring-necked duck nest success, hen survival, and brood survival before fledging in the Laurentian forest; and
2. To examine how these vital rates vary along a gradient of human development and recreational use (e.g., number of dwellings, boat access, proximity to roads).

STUDY AREA

The study area is approximately 65 km x 65 km and lies in the heart of the Laurentian mixed forest province of Minnesota. This area is characterized by mixed coniferous and hardwood forest interspersed with lakes. Wetlands in the area commonly have wild rice (*Zizania palustris*) or other emergent vegetation, sedges (*Carex* spp.), and floating bog mats along the margins.

METHODS

We searched for ring-necked duck nests in the spring and summer of 2008 and 2009. 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 2004-2009 and from 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 (GAP types 12 and 13 surrounded by GAP types 10, 14, and 15). We identified 103 lakes within a 25 mile radius of Bemidji with similar land cover attributes to those used in the 2 surveys and also searched the 6 lakes within our study area which had pairs or lone males in the ring-necked

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duck survey. In 2008, we searched 39 lakes. In 2009, we scouted 101 wetlands in early spring and focused nest searching efforts on the 36 wetlands where ring-necked ducks were seen. Scouting wetlands for ring-necked ducks before nest searching improved the efficiency of searches. We searched lakes with and without boat accesses (16% and 84%, respectively), 48-1583 m from roads, near both dirt (48%) and paved roads (52%), both with and without houses (49 and 51%, respectively).

To locate ring-necked duck 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. At each nest and at one random point located 25 m from 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 (*Typha* spp.), sedge), and distance to open water. Wetland size, distance to roads and dwellings, and wetland class were determined in GIS for use in models of nest survival.

Late in incubation, we trapped hens on nests with Weller traps (Weller 1957) to attach radiotransmitters. Because a surgical transmitter attachment method might be disruptive to incubating hens, we tried a bib-type transmitter attachment method which had been used with previous success in wood ducks (*Aix sponsa*; 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 in 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. We released birds at the edge of the wetland. Nests were monitored every 4-7 days to determine fate (abandoned, depredated, or successful).

After nests hatched, we monitored broods every 3-4 days. At each observation, we counted the ducklings present and aged them based on plumage characteristics (Gollop and Marshall 1954). Broods were monitored for 60 days or until total brood loss occurred. We considered hens to have lost their entire brood if hens were observed without any ducklings for 5 observations or if the hen was found >10 miles from the nesting lake. We continued to monitor hens after the brood-rearing period to examine hen survival until migration using the Kaplan-Meier method (Kaplan and Meier 1958).

RESULTS

Nest Survival

We searched for nests on 39 wetlands a total of 73 times between 22 May-22 July 2008 and 36 wetlands 54 times between 29 May-22 July 2009. We located 18 and 20 nests on 10 wetlands each year. The return per unit effort (i.e., #nests found/search) in 2009 was greater than the first year. 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 was 9.1 $\pm$ 0.6 (range: 7-15, n = 12) for nests known to be complete and 86.6 $\pm$ 0.1% of eggs (n = 109 eggs) hatched in successful nests. In 2009, 7 nests hatched, 9 were depredated after they were found, and 4 were abandoned, with at least 2 cases of abandonment likely due to trapping. The average clutch size was 8.3 $\pm$ 0.3 (range: 7-11, n = 19 nests, 158 eggs) and 89.5 $\pm$ 0.6% of the eggs hatched in nests that were successful. Ring-necked ducks nests were found predominately in leatherleaf (*Chamaedaphne calyculata*; 35%) and sedge (65%). Mayfield nest success for 35-days of laying and incubation (Mendall 1958, Mayfield 1975) was 29.6% in 2008 and 26.5% in 2009. Three females that lost their nest early in the season were later seen with males but no evidence of re-nesting was detected.

Hen Survival

We put transmitters on 8 hens in 2008 and 14 hens in 2009. 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 documented 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 28 days post-marking and the other 33 days 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 (16, 18, 29, 31, 52, 80 days post-marking). Evidence obtained at the recovery site 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 in 2009 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. Of the 8 hens that did not die during the monitoring period: 2 abandoned their nest (1 likely due to trapping), 1 nest was depredated, and 4 hatched nests, with 1 hen fledging young. Hen survival through September was 0.73 for 2008 and 0.54 for 2009.

Brood Survival

In 2008, 7 radiomarked hens had broods and 1 additional hen, which we did not trap in time to give a transmitter, also had a brood (n = 8 broods, 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) stages (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. Another brood made it to the IC stage, but we did not trap the hen in time to give her a transmitter, so 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. Brood movements were also observed in 2009. For example, a hen moved her 3 (IC) young from the nesting wetland to another wetland (~1207 m) from which they fledged. In another instance, a hen and her brood of 6 (IB) were seen walking to another wetland ~364 m from their nesting wetland.

We also observed duckling adoption. In 2008, 1 hen lost her nest, but then was observed to be unambiguously associating with a brood of 4 IA ducklings. We saw 2 cases of creching. One brood of 4 at hatch was later seen as 8 at the IB stage. The other instance involved a female that hatched 7 young and was later seen with 9 young at the IB stage. Two females left their nesting wetland after they lost their broods and were seen with 1 or 2 other hens and a brood on another wetland. In these situations, the hens without the broods were alert and protective of the other hens' brood as if it was their own. Young and some females were not marked, so uncertainty existed with regard to the relationships among young and females.

DISCUSSION

Our success finding nests has been comparable to that in other studies of ring-necked ducks (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 were within the range of Eberhardt's estimates of 44% (range 17-88%) based on 188 nests. The causes of nest failure in our study (24% flooding and 76%

depredation) were also similar to those of other studies (flooding 16-24%, depredation 67-80%, and desertion 5%, Mendall 1958, McAuley and Longcore 1989). Estimates of hatching success appear to be slightly 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, which may have chilled eggs and flooded nests.

Our hen survival rates for the period June-September 2008 were lower than those reported for hen mallards during April-September (0.60, Blohm et al. 1987, and 0.67, Brasher et al. 2006, 0.80, Cowardin et al. 1985). Our brood survival rates also appear 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%. Reliable estimation of brood survival was difficult in our study due to brood amalgamation and adoption. The degree to which these phenomena occur in ring-necked ducks is unknown. Creching has only been reported in ring-necked ducks once before by Toft et al. (1984) in the subartic taiga and was considerably lower (0.8% of broods) than in our study (20% of broods creched).

In 2010, we hope to collect additional data on nest success, hen survival, and brood survival. Results will be reported in future Summaries of Wildlife Research Findings.

ACKNOWLEDGMENTS

LITERATURE CITED

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

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

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

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

Gollop, J. B. and W. H. Marshall. 1954. A guide to aging duck broods in the field. Unpublished report. Mississippi Flyway Council.

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

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

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

Mayfield, H. F. 1975. Suggestions for calculating nest success. The Wilson Bulletin 87(4):456-466.McAuley, D. G., and J. R. Longcore. 1989. Nesting phenology and success of ring-necked ducks in east-central Maine. Journal of Field Ornithology 60:112-119.

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

Mendall, H. L. 1958. Ring-necked duck in the Northeast. University of Maine, Orono.

Minnesota Department of Natural Resources. 2006. A Vision for Wildlife and its Use – Goals and Outcomes 2006 – 2012, Minnesota Department of Natural Resources, St. Paul, Minnesota, USA, http://files.dnr.state.mn.us/aboutdnr/reports/strategic-documents/wildlife-stategic-plan05-12.pdf Last accessed 17 February 2010.

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

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

Sorenson, M. D. 1989. Effects of neck collars on female redheads. Journal of Field Ornithology 60(4):523-528.

Toft, C. A., D. L. Trigger, and H.W. Murdy. 1984. Seasonal decline in brood sizes of sympatric waterfowl (*Anas* and *Aythya*, Anatidae) and proposed evolutionary explanation. Journal of Animal Ecology 53:75-92.

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

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

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

2009 RING-NECKED DUCK BREEDING PAIR SURVEY

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SUMMARY OF FINDINGS

A pilot study was conducted in 2004 – 2006 to develop a survey for Minnesota's ring-necked duck (*Aythya collaris*) resident breeding population because little was known about the distribution and abundance of breeding ring-neck ducks in the state. We employed the survey design and methods developed during the pilot study (Zicus et al. 2008) to estimate the breeding population in 2007. In 2008 and 2009, we surveyed only 3 of 6 geographic strata and 2 of 4 habitat classes due to budget limitations. Helicopter-based counts in 2009 entailed 6 survey-crew days from 5 – 12 June totaling ~39 hrs of flight time. In 2009, the resident breeding population for the 3 geographic strata was estimated to be 11,000 indicated breeding pairs and 23,000 birds, which are similar to the 2008 estimate and the recalculated estimates for 2006 and 2007.

INTRODUCTION

Growing concern among biologists about the status of ring-necked ducks in Minnesota prompted the initiation of a pilot study to develop a breeding pair survey (Zicus et al. 2008). At the time, little was known about the breeding distribution and abundance of resident ring-necked ducks in Minnesota. Concerns were raised, in part, due to counts from 10 wetlands in the Bemidji area, which have shown a ~70% decline in ring-necked duck breeding pairs since 1969 (Zicus et al. 2004). Counts from this geographically limited survey suggest that the Minnesota population may be declining despite continental increases (U.S. Fish and Wildlife Service 2008). Additionally, the species was identified as a forest indicator because of its unique habitat associations (Minnesota Department of Natural Resources 2006). The importance of this species to Minnesota is also reflected in the number of ring-necked ducks harvested annually, often the 3rd most common duck taken by hunters (U.S. Fish and Wildlife Service, unpublished reports). The primary objectives of this survey have been to estimate breeding pair numbers and monitor population trends in northern Minnesota.

METHODS

Number of breeding pairs and population size within a stratified random sample of survey plots have been estimated using 2 stratification variables: (1) Ecological Classification System (ECS) sections; and (2) presumed nesting-cover availability (i.e., a surrogate for predicted breeding ring-necked duck density, Zicus et al. 2008). The pilot study and the first year of the operational survey (2007) were restricted to an area believed to be primary breeding range of ring-necked ducks for logistical efficiency (Zicus et al. 2008) and included 6 ECS sections (Figure 1). In 2008 and 2009, 3 of the ECS sections were dropped from the survey (Figure 1). Public Land Survey (PLS) sections (~2.6-km^2 plots, range = 1.2 – 3.0 km^2) were used as primary sampling units. The PLS sections at the periphery of the survey area that were <121 ha in size were removed from the sampling frame to reduce the probability of selecting these small plots.

We used the same habitat class definitions that were used for stratification in 2006 (Table 1; Zicus et al. 2008). Similar to 2006, in 2008 and 2009, a stratified sampling design was used to estimate breeding ducks in the best ring-necked duck habitat (habitat class 1 and 2 plots). The sampling frame consisted of 6 strata (i.e., 3 ECS sections x 2 habitat classes, Figure 1A), and we proportionally allocated 175 plots to the 6 strata. In previous surveys we also used a 2-stage simple random sampling design to estimate population size in the remainder of the survey area (habitat class 3 and 4 plots). Although habitat class 3 and 4 plots provided information on

use by ring-necked ducks of what we consider to be poorer quality habitat, only 8.6% of the breeding pairs were found there in 2006 and 2007 (Rave et al. 2008). When survey funds were reduced, a decision was made to not survey these plots where few ducks were expected to be found. For each plot, location, date, and time were recorded as were all ring-necked ducks observed on study plots and their sex and social status (Zicus et al. 2008). We considered pairs, lone males, and males in flocks of 2–5 to indicate breeding pairs (IBP; J. Lawrence, MNDNR, personal communication). The resident breeding population in the survey area was considered to be twice the IBP plus the number of lone females, flocked females, mixed sex groups, and single-sex groups >5 birds. We used the R library survey (Lumley 2009, R Development Core Team 2009) to estimate IBP and resident breeding population totals for habitat class 1 and 2 plots in each ECS section and the entire survey area.

RESULTS

In 2009, plots were well distributed throughout the study area (Figure 1B). Most plots (104) were located in the Northern Minnesota Drift and Lake Plains section, while the fewest plots (20) were located in the Lake Agassiz, Aspen Parklands section (Table 2). The sampling rate was higher in the Lake Agassiz, Aspen Parklands section than the other 2 ECS sections (5.9% versus 1.4% and 1.5%; Table 2). We were unable to survey 1 of the 175 plots in the Northern Minnesota Drift and Lake Plains section due to access limitations for 1 plot at the National Guard's Camp Ripley in Little Falls, Minnesota.

The survey was conducted 5–12 June and entailed 6 survey-crew days totaling ~39 hrs of flight time. A total of 273 ring-necked ducks were observed in 57 (33%) of 174 plots (Table 3). By habitat type, birds were detected on 31 (34%) of habitat class 1 plots and 26 (31%) of habitat class 2 plots. Overall, counts on occupied plots ranged from 1 to 19 birds (median = 4 birds/plot). Numbers of IBP on occupied plots ranged from 0 to 8 (median = 3 IBP/plot). Numbers of birds on occupied plots ranged from 1 to 23 ducks (median = 6 breeding birds/plot). Of the birds observed, 61% were classified as pairs, 17% lone males, 16% flocked males, 5% mixed groups, and <1% lone females. Of IBP, 47% were classified as pairs, 27% lone males, and 25% flocked males (Figure 2). These IBP ratios suggest that survey timing was reasonably good for estimating the resident breeding population.

Estimated IBP in the survey area was 10,947 pairs (SE = 1,563; Table 4, Figure 3A). The estimated resident breeding population of ring-necked ducks in the survey area was 19,488 birds (SE = 3,240; Table 4, Figure 3B). Because of sampling frame changes in 2008 and 2009, estimates from 2006 and 2007 were re-calculated with a 3 ECS sampling frame. Data from 2004 and 2005 were not re-calculated, because habitat classifications have also changed since those surveys were conducted. Estimates (IBP and breeding population) from 2009 were similar to 2006 and slightly higher than 2007 and 2008 but were within the error of prior surveys. The resident breeding population ranged from a high of 7,064 pairs and 14,948 breeding birds in the Northern Minnesota Drift and Lake Plains section to a low of 436 pairs and 871 breeding birds and in the Lake Agassiz, Aspen Parklands section (Table 5).

DISCUSSION

A number of trade-offs were involved in reducing the survey scope. By limiting the survey to 3 ECS, we are no longer monitoring populations in northeastern Minnesota. Birds in the area dropped from the survey are at relatively low densities; however, this area is quite large and represented approximately 30% of the resident breeding population in Minnesota based on surveys conducted in 2004 – 2007 (Table 5). Although we lost information on distribution and abundance by dropping 3 ECS, we have gained precision for the area that was sampled with reduced standard errors for the estimates. We also dropped the habitat class 3 and 4 plots. Dropping these plots allows us to focus where greater numbers of birds should be located

based on the presence of suitable nesting habitat. These plots represented 12% of the population estimates from 2006 and 2007 (Rave et al. 2008).

The resident breeding population appears to be relatively stable in the few years that this survey has been conducted, remaining between 18,000 and 23,000 breeding birds based on the estimates for the 3 ECS; however, many additional years are needed to detect population trends. Further, the survey was designed to estimate numbers of breeding ring-necked ducks and monitor population trends and as such is not optimized for detecting changes in the size of the resident population. Additionally, the survey is now focused on some of the best nesting habitat in the state. We do not know how this will affect our ability to monitor this resident breeding population.

The survey was also not designed explicitly to describe the distribution of resident breeding ring-necked ducks, but observations accumulated thus far have improved our knowledge of ring-necked duck distribution in the survey area (Figure 4). Most of the IBP and breeding population to date have been located along the north and northwest margin of the Northern Minnesota Drift and Lake Plains section. Another concentration of breeding ring-necked ducks is found at Agassiz National Wildlife Refuge in the center of the Lake Agassiz, Aspen Parklands section. From 2005 through 2008, very few ring-necked ducks have been observed along the southern margin of the study area, although there have been a number of survey plots in this area. In 2009, we did find a number of ring-necked ducks in the southern portion of the Minnesota and Northeast Iowa Morainal ECS (Figure 4). This survey is planned to continue in 2010.

ACKNOWLEDGMENTS

LITERATURE CITED

Giudice, J. H., J. R. Fieberg, M. C. Zicus, D. P. Rave, and R. G. Wright. 2010. Cost and precision functions for aerial quadrat surveys: A case study of ring-necked ducks in Minnesota. Journal of Wildlife Management 74(2):342 – 349.

Lumley, T. 2009. Survey analysis in R. <http://faculty.washington.edu/tlumley/survey/>. Accessed 13 April 2010.

Minnesota Department of Natural Resources. 2006. *A vision for wildlife and its use – goals and outcomes 2006 – 2012*. Minnesota Department of Natural Resources, St. Paul.

R Development Core Team. 2009. R: a language and environment for statistical computing. R Foundation for Statistical Computing, Vienna, Austria. http://www.R-project.org.

Rave, D. P., M. C. Zicus, J. R. Fieberg, J. H. Giudice, and R. G. Wright. 2008. Minnesota's ring-necked duck breeding pair survey. Pages 469 – 482 in M. W. DonCarlos, R. O. Kimmel, J. S. Lawrence, and M. S. Lenarz, editors. Summaries of wildlife research findings 2007. Minnesota Department of Natural Resources, St. Paul.

Smith, G. W. 1995. A critical review of the aerial and ground surveys of breeding waterfowl in North America. National Biological Service, Biological Science Report 5, Washington, D.C.

U. S. Fish and Wildlife Service. 2008. Waterfowl breeding population and habitat survey. U. S. Fish and Wildlife Service Division of Migratory Bird Management, Laurel, Maryland. http://mbdcapps.fws.gov/.

Zicus, M. C., R. T. Eberhardt, J. Dimatteo, and L. L. Johnson. 2004. Bemidji area ring-necked duck survey. Pages 169 – 183 *in* M. W. DonCarlos, R. O. Kimmel, J. S. Lawrence, and M. S. Lenarz, editors. Summaries of wildlife research findings 2003. Minnesota Department of Natural Resources, St. Paul.

Zicus, M. C., D. P. Rave, J. R. Fieberg, J. H. Giudice, and R. G. Wright. 2008. Distribution and abundance of Minnesota breeding ring-necked ducks Aythya collaris. Wildfowl 58:31 – 45.

Table 1. Habitat classes assigned to Public Land Survey section plots in the Minnesota ring-necked duck breeding pair survey area, June 2004 – 2009.

Habitat class	Definition[a]		Percent of survey area			
	2004	2005 - 2008[b]	2004	2005	2006-2007	2008-2009
1	Plots with > the median amount of MNGAP class 14 and/or 15 cover within 250 m of and adjacent to MNGAP class 12 cover (i.e., high pair potential).	Plots with > the median amount of MNGAP class 10, 14, and/or 15 cover within 250 m of and adjacent to MNGAP class 12 and/or 13 cover (i.e., high pair potential).	15.3	24.5	21.5	70.7
2	Plots with < the median amount of MNGAP class 14 and/or 15 cover within 250 m of and adjacent to MNGAP class 12 cover (i.e., moderate pair potential).	Plots with < the median amount of MNGAP class 10, 14, and/or 15 cover within 250 m of and adjacent to class 12 and/or 13 cover (i.e., moderate pair potential).	15.3	24.5	21.5	29.3
3	Plots with no MNGAP class 14 and/or 15 cover that include MNGAP class 12 cover that is within 250 m of a shoreline (i.e., low pair potential).	Plots with no MNGAP class 10, 14, and/or 15 cover that include class 12 and/or 13 cover that is within 100 m of a shoreline (i.e., low pair potential).	25.2	7.7	13.5	0.0
4	Plots with no MNGAP class 14 and/or 15 cover and no MNGAP class 12 cover within 250 m of a shoreline (i.e., no pair potential).	Plots with no MNGAP class 10, 14, and/or 15 cover and no class 12 and/or 13 cover within 100 m of a shoreline (i.e., no pair potential).	44.2	43.3	43.5	0.0

[a]Plots are Public Land Survey sections. MNGAP = Minnesota GAP level 4 land cover data. Class 10 = lowlands with <10% tree crown cover and >33% cover of low-growing deciduous woody plants such as alders and willows. Class 12 = lakes, streams, and open-water wetlands.   Class 13 = water bodies whose surface is covered by floating vegetation. Class 14 = wetlands with <10% tree crown cover that is dominated by emergent herbaceous vegetation such as fine-leaf sedges. Class 15 = wetlands with <10% tree crown cover that is dominated by emergent herbaceous vegetation such as broad-leaf sedges and/or cattails.

[b]Habitat class definitions in 2005 – 2009 were the same, but MNGAP class 10, 14, and 15 cover associated with lakes having a General or Recreational Development classification under the Minnesota Shoreland Zoning ordinance was not considered nesting cover in 2006 – 2009.

Table 2. Sampling rates in the habitat class 1 and 2 strata by Ecological Classification System (ECS) section for Minnesota's ring-necked duck breeding-pair survey, June 2004 – 2009.

ECS section	No. of plots[a]				No. of plots surveyed (Sampling rate [%])				
	2004	2005	2006-2007	2008-2009	2004	2005	2006-2007	2008	2009
W & S Superior Uplands[b]	1,638	2,461	2,218	-	18 (1.1)	22 (0.9)	20 (0.9)	-	-
Northern Superior Uplands	1,810	4,648	4,209	-	13 (0.7)	36 (0.8)	33 (0.8)	-	-
N Minnesota & Ontario Peatlands	1,817	2,737	2,389	-	26 (1.4)	35 (1.3)	30 (1.3)	-	-
N Minnesota Drift & Lake Plains	5,048	8,383	7,145	7,145	78 (1.5)	94 (1.1)	77 (1.1)	108 (1.5)	104 (1.5)
Minnesota & NE Iowa Morainal	3,510	4,033	3,561	3,561	50 (1.4)	35 (0.9)	32 (0.9)	53 (1.5)	51 (1.4)
Lake Agassiz, Aspen Parklands	316	363	340	340	15 (4.7)	8 (2.2)	8 (2.4)	13 (3.8)	20 (5.9)

[a]Number of Public Land Survey sections in the ECS section(s).
[b]Western and Southern Superior Uplands sections combined due to the small area of the Southern Superior Uplands occurring in the survey area.

Table 3. Survey results for habitat class 1 and 2 strata in the Minnesota ring-necked duck breeding pair survey area, June 2004 – 2009.

Year	No. of plots surveyed	No. plots with birds (%)	Birds[a]			IBP[b]			Resident breeding birds[c]		
			Total	Per plot	Per occupied plot	Total	Per plot	Per occupied plot	Total	Per plot	Per occupied plot
2004	200	50 (25)	278	1.39	5.56	160	0.81	3.20	353	1.77	7.06
2005	230	37 (16)	147	0.64	3.97	92	0.43	2.49	218	0.95	5.89
2006	200	50 (25)	279	1.40	5.58	167	0.85	3.34	375	1.88	7.50
2007	200	52 (26)	152	0.76	2.92	137	0.72	2.63	296	1.48	5.69
2008	174	58 (33)	296	1.70	5.10	173	0.99	2.98	364	2.09	6.28
2009	174	57 (33)	273	1.57	4.79	173	0.99	3.04	362	2.08	6.35

[a]Total number of ring-necked ducks counted during the survey.
[b]The number of indicated breeding pairs (IBP) is the sum of the pairs, lone males, and males in flocks of 2–5.
[c]The total resident breeding population in the survey area was considered to be twice the IBP plus the number of lone females, flocked females, mixed sex groups, and single-sex groups >5 birds.

Table 4. Estimated indicated breeding pairs (IBP) and resident breeding population size in the habitat class 1 and 2 strata in the Minnesota ring-necked duck breeding pair survey area, June 2004 – 2009.

	IBP (CV[%])			Resident breeding population (CV[%])	
Year	6 ECS[a]	3 ECS[b]		6 ECS[a]	3 ECS[b]
2004	9,443 (17.8[c])	-		20,321 (18.1[c])	-
2005	7,496 (20.0[c])	-		17,279 (21.5[c])	-
2006	14,770 (17.6[c])	9,851 (23.8)		32,621 (17.4[c])	21,849 (23.1)
2007	12,787 (17.7)	8,705 (19.9)		26,026 (17.5)	17,863 (19.5)
2008	-	9,439 (16.8)		-	19,488 (16.6)
2009	-	10,947 (14.3)		-	22,987 (15.0)

[a]Population estimates were based on a stratified random sample of habitat class 1 and 2 Public Land Survey (PLS) sections in 12 strata (2 habitat classes and 6 Ecological Classification System [ECS] sections).
[b] Population estimates were based on a stratified random sample of habitat class 1 and 2 Public Land Survey (PLS) sections in 6 strata (2 habitat classes and 3 Ecological Classification System [ECS] sections). Population estimates were not adjusted for 2004 and 2005, because the habitat classifications have also changed since those surveys were conducted.
[c]Variance estimate is biased low because no birds were observed in one or more strata. As a result, the confidence interval is too narrow and the CV is optimistic.

Table 5. Estimated indicated breeding pairs (IBP) and resident breeding population by Ecological Classification System (ECS) section in the habitat class 1 and 2 strata in the Minnesota ring-necked duck breeding pair survey area, June 2005 – 2009.

	IBP (CV [%])				
ECS section	2005	2006	2007	2008	2009
W & S Superior Uplands[b]	444 (99.5[c])	669 (59.1)	671 (99.6)	-	-
Northern Superior Uplands	1,169 (46.8)	2,679 (33.7)	2,694 (46.5)	-	-
N Minnesota & Ontario Peatlands	239 (54.1[c])	1,572 (34.7)	717 (46.5)	-	-
N Minnesota Drift & Lake Plains	3,490 (33.0)	6,334 (31.5)	5,686 (26.0)	4,948 (24.6)	7,064 (17.1)
Minnesota & NE Iowa Morainal	918 (43.6)	2,102 (53.9)	2,118 (38.8)	3,689 (26.0)	3,449 (28.4)
Lake Agassiz, Aspen Parklands	1,235 (40.1[c])	1,414 (35.2)	902 (40.9)	803 (38.4)	436 (35.5)

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

Table 5. Continued.

ECS section	Resident breeding population (CV [%])				
	2005	2006	2007	2008	2009
W & S Superior Uplands[b]	889 (99.5[c])	1,338 (59.1)	1,342 (99.6)	-	-
Northern Superior Uplands	2,339 (46.8)	5,357 (33.7)	5,388 (46.5)	-	-
N Minnesota & Ontario Peatlands	477 (54.1[c])	4,076 (42.3)	1,434 (46.5)	-	-
N Minnesota Drift & Lake Plains	6,981 (33.0)	14,816 (29.6)	11,651 (25.4)	10,264 (24.3)	14,948 (18.2)
Minnesota & NE Iowa Morainal	4,122 (56.4)	4,204 (53.9)	4,236 (38.8)	7,377 (26.0)	7,170 (29.2)
Lake Agassiz, Aspen Parklands	2,471 (40.1[c])	2,829 (35.2)	1,976 (42.3)	1,846 (41.4)	871 (35.4)

[a]Western and Southern Superior Uplands sections combined due to the small area of the Southern Superior Uplands occurring in the survey area.
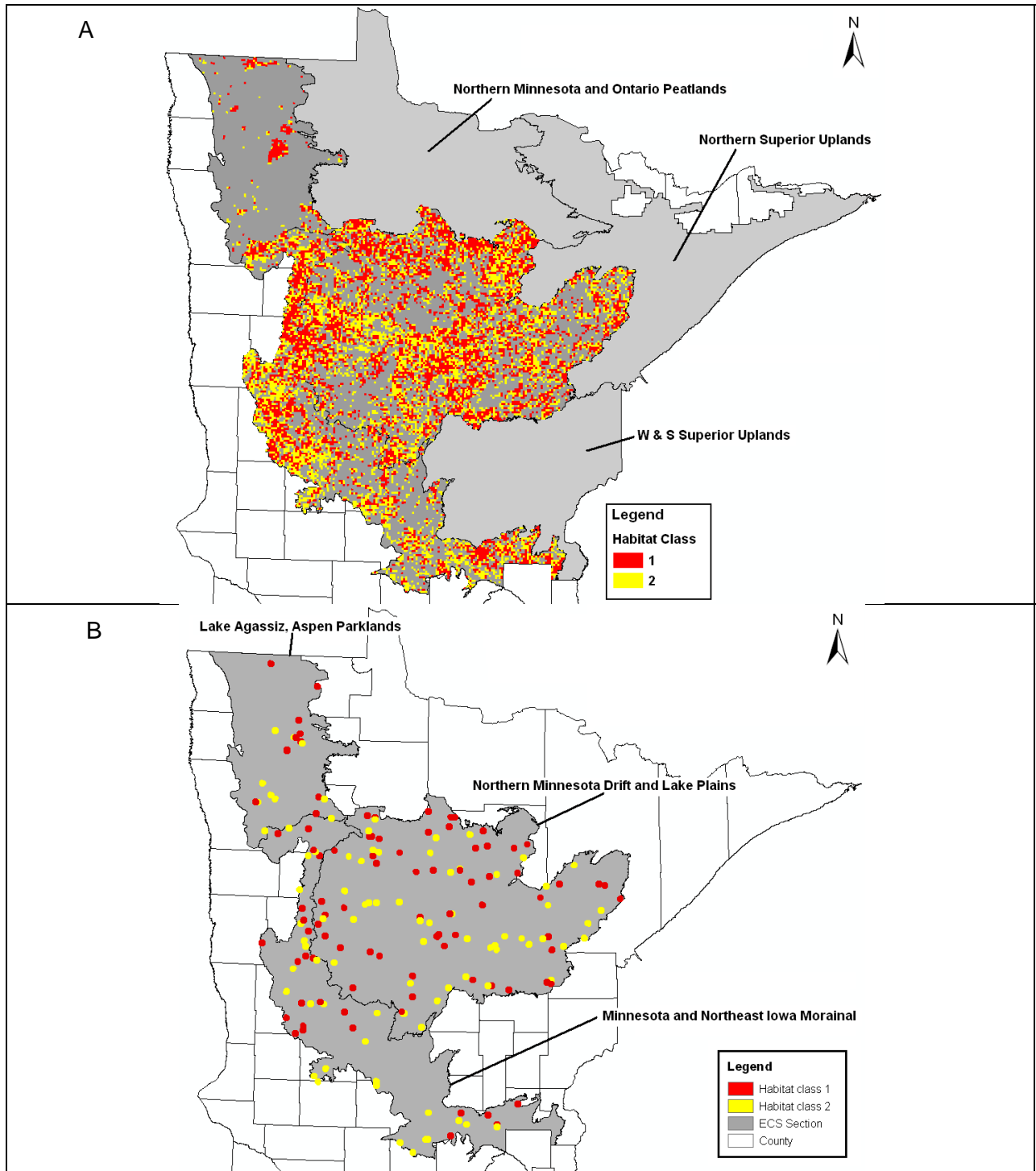
Figure 1. In the 3 Ecological Classification Section (ECS) sampling frame (A) all Public Land Survey (PLS) plots and (B) 2009 survey plots (enlarged for visibility) indicated by habitat class for Minnesota's ring-necked duck breeding pair survey.
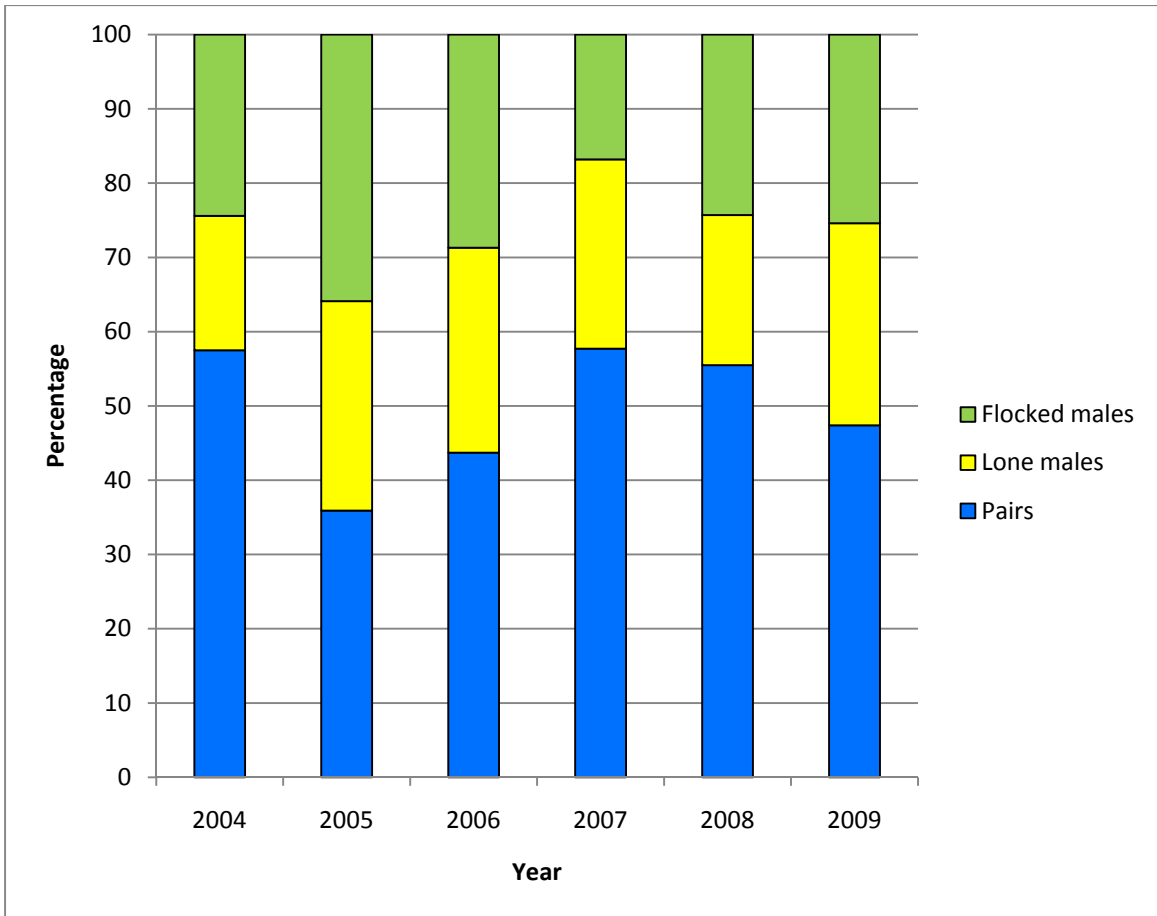
Figure 2. Social status of the indicated breeding pairs observed in the Minnesota ring-necked duck breeding pair survey area, June 2004 – 2009. Surveys were conducted 6 – 17 June 2004, 12 – 24 June 2005, 6 – 16 June 2006, 5 – 13 June 2006, 9 – 17 June 2008, and 5 – 12 June 2009.
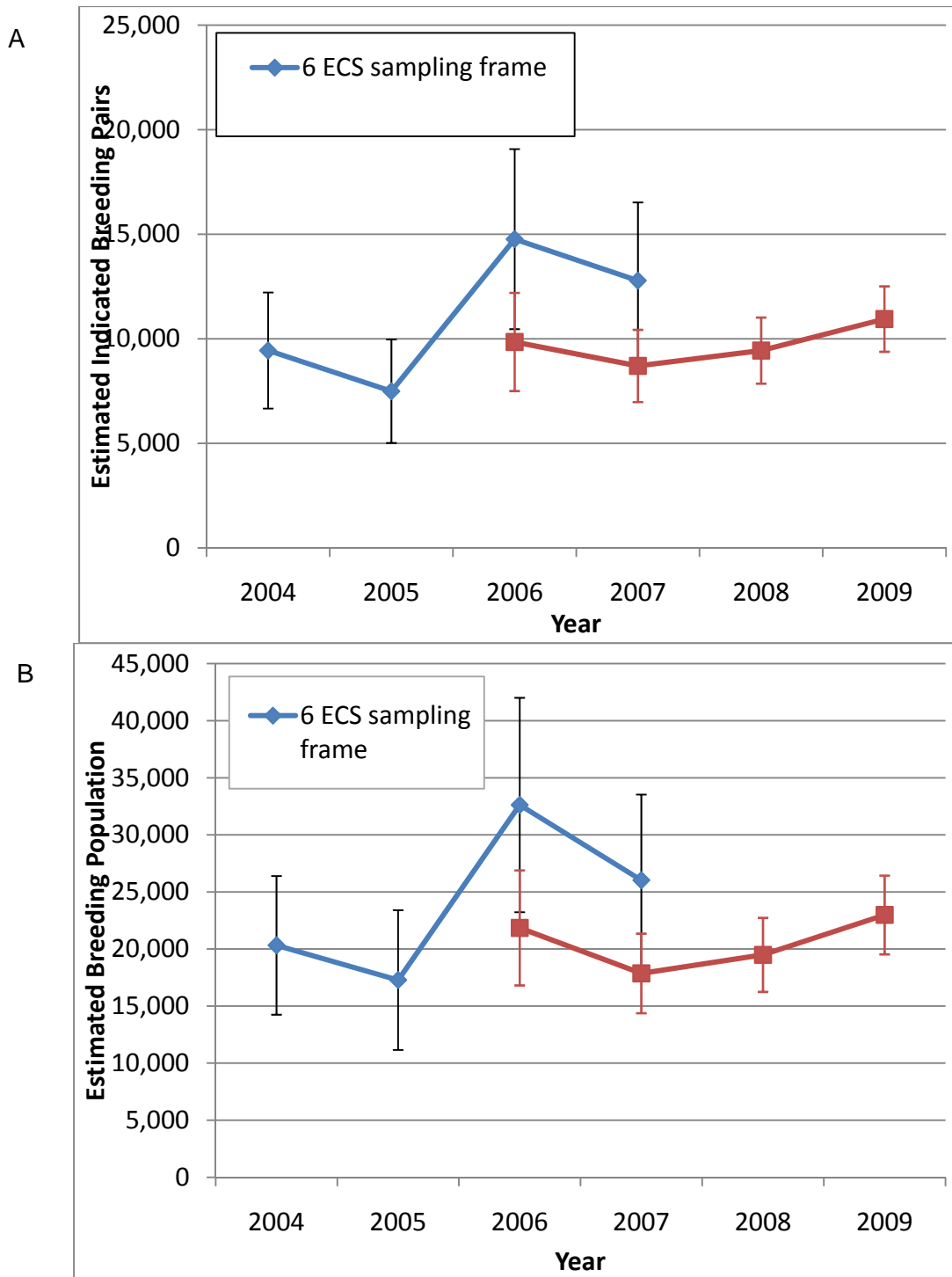
Figure 3. For the habitat class 1 and 2 strata (A) estimated indicated breeding pairs with SE bars and (B) estimated ring-necked duck resident breeding population with SE bars in the Minnesota ring-necked duck breeding pair survey area, June 2004 – 2009. Estimates were based on a stratified random sample of Public Land Survey (PLS) sections in habitat classes 1 and 2 for 6 Ecological Classification System (ECS) sections in 2004 – 2007 and for 3 ECS sections in 2008 and 2009. Estimates from 2006 and 2007 were recalculated using the same sampling frame as 2008 and 2009 (3 ECS instead of 6 ECS) for comparison; population estimates were not adjusted for 2004 and 2005, because the habitat classifications have also changed since those surveys were conducted.
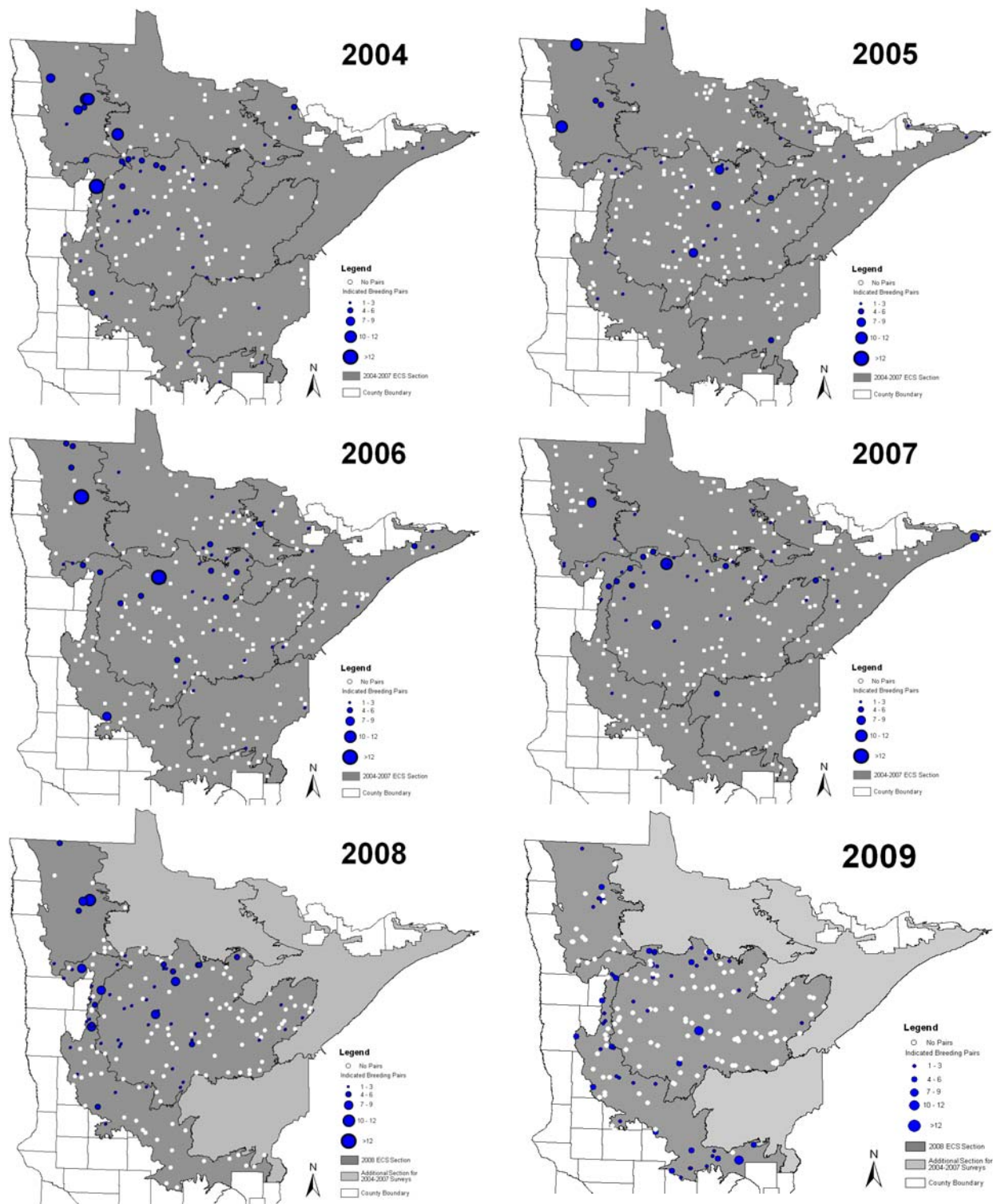
Figure 4. Plot locations and numbers of indicated breeding pairs (IBP) observed on survey plots in the Minnesota ring-necked duck breeding pair survey area in June 2009 (bottom right). White circles indicate plots where no indicated pairs were seen. Maximum number of indicated breeding pairs per plot was 8 pairs in 2009 (13 in 2004; 11 in 2005; 16 in 2006; 11 in 2007; 10 in 2008). The Ecological Classification System (ECS) sections are also shown.